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Camera methods for the assessment of coastal biodiversity in low visibility environments

A thesis submitted in fulfilment of the requirements for the degree

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By

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Summary

Coastal marine environments are important ecological, economic and social areas providing valuable services such as coastal protection, areas of recreation and tourism, fishing, climate regulation, biotic materials and biofuels. Marine renewable energy developments in the coastal environment are becoming a key objective for many countries globally. Assessing and monitoring the impacts of these developments on features, such as coastal biodiversity, becomes a difficult prospect in these environments due to the complexity of marine process at the locations in which these developments are targeted.

This thesis explores the main challenges faced when assessing biodiversity in dynamic coastal environments, in particular those susceptible to high levels of turbidity. Various underwater camera techniques were trialled in reduced visibility environments including baited remote underwater video (BRUV), drop-down video and hydroacoustic methods.

This research successfully refined BRUV guidelines in the North-East Atlantic region and identified key methodological and environmental factors influencing data collected BRUV deployments. Key findings included mackerel as the recommended bait type in this region and highlighting the importance of collecting consistent metadata when using these methods.

In areas of high turbidity, clear liquid optical chambers (CLOCs) were successfully used to enhance the quality of information gathered using underwater cameras when monitoring benthic fauna and fish assemblages. CLOCs were applied to both conventional BRUV camera systems and benthic drop-down camera systems. Improvements included image quality, species and habitat level identification, and taxonomic richness.

Evaluations of the ARIS 3000 imaging sonar and its capability of visualising distinguishing identifying features in low visibility environments for motile fauna showed mixed results with morphologically distinct species such as elasmobranchs much clearer in the footage compared to individuals belonging to finfish families.

A combined approach of optical and hydroacoustic camera methods may be most suitable for adequately assessing coastal biodiversity in low visibility environments.

Declaration Statements

Declarations

This work has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree.

Signed ....[redacted]........... (candidate)

Date: 01/05/2021

STATEMENT 1

This thesis is the result of my own investigations, except where otherwise stated. Where correction services have been used, the extent and nature of the correction is clearly marked in a footnote(s).

Other sources are acknowledged by footnotes giving explicit references. A bibliography is appended.

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Ethics Statement

Approval Number: SU-Ethics-Student-091017/284

Reference Number: STU_BIOL_71259_091017092553_1

Prior to conducting this research, Swansea University's Research Ethics and Governance Framework was read.

This project makes use and generates new data. This study did **NOT** pose a potential risk to the environment, such as the escape of invasive species, genetically modified organisms (GMO), work involving human or animal pathogens, environmental contaminants, radioactive material or active outdoor vegetation fires. This study also did **NOT** involve humans as the focus of research.

This study was not interventional; it did **NOT** involve the capture, handling, and confinement of living vertebrates or cephalopods.

No conflicts of interest are to be declared.

Signed

[redacted]

Robyn Jones

01/05/2021

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Chapter specific acknowledgements have been added to the end of each chapter.

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Abbreviations

ACI	Acoustic camera imagery
AIC	Akaike information criterion
ANOVA	One-way analysis of variance
AR	Autonomous recorders
ARIS	Adaptive resolution imaging sonar
AVCHD	Advanced video coding high definition
AVI	Audio video interleave
BRUV	Baited remote underwater video
BSH	Broad scale habitat
CAP	Canonical analysis of principal coordinates
CATAMI	Collaborative and Annotation Tools for Analysis of Marine Imagery
CBD	Convention on Biological Diversity
CEFAS	Centre for Environment, Fisheries and Aquaculture Science
CLOC	Clear liquid optical chamber
Cm	Centimetre
DDV	Drop-down video
DIDSON	Duel-frequency identification sonar
DISS	Digital image scanning sonar
EBM	Ecosystem Based Management

EIA	Environmental impact assessment
EPA	Environmental Protection Agency
ES	Environmental statement
EU	European Union
EUNIS	European Nature Information System
FAO	Food and Agriculture Association
FOV	Field of view
GEO BON	Group on Earth Observations Biodiversity Observation Network
GES	Good environmental status
GIS	Geographic information system
GLM	General / Generalized linear model
JNCC	Joint Nature Conservation Committee
KESS	Knowledge Economy Skills Scholarships
KHz	Kilohertz
L	Litre
LED	Light-emitting diode
m	Metre
MaxN	Maximum number of individuals
MBES	Multibeam echosounders
MHz	Megahertz

mm	Millimetre
MNCR	Marine Nature Conservation Review
MPA	Marine protected area
MSFD	Marine Strategy Framework Directive
n	Number
NEPA	U.S National Environment Policy Act
NGO	Non-government organisation
nMDS	Non-metric multi-dimensional scaling
PAM	Passive acoustic monitoring
PERMANOVA	Permutational multivariate of analysis of variance
PERMDISP	Permutational analysis of multivariate dispersions
PVC	Polyvinyl chloride
RMSE	Route mean square error
ROV	Remotely operated underwater vehicle
RUV	Remote underwater video
SAC	Special area of conservation
SACFOR	Abundance scale
SE	Standard error
SEA	Strategic environmental assessments
SD	Secure digital / Standard deviation

SIMPER	Similarity percentages
SPM	Suspended particulate matter
Spqx	Square pixels
SSS	Side-scan sonar
TSS	Total suspended sediment
UAV	Unmanned aerial vehicles
UK	United Kingdom
UNCLOS	United Nations Convention on the Law of the Sea
UVC	Underwater visual survey
VIF	Variance inflation factor
WFD	Water Framework Directive

General Introduction

Coastal marine environments are important ecological, economic and social areas (Martínez et al., 2007). Healthy marine ecosystems provide valuable services such as coastal protection, areas of recreation and tourism, fishing, climate regulation, biotic materials and biofuels (Liquete et al., 2013; Canonico et al., 2019). These services have led to coastal environments being centres of human activity (Martínez et al., 2007). Measures including Marine Protected Areas (MPAs) have been implemented in the North-East Atlantic Region in order to conserve, promote and monitor biodiversity in specific environments (Jones, 2012).

Marine renewable energy developments in the coastal environment are becoming a key objective for many countries globally, as they are considered a more sustainable alternative to the production of electricity and green fuels through energy resources such as tides, currents, waves and offshore wind (Gill, 2005; Linley et al., 2009). Turnover in the UK's low carbon economy now is valued at around £122 billion and has been growing at an average rate of over 7% per year since 2010 in nominal terms. Over £42 billion pounds has been invested in renewables, nuclear and carbon capture storage (DECC 2015) due growing concerns into the impacts of fossil fuels such as coal, oil and natural gas on the state of the marine environment and climate change (Cruz & Krausmann, 2013; Demirbas, 2009; Pelc & Fujita, 2002). The UK's target for offshore wind is to provide enough power to power all homes by 2030. Although accessing these nearshore environments is mostly favourable, many challenges still exist with regards to developing this technology in coastal areas, (Pelc & Fujita, 2002). Modifications to wave climates, flow patterns, and marine habitats, particularly through increased underwater noise and collision risk, are identified as key ecological issues (Bonar et al., 2015).

Assessing and monitoring the impacts and interactions of marine renewable energy developments on coastal biodiversity becomes a difficult prospect due to the complexity and dynamics of the marine environmental processes and renewable developments being monitored (Fox et al., 2018). Examples of these challenging processes include the direction and speed of prevailing winds, underwater visibility, large tidal ranges, high velocity currents and increased wave action. The restrictions of deploying damaging extractive survey methods close to infrastructure or in sensitive habitats protected under MPA management further limits the assessment and monitoring the biodiversity in these environments (Griffin et al. 2016). This uncertainty places a considerable burden on developers who must collect biological data through baseline and post-deployment monitoring programs under the Environmental Impact Assessment process (Fox et al., 2018).

Current research into monitoring methods around marine renewable energy developments has included remote sensing, eDNA (Canonico et al., 2019), unmanned vehicles (Verfuss et al., 2019), fish netting (Fox et al., 2018), and various optical and acoustic technology (Polagye et al., 2020). Advancements in camera research in the North-East Atlantic Region include both static and towed equipment for the monitoring of marine biodiversity. Configurations below and above water provide long- term baseline and impact- related data for development sites and energy converters, as well as monitor impacts of operational energy convertors on fish, mammal, and bird behaviour (Bicknell et al., 2016). The development of rapid, cost-effective and reliable remote underwater monitoring methods has been identified as crucial to supporting evidence-based decision-making by planning authorities and developers when assessing environmental risks and benefits of offshore structures. A recent example of such technological advancements in the North-East Atlantic region includes the novel underwater imaging system PelagiCam. This system allows for semi-automated monitoring of mobile marine fauna at offshore structures for streamline biological data acquisition. (Sheehan et al., 2019).

Objectives of this Thesis

The objectives of this research were to identify the main challenges faced when assessing coastal biodiversity in dynamic areas associated with or targeted for coastal renewable developments, and trial and test various non-destructive underwater camera methods in reduced under visibility environments associated with marine renewable energy developments.

During this PhD we trialled three underwater video techniques in reduced visibility coastal environments around the UK, predominantly around the South Wales coast with the aim of quantifying and improving the quality of information gained through underwater imagery. Such information included species enumeration, abundance and habitat classifications. These three methods included the refinement of baited remote underwater video (BRUV) techniques with the aim of providing an insight for future guidelines for the application of these methods in North-Eastern Atlantic coastal waters; the introduction of a clear liquid optical chamber to improve underwater image clarity for use in motile and benthic faunal assemblage assessments, and the use of ARIS 3000 imaging sonar for identifying fish assemblages and characterising benthos associated with various coastal habitats. The following research questions were applied to this thesis:

Chapter 1: What are the main challenges of assessing coastal biodiversity in the North-East Atlantic Region?

Chapter 2: What factors influence the information gathered by BRUV methods in coastal waters around the UK? *Published.*

Chapter 3: What is the influence of bait on remote underwater video observations in shallow-water coastal environments associated with the North- East Atlantic Region? *Published.*

Chapter 4: Is the identification of motile fauna in turbid, low visibility environments improved through the introduction of a clear liquid optical chamber to improve image clarity around UK coastal waters? *Published.*

Chapter 5: Is the assessment of subtidal benthic coastal habitats in turbid, low visibility environments improved through the introduction of a clear liquid optical chamber to improve image clarity around UK coastal waters? *Published.*

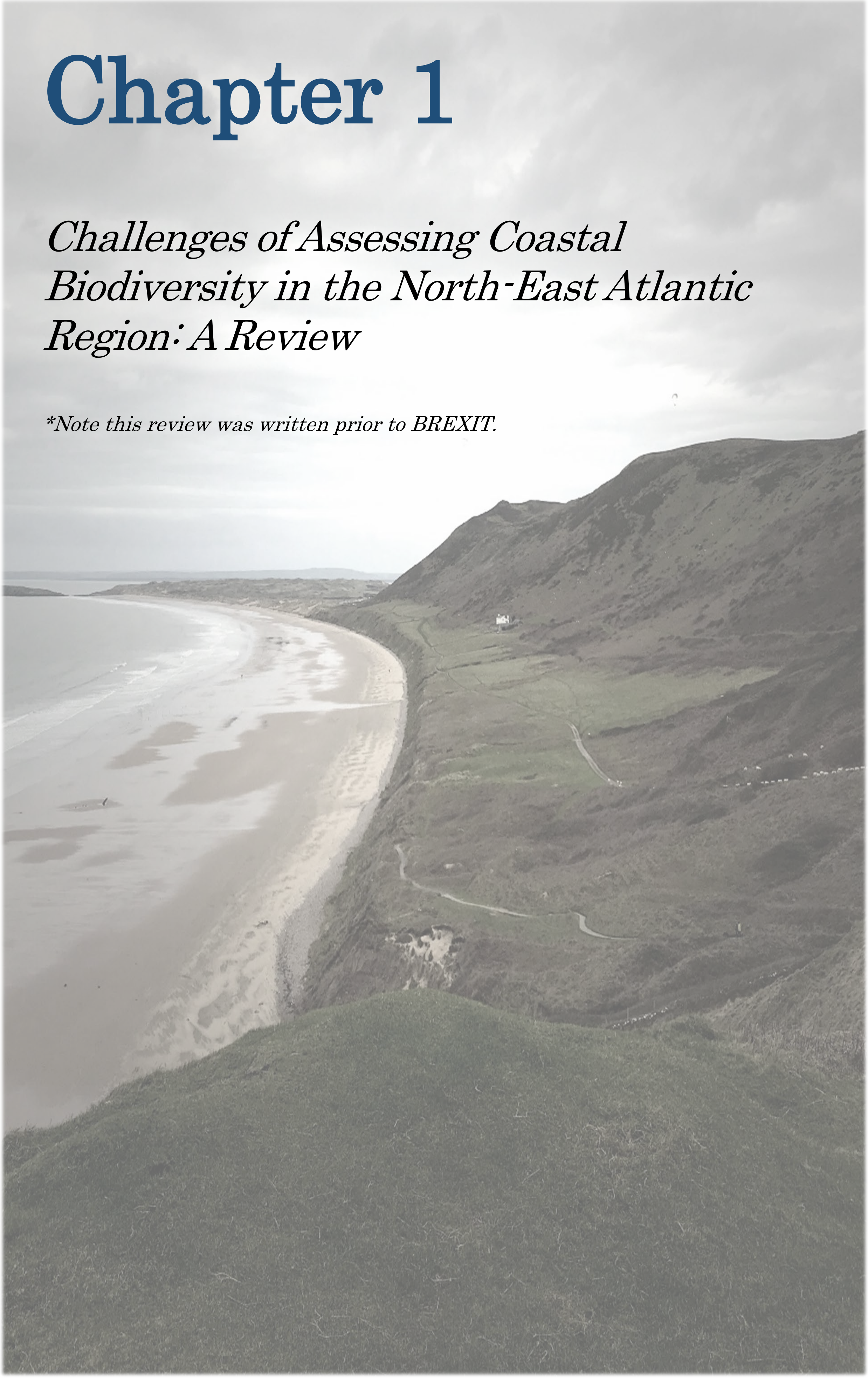
Chapter 6: What are the capabilities of the ARIS 3000 imaging sonar for identifying fish assemblages native to UK coastal waters?

Appendix I: What are the capabilities of the ARIS 3000 imaging sonar for characterising biogenic structures formed by *Sabellaria* in highly turbid environments? *Published.*

Chapter 1

Challenges of Assessing Coastal Biodiversity in the North-East Atlantic Region: A Review

**Note this review was written prior to BREXIT.*



Abstract

The marine environment provides a number of highly valued ecosystem services and goods for the human population. The need to adequately assess and monitor this environment has arisen from the increase in human dependence on these resources which it provides. This review explored the main survey methods used to assess marine biodiversity and discusses their advantages and disadvantages and recommends monitoring types depending on target biodiversity groups.

Challenges faced when assessing coastal environment in the North-East Atlantic region included, but not restricted to; accessibility to the target area; weather; high current velocities and turbidity levels; human, methodological and external sampling bias; analysis duration; cost-effectiveness and selectivity of gear. Techniques used to assess biodiversity may be split into extractive and non-extractive methods. Extractive methods such as trawling and benthic sediment grabbing are destructive in nature but are widely used as they are not limited by factors such as high current velocities and underwater visibility. In contrast to this, non-extractive methods such as underwater cameras and aerial surveys are limited by such factors, influencing accuracy in their assessment of coastal biodiversity.

With traditional extractive survey methods damaging the baseline environment and limited by their proximity to seabed infrastructure, the use of non-extractive equipment may be considered a preferable alternative for monitoring coastal biodiversity. In order for this option to be feasible and accurate, modifications and further research need to be made to these methods in their current form.

Keywords

Biodiversity assessments; Coastal environments; Environmental monitoring; Sampling techniques.

1.1. Introduction

The world's coastline covers a distance of approximately 1,634,701km (Martínez et al., 2007) and encompasses a variety of habitats each with their own unique organisms and ecosystems (Raffaelli, 2006). For this review, the definition of the coastal zone follows the JRC-IES Coastal Zone Technical Note which identifies a 10km buffer seaward from the coastline and a 2km buffer inland (Lavalle et al., 2011). The North-East-Atlantic region encompasses a coastline area of approximately 20,585km and includes the following features estuaries intertidal flats, and coastal lagoons (OSPAR 2000).

The Convention on Biological Diversity defines biodiversity in Article 2 as '*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems'* (United Nations, 1992). Biodiversity increases the ability of ecosystems to respond and adapt to changes in the environment from disturbances such as climate change and those anthropogenic in origin (Hoegh-Guldberg & Bruno 2010). Areas of increased biodiversity also provide valuable goods and services for the human population (Hiddink et al., 2008). The coastal zone is home to diverse flora and fauna, and it provides highly valued ecosystem services (Pakeman et al., 2017) including nutrient recycling, food production and coastal defences (Costanza et al., 1997).

For the purpose of this chapter, coastal biodiversity has been split into five main groups; fish; benthos (including intertidal habitats and marine flora); marine mammals; plankton and birds; however, techniques of assessing bird and plankton biodiversity will not be considered further in this review.

1.1.1. North-East Atlantic Coastal Biodiversity

1.1.1.1. Fish

It is estimated that approximately 78% of all marine fish species are found within the worlds continental shelf (Cohen, 1970). Coastal and nearshore habitats such as estuaries, coral reefs and marshes are important nursery areas which support high abundances and diversity of fish species (Beck et al., 2001). Furthermore, fish biodiversity within the world's continental shelves is essential for the sustainability of commercial fisheries (Hiddink et al., 2008). Within the North Atlantic Ocean, nearly 1100 species of fish are known, 600 of which are pelagic, while the rest is demersal (OSPAR 2000). Threats to fish biodiversity include overfishing, habitat loss, pollution and climate change (Gray, 1997). Fishery resources account for approximately 19% of human protein intake and are therefore considered an important commodity which needs conserving (Shao, 2009).

1.1.1.2. Benthos (Including Supratidal Dunes)

Within coastal ecosystems, benthic biodiversity is essential to ecological functions such as decomposition, nutrient recycling and nutrient production (Levin et al., 2001). High benthic biodiversity also has the potential to increase sediment stability in turn reducing erodibility and may also provide food for human consumption. Anthropogenic disturbance to benthic and intertidal communities may occur directly from actions such as fishing and coastal and offshore industrial developments directly on the seabed i.e., oil & gas, renewable energy (Snelgrove, 1997). Research has suggested that benthos subject to reduced disturbance levels show positive responses within three years for species richness, total abundance, and assemblage composition for certain indicator species (Sheehan et al 2013). Coastal habitats such as saltmarshes, seagrass meadows, reefs and supratidal sand dunes can also act as valuable natural coastal protection for the human population from events such as sea level rise and flooding (Arkema et al., 2013).

1.1.1.3. Marine Mammals

Marine mammals represent the smallest number of different species in the marine environment; however, they account for a far larger biomass in comparison to fish and benthos (Kaschner et al., 2011) and play key roles in ecosystems through predation (Schipper et al., 2009) and nurtrient recycling (Bowen 1997). Two species of seal and 32 different species of cetacean have been observed in the Atlantic Ocean (OSPAR, 2000). Cetaceans are classified as keystone indicator species (Harwood 2001) and are often considered high profile taxa due to their charismatic nature, capturing the public's attention when it comes to raising awareness of the marine environment (Parsons et al., 2015).

1.1.2. Importance of Assessing Coastal Biodiversity

The need to assess marine biodiversity has arisen from the increased human dependence and exploitation / development in the marine environment (Selig et al., 2013) through the services in which it provides. Such services include coastal protection, tourism, nutrient recycling, and food production including fisheries and aquaculture (Barbier et al., 2011; Costanza et al., 1997; Liquete et al., 2013). This in turn has led to the overexploitation of these areas (Heiskanen et al., 2016). Reducing this anthropogenic impact and monitoring the marine environment has therefore become a necessity.

Approaches to assessing and monitoring this diversity varies globally. Even with established monitoring systems, there is still variation in individual countries priorities, objectives and standards, causing difficulties to the decision-making process with regards to the coastal environment (Teder et al., 2006).

1.1.2.1. Protected species and habitats

The Strategic Plan for Biodiversity 2011-2020 developed by the Parties to the Convention on Biological Diversity (CBD) is a global framework with the aim of safeguarding biodiversity with the development of national targets (United Nations, 2011). Several legal instruments (conventions and directives) aim to protect and conserve the marine life in the North-east Atlantic Ocean. These include Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR), International Council for the Exploration of the Sea (ICES), EU Birds and Habitat Directives, North Atlantic Marine Mammal Commission (NAMMCO) and the BERN Convention (OSPAR 2000).

Within European countries of the North-East Atlantic region, the EU Marine Strategy Framework Directive (MSFD) sets out 11 qualitative descriptors for "good environmental status" (GES) (Andersen et al., 2014). This status not only recognises the state of species and habitat biodiversity but also includes aspects such as seafloor integrity and food webs (Andersen et al., 2014; Cochrane et al., 2016). The Directive came into force on 15 July 2008 and was transposed into UK law by the Marine Strategy Regulations 2010 (Department for Environment Food and Rural Affairs, 2014). Supporting this, the Water Framework Directive (WFD) has introduced an international commitment to assess the ecological status of transitional waters where fish assemblages are considered a vital ecological component (Coates et al., 2007).

The Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora, also known as the EU Habitats Directive, adopted in 1992, protects a range of threatened habitat types and species listed in Annex I and Annex II respectively (European Commission, 1992). These are often considered habitats of key species and indicators of biodiversity (Heiskanen et al., 2016). The protection of these Annex I and II habitats and species contributes to a network of Special Areas of Conservation (SACs) (European Commission, 1992). However, the balance of species and habitats protected under this directive is currently heavily weighted on the terrestrial side (77 terrestrial species within the UK Atlantic biogeographic region compared to 16 species within the UK Marine Atlantic region) (European Commission, 2018). These measures are just an example of the different policies in place to promote the protection of biodiversity, with individual nations and regional seas also having their own assessment and monitoring programmes in place (Heiskanen et al., 2016).

Specifically, within the UK for example, species and habitats which are strategically monitored fall within the following components: seabirds, marine mammals (cetaceans and seals), fish and cephalopods, benthic habitats, plankton, and ecosystem processes and functions (Joint Nature Conservation Committee, 2016; Noble-James et al., 2017). Further aspects such as water quality and pollution are also monitored. The UK Marine Biodiversity Monitoring Strategy aims to provide options for assessing different biodiversity components identified for the development of the MSFD. The principles of these monitoring strategies follow a risk-based approach to reduce the scale of monitoring required and prioritising monitoring activities. This strategy aims to ensure that species and habitats deemed 'high risk' and ecosystem components sensitive to pressure are a monitoring priority. Once determined, monitoring may be undertaken through sentinel monitoring of long-term trends (Type 1), operational monitoring of pressure-state relationships (Type 2) and / or investigative monitoring to determine management needs and effectiveness (Type 3) (Joint Nature Conservation Committee, 2016).

The Joint Nature Conservation Committee Marine Monitoring Handbook (Davies et al., 2001) provides guidance on method suitability, and states that monitoring methods in the marine environment should be chosen based on their; likelihood of damage to the target species and / or environment; ability to provide a type of measurement consistent with the objective of the target species or habitat; ability to measure the attribute across an appropriate range of conditions; ability to provide precise observations on scales of change; and within the budget available.

1.1.2.2. Ecosystem-Based Management

Ecosystem-based management (EBM) has been promoted as an efficient way of implementing environmental and water policies (Apitz et al., 2006; Rouillard et al., 2018). EBM '*takes into account the interconnectedness and interdependent nature of ecosystem components and emphasizes the importance of ecosystem structures and functions which provide a range of services'* (Curtin & Prellezo 2010). EBM aims to create management systems that better protect the dynamics and requirements of healthy ecosystems (Rouillard et al., 2018). Its importance is highlighted in the Millennium Ecosystem Assessment (2005), and its principles underpin policy developments such as the WFD (2000/60/EC) and the MSFD (2008/56/EC) (Holt et al., 2011). Examples of regional management measures that are based on this ecosystem approach include the Helsinki Commission Baltic Sea Action Plan (Backer et al., 2010) and the A Land Use Strategy for Scotland (Scottish Government 2011). Nature Directives, WFD and MSFD support many keys aspects of EBM. These include ecological integrity, acknowledgement of multiple scales, multi-disciplinary knowledge, stakeholder participation, transparency, policy coordination, adaptive management. This provides an opportunity for streamlining and coordinating between directives (Rouillard et al., 2018).

1.1.2.3. Environmental Impact Assessments

The United Nations Convention on the Law of the Sea (UNCLOS) defines the responsibilities of individual nations to the use of the marine environment (Davidson, 1997). With the increasing influence of humans on the marine environment, marine licence applications are assessed to understand the likely impacts of proposed activities. Within the North-East Atlantic region, under the Environmental Impact Assessment Directive (85/337/EEC), for projects which are likely to significantly affect the environment, an Environmental Impact Assessment (EIA) must be submitted (UK Parliament, 2017).

The aim of an EIA is to ensure that decisions regarding a proposed development take into account the physical and biological environment and any implications which may arise on the existing environment (Rosenberg et al., 1981). In order to provide this information, the need for science to inform policy is required through a variety of environmental baseline surveys, habitat assessments and monitoring surveys (Borja et al., 2017). In the UK, there is currently no required format but must include the information set out in Schedule 3 to the Marine Works (Environmental Impact Assessment) Regulations 2007 which include an accurate description of the current state of the environment, and the likely evolution of the baseline in the absence of the project such as:

- Human beings, fauna and flora;
- Soil, water, air, climate and the landscape;
- Material assets and the cultural heritage; and

EIAs and Environmental Statements (ES) assess biodiversity which has the potential to be significantly affected by a project, these include:

- Direct and indirect effects;
- Secondary effects;
- Cumulative effects;
- Transboundary;
- Short-term, medium-term and long-term effects;
- Permanent and temporary effects; and
- Positive and negative effects.

This literature review 1) identifies the challenges of monitoring coastal biodiversity in the North-East Atlantic region, 2) explores the main survey methods used to assess this biodiversity and discusses their advantages and disadvantages with the aim of identifying holes and limitations in current assessment methods used to support current legislation and EIAs.

1.2. Literature Review Methods

Literature searches were conducted based on the guidance set out by Pullin & Stewart (2006) using Google Scholar and Web of Science™. Literature searches for this review were undertaken between October 2017 and June 2020. Detailed literature searches included (but not restricted to) the following search terms:

marine biodiversity assessments, coastal environment monitoring, marine sampling techniques and marine monitoring techniques. These terms were used to inform more detailed subsequent searches where required. Grey literature was used with caution throughout this review. Checks on the author / organisation responsible for the article, target audience, date of issue and whether any bias was involved were all taken into consideration.

1.3. Challenges of Assessing Coastal Biodiversity

The coastal marine environment is a dynamic and complex area influenced by processes such as weather, climate, temperature, salinity, seabed morphology, circulation, sedimentary processes, sea level, tidal ranges, wave action and turbidity (Carter & Carter 1988; Huthnance, 2010). Such processes can heavily influence the effectiveness and success of an environmental survey.

The North-East Atlantic region experiences large semidiurnal tidal ranges which in-turn generate locally large near bed currents (Bricheno et al., 2015). Locations such as the Severn Estuary, UK and along the coast of Brittany, France are known for having tidal ranges more than 12m (Green & Smith, 2009) and 5m (Britannica 2021) respecitvely. With this large tidal range comes high current velocities (Stride, 1982), turbidity, debris, sensitive habitats and decreased accessibility to the environment with surveys heavily restricted in the time spent at any one location per day (Natural Resources Wales, 2018). Accessibility to coastal marine areas may also be further restricted by meteorological changes such as prevailing weather conditions influencing wind and swell further restricting survey time in specific locations (Walker et al., 2013). Permission and licences may also need to be granted to access and samples specific sample sites (Boyes & Elliott, 2015), especially with regards to the intertidal zone and the availability of harbours / launching facilities for boat activity and drones (Murfitt et al. 2017).

High current velocities and turbulence associated with these large tidal ranges may also restrict marine surveys through the displacement of equipment placed on the seabed (Uihlein & Magagna 2016). In this instance, towed equipment may be a preferred option to avoid the risk of losing valuable resources (Rooper 2008). Other physical processes associated with these dynamic environments also influence the movement and sorting of sediment. Suspended particulate matter

(SPM) determines turbidity in the water column (The Scottish Government, 2010) which in high levels, ultimately significantly reduces underwater visibility in these environments (Schechner & Karpel, 2005), restricting the use of visual underwater surveys using divers and underwater cameras. In this instance, the preferred option is use of traditional extractive sampling techniques (e.g., beam trawling, grab sampling) to assess these habitats, and in light of their destructive nature, are usually carried out at a distance from the target feature (Griffin et al., 2016). This sampling distance varies and is usually due to restrictions imposed by both industries to protect valuable infrastructure and regulatory bodies to protect valuable habitats and species (MMO 2014). This, in tandem with spatial variations in biodiversity means that communities are notoriously under sampled on and around the vicinity of protected coastal habitats, and despite the protected status of certain coastal habitats, the linkages between these protected habitats and the motile communities they support are poorly understood (Jones et al., 2019).

It is these dynamic coastal environments which provide an opportunity for the development of marine renewable energy which already considered to be under anthropogenic pressure due to their accessibility, biological productivity and other valuable ecosystems and services (Costanza et al., 1997; Gill, 2005). With this in mind, accurately collecting comprehensive environmental baseline data in these environments is essential for appropriately assessing the status of the target environment and magnitude of potential impacts (Gill, 2005).

1.4. Current Methods for Assessing Coastal Biodiversity

1.4.1. Extractive Methods

1.4.1.1. Trawling

Trawling encompasses a range of techniques and can be split into bottom trawling and mid-water trawling (Galbraith et al., 2004). Bottom trawling is the process of towing along, or close to, the seabed whereas mid-water towing is the process of towing the trawl through the water column (FAO, 2016). These types of survey are used to target bottom, demersal and pelagic fish species and are used globally. Beam and otter trawls are two of the most widely used fishing gears in the North Sea (Jennings et al., 1999). Trawl surveys have often been used to monitor fish stocks in long term data sets (Magurran et al., 2010) and can provide information on fish abundance and species composition (Smith, 1996).

Advantages

Trawling surveys have been conducted in the North Atlantic region for numerous marine renewable energy monitoring programs, specifically windfarms (MMO 2014). These can be used as good reference points to monitor change in fish populations over time (Magurran et al., 2010). Trawling can also be considered an effective approach to covering a large 'swept' area (Sparre & Venema, 1998) and estimates can be made on absolute stock sizes (Ligtvoet et al., 1995).

Trawling, like other extractive methods are not limited by specific complex conditions of the water column in coastal environments (Table 1). For instance, turbidity and high velocity currents do not affect quality of the extracted sample; however, changes to trawl speeds due to strong currents may decrease the effectiveness of the trawl (Weinberg, 2003). Methods allow for samples to be extracted and identified either on-board the vessel or in laboratory conditions eliminating the limitation of *in situ* visibility.

Limitations

The ecological impacts of trawling on the seafloor must be considered when undertaking this type of survey, especially within marine reserves and SACs where conservation objectives must be upheld and disturbance to protected habitats is restricted (Hall-Spencer et al., 2002) (Table 1). Subsequent changes to an ecosystem, such as bedforms or reef structures, may occur from extractive surveys and may have an adverse impact on the conservation and monitoring objectives at a specific site or development (Lindholm et al., 2015).

Trawling is an expensive method of assessing biodiversity with surveys usually using large vessels which are unable to operate in very shallow areas (Ligtvoet et al., 1995). Issues with standardisation to ensure that multi-vessel surveys are comparable when analysing trawling data have been a major concern (Galgani et al., 2010). The extent of trawling standardisation varies; it has also been recorded that 'standardised vessels' may still have differing catch capabilities. Changes to any existing gear on a vessel is also expensive and may influence the catchability of target species (Bagley et al., 2015). This variation in catchability can influence unreliable population assessments (Hoffman et al., 2009). Further to this, the selectivity of fishing gear (mesh size and tow speed) for target species limits the assessment of biodiversity through this survey method (Stepputtis et al., 2016); however, methods such as a codend mesh can limit this selectivity if implemented (Ligtvoet et al., 1995).

When towing equipment behind a vessel, there is a risk of it snagging on exposed structures. Renewable energy developments and other subsea infrastructure poses a huge risk to trawlers through snagging and also to the integrity of the existing infrastructure (Det Norske Veritas, 2010). The risks of these methods limit their use within the close vicinity of any installations or protected habitats within the target area. For instance, EIA's and subsequent monitoring for offshore windfarms in the UK regularly use trawl data to assess the state of fish assemblages in the area. However, regardless of statutory requirements, trawl survey data usually refers to that taken from 'within the wider wind farm area' rather than within close proximity to the infrastructure which would inevitably provide more accurate data on the impacts of the development (Griffin et al., 2016; MMO 2014). Furthermore, the gear used on trawl surveys may also be limited by any hard substrate present on the seabed.

1.4.1.2. Seines, Traps and Gillnets

Seine nets haul and herd fish and can either be deployed from the beach or a boat targeting mainly demersal species (FAO, 2018). Gillnetting uses a wall of netting which hangs in the water column and anchored on the seabed. These nets are designed so the fish are either gilled, entangled or enmeshed in the netting (FAO, 2016) and can be used almost everywhere (Ligtvoet et al., 1995). Fyke nets consist of cylindrical net bags mounted on rigid structures with wings at the entrance to guide fish into the net. These nets, like gillnets, are usually fixed to the bottom using anchors (FAO, 2016). This method is commonly used in estuaries and shallow water.

Advantages

These methods are one of the simplest and easiest forms of fishing (Clay, 1981), and can be used as a repetitive sampling tool (Askey et al., 2007). They are considered to be a cost-effective means of fishing for target fish species due to their inexpensive equipment compared to trawling (Askey et al., 2007; Clay, 1981). They are also considered to be less destructive, easy to deploy and are preferable in areas where trawling is unable to operate (Davies et al., 2001; Ligtvoet et al., 1995). Furthermore, limitations such as underwater visibility are not applicable to these methods (Table 1).

Limitations

Seines, traps and gillnet activities also pose a risk to protected species such as marine mammals and turtles through entanglement (Dawson, 1991; Naismith & Knights, 1994).

Trawling provides a more comprehensive assessment of fish biodiversity when compared to methods such as gillnets due to the efficiency of selectivity of gillnets (very size selective) (Li et al., 2017) and the fact they can only catch active fish (Ligtvoet et al., 1995). Gillnets are a passive type of gear and are not useful for estimates of absolute stock size but can provide indices of relative stock size (Ligtvoet et al., 1995) and diversity. Similarly to trawling, limitations may also be placed on the proximity in which seine nets, gillnets and fyke nets may be deployed to marine infrastructure (Griffin et al., 2016). These methods of assessing biodiversity are small scale when compared to the area potentially covered by trawls (Butcher et al., 2005). Deployments of these net types are also difficult in rough sea conditions and high current speeds which often occur in coastal environments as they are often susceptible to displacement (Davies et al., 2001).

1.4.1.3. Benthic grabs

Benthic grab sampling encompasses a range of techniques used to collect a sediment sample from the seabed for analysis of biological infauna, particle size and contaminants (EPA, 2002). There are several types of benthic grab, these include Hamon, Day, Shipek, van Veen and Petersen samplers (Hails, 1982). The Hamon grab has been identified as the most effective grab type for mixed sediments (Boyd et al., 2006) whereas the day grab has been identified as the most effective type for soft homogenous sediments (Rogers et al., 2008) and typically sample a $0.1m^2$ area. Grabs consist of two 'bucket' sections. When in contact with the seabed, these buckets close into the seabed and extract a sample before being hauled back up to the vessel. Although the concept of these techniques has remained the same over the years, developments have been made into the gear efficiency and positioning equipment.

Advantages

This method is easily conducted from a range of boat sizes and provides detailed quantitative data on sedentary infaunal and epifaunal species which are open to use in various statistical analysis and interpretation (Eleftheriou, 2013). The interpretation of results from grab samples does not usually require advanced technology or software; however, personnel must be adequately trained in taxonomy and/or faunal identification. This sampling method is often used as a ground truthing tool alongside non-extractive methods such as underwater cameras and acoustic surveys (Davies et al., 2001).

When using the same sampling gear and accurate positioning information, this method can be easily replicated over time providing a good indication of benthic environmental changes (Eleftheriou & McIntyre, 2005). This method also provides data for certain habitat characterisations where considerable comparable information is available and also for quantitatively determining habitats using various multivariate data outputs (Eleftheriou, 2013). Due to the extractive nature of this method, it is not restricted by visibility in the water (Table 1) column and samples can be retained for future analysis.

Limitations

Benthic grabbing and subsequent faunal analysis are a time consuming and costly process (Kingston & Riddle, 1989). Similar to trawling, restrictions may be placed on these techniques in areas of significant subsea infrastructure such as cables and pipelines or areas paramount to supporting any existing infrastructure and predicted sensitive benthic habitats (Noble-James et al., 2017).

Large variations in benthic community structure and habitats can occur over a small spatial scale when implementing this method. In this instance, a more intensive sampling regime is needed to account for this variation (Underwood & Chapman, 2013) with benthic grabs also not useful in sampling rare fauna (McIntyre, 1956). Benthic grab surveys are also limited by substrate type with different sediment types requiring different grab equipment each with variations in efficiency (Underwood & Chapman, 2013; Word, 1975). This type of survey method targets fine grained, cohesive sediment such as silt and clay, and non-cohesive sands including shell, and gravel (Hails, 1982). Benthic grabs are not effective on sediments that may prevent grab closure such as rocky outcrops

(Davies et al., 2001; Jørgensen et al., 2011). This method therefore becomes difficult to implement in coastal monitoring surveys in high energy tidal stream and wave sites where rocky ledges, boulders and soft-sediment patches are common features (Sheehan et al., 2010).

The deployment of grab equipment relies on existing information either from previous surveys or from pre-camera surveys or other ground surveys to target areas of interest and reduce damage to protected habitats (Davies et al., 2001). Larger and more mobile fauna tend to be underestimated via this method as they are more likely to have the ability to avoid the equipment when disturbed by burrowing deeper into the sediment or have a body size too big for the grab (Costello et al., 2017; Kendall & Widdicombe, 1999). Differences may also occur in the data formats of epifauna and infauna (Tagliapietra & Sigovini, 2010).

Table 1: Overview of the advantages and limitations of extractive sampling for assessing coastal biodiversity in the North-East Atlantic region

Assessment Technique	Data Type	Advantages	Disadvantages	Applicable in Low Visibility Environments	Applicable in High Velocity Environments
Bottom trawling Mid-water trawling	• Fish abundance. • Total fish counts. • Biomass. • Species composition.	• Numerous long-term data sets already exist. • Can cover a large area in a short period of time. • Not limited by conditions of the water column. • Can estimate absolute stock size.	• Issues with standardisation. • Gear is expensive. • Risk of snagging on marine infrastructure. • Selectivity of gear-but can be limited. • Ecological impacts must be considered-Destructive. • Limited by depth.	**Yes.** Trawling methods are not limited by low visibility coastal environments.	**Yes.** Trawling methods are not limited by high velocities in coastal environments unless boat trawl speed is compromised.
Gillnetting Fykes Seine	• Fish abundance. • Total fish counts. • Biomass. • Species composition.	• Simple forms of fishing. • Cost-effective. • Easy to use. • Less destructive than trawling to the seabed. • Not restricted by sediment type.	• Highly selective • Risk of snagging on marine infrastructure. • Entanglement of non-target species. • Cover a small area. • Cannot estimate absolute stock size. • Passive. • Risk of displacement. • Restricted by water depth.	**Yes.** Seines, traps and gillnets are not limited by low visibility in coastal environments.	**No.** Seines, Traps and Gillnets are not applicable in very high velocity environments where displacement may occur.
Hamon Day grab Shipek, van Veen Petersen Cores Dredges Sleds	• Biological infauna abundance • Biomass • Species composition • Particle size analysis • Contamination	• Easily conducted from different vessel sizes. • Provides detailed quantitative data. • Can be replicated easily with accurate positioning equipment.	• Large variations in benthic community structure and habitats can occur over a small spatial scale. • Ecological impacts must be considered-Destructive. • Limited by substrate type.	**Yes.** Benthic grabbing techniques are not limited by low visibility coastal environments.	**Yes.** Benthic grabbing techniques are not limited by high velocity coastal environments.

Assessment Technique	Data Type	Advantages	Disadvantages	Applicable in Low Visibility Environments	Applicable in High Velocity Environments
		• Suitable for multivariate analysis. • Not restricted by water column characteristics.	• Relies on existing information before deployment. • Megafauna can be too big for the grab. • Time consuming and costly. • Restricted sampling around infrastructure.		

1.4.2. Non-Extractive Methods

1.4.2.1. Underwater Cameras

The use of underwater camera methods for assessing marine biodiversity has increased in recent times with technological progress in battery life, underwater housings, and storage now making these methods available for a number of users (Bicknell et al., 2016; Mallet & Pelletier, 2014). These methods are considered an alternative to destructive methods, especially in sensitive and complex habitats, and around marine infrastructure (Griffin et al., 2016; Unsworth et al., 2014). Underwater video encompasses a range of different techniques and can be applied to different species and habitat assessments. There are several types of camera which fall under this method, these include remote underwater video, baited remote underwater video (BRUV) (Cappo, et al., 2006), benthic drop-down video (Bethoney & Stokesbury, 2018; Jenkins et al., 2018), towed video (Sheehan et al., 2016), remotely operated vehicles (ROVs), autonomous underwater vehicle (AUV) and diver-operated video (Mallet & Pelletier, 2014; Sward et al., 2019). More recently, high-resolution Serial Peripheral Interface (SPI) cameras using Raspberry Pi microcomputers have been identified as customisable, cheap (<200 euro) marine camera systems (Purser et al., 2020).

Advantages

One of the advantages of underwater cameras lies in the variety of different types of camera which exist and applicability for reaching different survey objectives in various marine environments including benthic habitat ground-truthing, fish species behaviour and habitat use, inter- and intra-specific interactions and population level monitoring (Bicknell et al., 2016) (Table 2). Cameras may be baited or un-baited depending on the survey and also remotely deployed (Murphy & Jenkins, 2010). In the North-East Atlantic region, camera methods including benthic drop-down and towed video have been used for benthic habitat assessments whilst BRUVs have previously been used for fish assessments around marine renewable energy developments such as windfarms (Hitchin et al., 2015; Sheehan et al., 2016; Griffin et al., 2016). To counter environments subject to low visiblity, 'freshwater housings' have been implemented in benthic habitat assessments in both the oil & gas and marine renewable enrgy industries (Hitchin et al., 2015).

Underwater cameras are non-extractive and can provide a high-definition view of the target area and access those habitats or environments that may be inaccessible by conventional methods (Unsworth et al., 2014), such as subsea infrastructure (Bicknell et al., 2019). Unlike trawling or benthic grab sediment sampling, underwater cameras are not restricted by hard bottom substrates and are often critical for monitoring this type of environment (Pohle & Thomas, 1997). Comparisons have previously been made between camera surveys and underwater visual census using divers and show that underwater video is more cost-effective in terms of time spent out in the field or in a laboratory (Francour et al., 1999). Exclusion of divers when surveying has removed time and depth limitations as well as reductions in survey cost and diver bias (Stobart et al., 2015). It has also reduced the health and safety risk to personnel (Griffin et al., 2016; Jones et al., 2019).

The difficulty of accurately assessing faunal lengths has been overcome by using stereo camera systems which use two cameras pointed at the same area to create a 3D image for analysis (Boom et al., 2014; Murphy & Jenkins, 2010; Unsworth et al., 2014) allowing various measurements to be taken. BRUVs are an example of this and use either a single or two cameras (stereo) to record an area and use bait to attract fish (Dorman et al., 2012; Ghazilou et al., 2016; Harvey et al., 2012; Mallet & Pelletier, 2014). The target species and the range of action are determined by the type of bait used (Ebner & Morgan, 2013; Hannah & Blume, 2014; Harvey et al., 2007). The use of baited cameras provides a means of collecting ecological data on motile fauna (Cappo et al., 2004; Griffin et al., 2016) and decreases the occurrence of zero counts and increase the repeatability between surveys (Murphy & Jenkins, 2010). Cameras can be set up either horizontally or vertically to target fish assemblages and benthic habitats respectively (Mallet & Pelletier, 2014), and can be used in both coastal waters and deeper offshore waters and are potentially a useful tool in low visibility environments when the position of the bait is relatively close to the camera (Cappo et al., 2006; Jones et al., 2019; Lowry et al., 2012; Unsworth et al., 2014). The static nature of these systems is desirable for monitoring biodiversity around marine developments and infrastructure (Griffin et al., 2016). BRUV systems have been identified as providing better statistical power than un-baited systems in the detection of spatial and temporal changes in the relative abundance and structure of fish assemblages (Stobart et al., 2015).

To cover larger spatial areas, towed cameras may be used (Bicknell et al., 2016; Sheehan et al., 2010). These cameras may be used either at the seabed or in mid-water (Mallet & Pelletier, 2014) towed behind a vessel to record habitat data during transects of an area for future spatial analysis (Stoner et al., 2007).

Limitations

One of the main challenges identified in the literature with using underwater cameras is gaining accurate data in low visibility environments (<4m visibility) (Jones et al., 2019) (Table 2). Coastal areas are often described as highly dynamic environments especially around estuaries where areas of high turbidity, organic matter, plankton and sediment loads are present (Uncles et al., 2002). Research has been conducted into the improvement of autonomous underwater vehicles navigation and surveillance (Cho & Kim, 2017) and using highly sensitive cameras in aquaculture ponds (Hung et al., 2016); however, little research has been conducted into improving visibility of underwater cameras for assessing and monitoring coastal biodiversity. As previously discussed, turbulence and large tidal ranges are common in coastal areas especially in the North-East Atlantic region. Static monitoring equipment placed in these environments have the potential to be moved and visibility in the water column further reduced (Fraser et al., 2016). With non-extractive sampling such as underwater cameras a vital and preferred method in assessing sensitive coastal habitats (Davies et al., 2001), limitations placed on the cameras through the extreme environments which they are deployed currently significantly reduces their effectiveness in certain locations.

The duration of image analysis is also still an issue when it comes to video data with different video techniques exhibiting differences in information provided and requiring differing effort to analyse. The analysis of a camera observation can take anything from a few minutes to an hour depending on the survey type, objectives and the deployment times (Mallet & Pelletier, 2014). Further issues may include human error when identifying species as it is far more difficult to accurately identify marine species, especially benthos, using images alone (Durden et al., 2016; Mallet & Pelletier, 2014). Fauna such as hydroids, bryozoans and fine algae are notoriously difficult to identify through underwater cameras. In this instance, further ground-truthing is usually required (Davies et al., 2001).

With regards to baited video, bias towards predators and/or scavenger species has previously been recorded, (Stobart et al., 2015, 2007) with different bait types also influencing the species of fish attracted to the area (Dorman et al., 2012; Ghazilou et al., 2016; Hannah & Blume, 2014; Harvey et al., 2007; Whitmarsh, Fairweather, & Huveneers, 2017).

Research suggests that for underwater camera surveys, a combination of methods, such as BRUVS and different fish nets or drop-down benthic cameras and benthic sediment grabs, may be a more accurate way of assessing biodiversity as it potentially counteracts each of their inherent biases (Davies et al., 2001; Murphy & Jenkins, 2010; Watson et al., 2005). Such integrated approaches have been identified in the WFD, with respect to fish communities (Coates et al., 2007).

1.4.2.2. Acoustic Surveys

Benthic acoustic surveys include various types of equipment including multibeam echosounders (MBES), side scan sonar (SSS) and sub bottom acoustic profilers (SBPs) (Table 2). These methods provide detailed information on water depth, seabed morphology, objects and features. These methods are predominantly used for habitat assessments and mapping (Bates & Moore, 2002). Marine acoustic surveys have also been applied to the monitoring of marine mammals (Mellinger et al., 2007; Verfuss et al., 2018) and fish assemblages using methods such as passive acoustic monitoring (PAM) (Merchant et al., 2014; Verfuss et al., 2018), fixed autonomous underwater sound recorders (ARs) (Sousa-Lima et al., 2013), fish acoustics (Jolly & Hampton, 1990; Maravelias et al., 1996), acoustic telemetry (tagging) (Abecasis et al., 2018) and acoustic cameras (Martignac et al., 2015).

Advantages

Acoustic survey techniques can be applied to the monitoring of many different aspects of biodiversity. They are considered superior to visual surveys with their greater detection ranges and continuous long-term monitoring in isolated areas, independent of weather or light conditions influencing visibility in more extreme environments (Wiggins & Hildebrand, 2007) (Table 2). Through the use of acoustic technology, accuracy of mapping and understanding of spatial patterns can be applied to the seabed through the use of advanced aerial technology and remote sensing techniques (Brown et al., 2011). There are no depth or substrate limitations when using acoustic survey equipment allowing for large areas of substrata to be mapped in a short space of time (Magorrian et al., 2009). Additionally, the use of acoustic measures such as the acoustic complexity index can be used to correlate with visual biodiversity estimates to offer an alternative to assess ecosystem health (Davies et al., 2020).

Unlike underwater camera surveys, acoustic methods are not restricted by poor water visibility (Davies et al., 2020) whether it's for marine mammal and fish detection or benthic habitat mapping (Dudzinski et al., 2011). However, research has suggested that acoustic detections of organisms are susceptible to acoustic backscatter in areas of extreme turbulence or turbid sediment-laden waters (Melvin & Cochrane, 2014). The use of novel acoustic technologies (hydroacoustics) show proven advantages in their usage around renewable energy developments (Fraser et al., 2017; Fraser et al., 2016).

Limitations

Similarly to extractive survey methods, limitations are placed on the proximity of towed equipment to marine infrastructure for risk of snagging and also require a boat to operate the equipment (Evans & Thomas, 2011) with acoustic equipment also usually expensive (Table 2). Environmental factors potentially influencing the detection range of methods such as acoustic telemetry include thermocline gradient, depth, and wind speed (Huveneers et al., 2016). Hydroacoustic devices used in turbulent areas heavily rely on established calibration, processing and analysis techniques (Fraser et al., 2017). Research suggests that acoustic surveys alone are not an adequate method of assessing biodiversity (Horne, 2000); ground-truthing through the use of grab samples, camera footage or ground discrimination surveys are usually required to provide information on biological community composition and specific species abundance. Furthermore, the identification of fauna to species level with confidence is currently almost impossible (Brown et al., 2011; Horne, 2000; Langkau et al., 2012; Mackinson et al., 2002; McClatchie et al., 2000; Vihervaara et al., 2017).

1.4.2.3. Aerial Surveys

Aerial surveys encompass a wide range of techniques including drone surveys, aircraft surveys, radar and earth observation data. Surveys using manned lightweight aircraft such as planes and helicopters have commonly been used to gain population and distribution estimates of visible marine mammals such as whales, dolphins and seals (Colefax et al., 2017; Keller et al., 2006). In recent times, technological advances in drone technology have allowed for their establishment in marine survey methodologies for multiple purposes including conservation, wildlife monitoring and habitat mapping (Christie et al., 2016; Duffy et al., 2017; Koh & Wich, 2012).

Earth observation data have also been identified as useful sources of information for habitat mapping/monitoring and for other environmental characteristics which have been identified as good proxies for biodiversity. Groups such as GEO BON (the Group on Earth Observations Biodiversity Observation Network) promote the use of these techniques for monitoring biodiversity (Kuenzer et al., 2014).

Advantages

Aerial surveys have the ability to cover large areas quickly, providing a detailed account of spatial environmental changes in a specific area overtime and provide a new vantage point when estimating marine mammal populations (Fretwell et al., 2014; Jean et al., 2010) (Table 2). Due to the functional and logistical limitations of manned aerial surveys (Colefax et al., 2017), unmanned aerial vehicles (UAVs) have become an increasingly popular alternative to assessing wildlife abundance, behaviours and habitat extents (Christie et al., 2016).

UAVs are an efficient, cost-effective approach to monitoring biodiversity (Colefax et al., 2017), allowing researchers to reach remote areas and observe animals from an advantageous perspective (Hodgson et al., 2017). The data acquired from UAVs are more accurate and human-risk free (Turner et al., 2016). The change from human observations to images allows for the collection of new types of information (Hodgson et al., 2017) using accurate GPS locations (Hodgson et al., 2013). The risk of missed sightings/misidentification of animals is also minimised using UAVs when compared to manned surveys (Aniceto et al., 2018; Fiori et al., 2017). Developments in lithium batteries, component miniaturisation and high-resolution image capture have made UAVs a versatile assessment method (Colefax et al., 2017). Aerial images allow for ease when monitoring large scale coastal ecosystems capturing centimetre resolution imagery and topographic data in situations where conventional approaches are not possible. These include intertidal areas where small windows of opportunity exist to survey exposed habitats (Murfitt et al., 2017).

In addition, earth observation data by satellite provides long-term, spatially continuous, regular, and repeatable observations over large areas (Cord et al., 2017). The development in earth observation data has enabled the establishment of comprehensive biodiversity monitoring schemes with large amounts of data freely available. *In situ* data are increasingly being stored on various geographic information system (GIS) platforms and are constantly updated allowing for gaps in biodiversity knowledge to be filled (Vihervaara et al., 2017) providing useful insights into the functioning of ecosystems and on the drivers of environmental change (Cord et al., 2017).

Limitations

There are a number of considerations that must be addressed before the deployment of manned or unmanned vehicles; these include pre-flight planning, establishing safe locations for take-off and landing, familiarising with international, regional and local legislation and site-specific planning (Duffy et al., 2017). Weather and the local environment are considered one of the main restrictions on the use of UAVs such as drones with complex winds restricting airtime and rough coastal terrain restricting landing. Similar restrictions may be placed on manned aircraft in certain weather conditions (Hodgson et al., 2017) (Table 2).

Aerial surveys are susceptible to external factor biases that influence availability errors caused by turbidity and the depth of the target animal below the water level reducing sightability (Fiori et al., 2017; Pollock & Kendall, 1987). Perception bias may also occur in manned surveys where an animal is theoretically available for detection but is not sighted due to factors related to the sampling methodology which may include environmental conditions or flight characteristics. Bias in relation to methodology may also occur, as accuracy of observations may decrease with transect width, cruising speed and height (Caughley, 1974). The manual review of images is also time consuming; the efficiency of UAV surveys relies on the development of image analysis algorithms to automate the detection of animals within images (Hodgson et al., 2017).

Earth observation data analysis has mainly been used to monitor terrestrial habitats (Cord et al., 2017) with coastal and offshore environments limited by submergence. Furthermore, the data acquired through earth observation data is of a lower spatial resolution and needs to be calibrated via *in situ* data. Variations in atmospheric condition may also influence the quality of images.

1.4.2.4. Intertidal Benthic Surveys

The intertidal zone covers the area between low tidal and high tide and can be in the form of sandy beaches, rocky shores, mudflats and marshes (Ray, 1991). Assessing biodiversity within the intertidal zone falls within both extractive and non-extractive survey methods. Epifauna assessments may be conducted by visual surveys, with infauna assessments conducted by taking small sediment samples for further identification in a laboratory (Costello et al., 2017). Intertidal surveys usually fall between techniques outlined in Phase I terrestrial mapping (Davies et al., 2001) and those within Marine Nature Conservation Review (MNCR) Phase II marine survey methodologies (Hiscock, 1998) and are conducted within a window when the shore is exposed.

Advantages

Accessing the intertidal zone is relatively easy at low tide with no expensive or specialist equipment usually required (Table 2). Rapid quantitative and qualitative assessments on abundances using an appropriate scale can be taken, especially on intertidal areas such as rocky shores where environmental gradients are sharper (Smith, 2005). Intertidal surveys can be considered a quick, inexpensive and straightforward method for monitoring certain attributes of a site, with trained scientists not always required to collect data (Godet et al., 2009). Data and visual observations collected *in situ* via field notes and photographs may be re-examined at a later date and/or confirmed by further analysis of any samples collected with no restrictions of visibility or high velocity currents.

Limitations

Whilst this survey method is not limited by substrate type, it is limited by accessibility to the shore with the dynamic morphology of intertidal areas logistically challenging (Bird, 2016) (Table 2). Areas such as the Severn Estuary, UK where large and quick turning tides can be a limiting factor, time spent at a specific location may be greatly reduced (Schlacher et al., 2008). Furthermore, *in* *situ* monitoring may not always provide adequate data to comply with monitoring objectives of a site or development (Davies et al., 2001).

1.4.2.5. Other Visual Surveys

Similar limitations to aerial surveys may be placed on visual surveys undertaken on vessels or land. Accessibility must also be taken into account when using these methods (Table 2).

Divers are also considered a valuable tool for visual observations underwater. Similar to underwater camera methods, underwater visibility and current velocities may also heavily restrict the ability of a diver to accurately observe the environment. As previously discussed, the use of divers when surveying has time and depth limitations as well as diver bias (Stobart et al., 2015). There is also a health and safety risk to personnel when using this technique in hazardous environments (Griffin et al., 2016).

Table 2: Overview of the advantages and limitations of non-extractive sampling for assessing coastal biodiversity in North-East Atlantic region

Assessment Technique	Data Type	Advantages	Disadvantages	Applicable in Low Visibility Environments	Applicable in High Velocity Environments
BRUV RUV Stereo Towed Diver Drop-down	• Fish abundance. • Fish lengths and biomass. • Species composition. • Habitat extents. • Species behaviour. • Seabed features. • Benthic epifauna.	• Many types. • Flexible applications to different environments. • Non-extractive/ destructive. • Not restricted by substrate. • Can access areas inaccessible by conventional methods. • Cost-effective in terms of time out in the field. • Can be Static or towed. • Can be deployed by non-scientific staff.	• Low visibility in turbid and turbulent environments. • Decreased stability in turbulent environments. • Duration of image analysis. • Baited camera bias towards predators /scavengers. • Human error in fish and benthic identification.	**No.** Underwater cameras are limited in low visibility coastal environments due to decreased image quality.	**No.** Underwater cameras are limited in high velocity coastal environments due to reduced visibility and displacement of equipment.
Multibeam Echosounders Side scan sonar PAM Acoustic telemetry Acoustic cameras AR	• Presence of mammals / fish • Water depth. • Seabed morphology • Objects and features.	• Considered superior to visual surveys. • Large detection ranges. • Continuous long-term monitoring in isolated areas independent of light or weather conditions. • Monitor anthropogenic impacts. • No depth or substrate limitations. • Can cover large areas. • No visibility limitations.	• Proximity of towed equipment to infrastructure. • Require a boat to operate equipment. • Equipment is usually expensive. • Susceptible to backscatter in turbulence. • These methods alone are not considered an accurate method of assessing biodiversity. • Difficult to confidently identify fauna to species level.	**Yes.** Acoustic surveys are not limited by low visibility coastal environments.	**No.** Acoustic surveys are limited in high velocity coastal environments where backscatter and equipment displacement may occur.
Manned aircraft	• Aerial photographs.	• Cover large areas quickly.	• Extensive pre-flight planning.	**No.** Aerial surveys are limited by low	**Yes.** Aerial surveys are not

Assessment Technique	Data Type	Advantages	Disadvantages	Applicable in Low Visibility Environments	Applicable in High Velocity Environments
UAV Earth Observation Data	• Intertidal habitat extents. • Presence and behaviour of marine mammals. • Satellite data monitoring a specific point over time.	• Can provide information on a specific area over time. • UAVs cost-effective, efficient, human-risk free. • Video data collected for future reference. • Use in remote locations and inaccessible habitats. • Free, public data.	• Manned aircraft restricted to vicinity of airfield. • UAVs restricted by weather and local environment. • Legislation. • Bias through external factors, methods and human error. • Time consuming analysis. • Atmospheric conditions.	visibility coastal environments associated with poor weather conditions and water turbidity.	limited by high velocity coastal environments. They are however limited by weather conditions.
Phase I Phase II Walkover survey	• Biotope extents • Biotope proportions • Epifauna and infauna abundances	• Cost effective. • No specialist equipment needed. • Quick • Data can be collected *in situ.*	• Can be limited by accessibility to the shore. • In situ monitoring not always an option. • Inaccurate on its own for biotope mapping.	**Yes.** Intertidal surveys are not limited by low visibility coastal environments.	**Yes.** Intertidal surveys are not limited by high velocity coastal environments.
Boat surveys Land surveys Diver surveys	• Species abundance • Species behaviour • Habitat extents	• Cost-effective • Quick means of providing an overview of the environment.	• Human bias • Accessibility to sites. • Species identification problems. • Reduced visibility in poor weather conditions. • Diver health and safety risks.	**No.** Visual surveys are limited by low visibility coastal environments due to varying meterogical characteristics.	**Yes/No.** Visual surveys are not limited by high velocity coastal environments when conducted on land or from a vessel. However, time spent on the shore may be limited in extreme tidal ranges and underwater currents may restrict the use of divers.

1.5. Discussion

As discussed, the coastal environment is a highly dynamic, diverse area, susceptible to anthropogenic pressures due to its accessibility for the human population and its highly valued ecosystem services. The development of various marine industries in this environment has led for the necessity of comprehensive baseline and monitoring surveys for the protection of marine coastal biodiversity. Marine monitoring methods for biodiversity maybe implemented in research (academia), industry (consultancies, surveyors), government organisations (Natural England, Natural Resources Wales) and by non-government organisations (JNCC).

Current methods of assessing coastal biodiversity each have their advantages and limitations depending on the environment in which they are used and the objectives of the survey. The main challenges identified in this review of accurately assessing the coastal environment in the North-East Atlantic region are as follows:

- Visibility;
- High current velocities;
- Destructive extractive sampling nature;
- Human, methodological and external sampling bias;
- Analysis duration;
- Cost-effectiveness;
- Selectivity of gear;
- Weather; and
- Accessibility.

This review has identified that within the UK especially, large tidal ranges influencing high velocity currents and increased water turbidity and site accessibility are considered to be a major limiting factor in the surveying of coastal biodiversity. This has therefore meant that preferred methods of assessment are extractive in nature, and not affected by such limitations.

Evidence suggests that a combination of methods may be recommended in order to provide an accurate overview of biodiversity such as those identified under the WFD (Coates et al., 2007) (Table 3), where a suite of techniques is preferred in order to obtain an accurate picture of assemblages being assessed (Gabriel et al., 2005). For instance, fish assessments in the Thames Estuary have utilised this approach through the use of seine netting, beam trawling, and otter trawling to reduce gear selectivity bias. The use of this multi-method approach has been recognised as an example of 'European Best Practice' estuarine fishery monitoring programme (Coates et al., 2007). Methods and techniques must be specific to survey objectives and target areas. All methods have biases which need to be recognised in data interpretation. Assumptions cannot be made that a conventional method is representative of all biodiversity within a target area (Costello et al., 2017).

The need for accurate EIAs, ES's and monitoring data for marine developments is essential for the preservation of marine biodiversity in coastal areas. These processes have however been restricted by ambiguities in legislation and a lack of clear implementation guidance for the assessment of significant effects (Maclean et al., 2014). For offshore energy industries and installations, the quality of environmental baseline data is satisfactory with descriptions of the environmental baselines variable with most aspects relying on existing sources of knowledge without the collection of new data (Barker & Jones, 2013). Differences in the quality of baseline data can be highlighted in the differences in benthic and fish assemblage baseline sections. For most offshore ES and EIAs, benthic surveys are often carried out through benthic grab and camera sampling; however, the same new surveys to collect new baseline data on fish assemblages are not usually applied unless existing data sources are extremely lacking (MMO 2014). Data and maps used for fish assessments may rely on out-of-date data such as Ellis et al., (2012) and Coull et al. (1998) which provide information on the spawning and nursery areas for commercially important species rather than fish assemblages as a whole which are specific to an area.

Where development specific environmental baseline surveys have been carried out, these have been traditionally focused around benthic grabs and trawling (MMO 2014). Whilst these methods provide a good insight into benthic fauna and fish biodiversity, they are limited in their use within close proximity to existing infrastructure and damage caused to the environment, especially habitats and species which are protected (Eleftheriou, 2013).

Testing novel methods and improving existing methods for assessing coastal biodiversity is an important step in addressing the survey limitations highlighted in this review. With Marine Protected Areas (MPAs) becoming an increasingly important tool in marine management, acquiring accurate baseline data has become paramount in the decision-making process and clearly defining conservation objectives. Reducing scientific uncertainty about coastal biodiversity will undoubtedly lead to more well designed and managed MPAs and benefit coastal and marine ecosystems, and their human components, if they are designed to be flexible and adaptive (Agardy et al., 2003).

Table 3: Recommended sampling techniques for assessing coastal biodiversity in the North-East Atlantic region

	Recommended Sampling Methods	**Literature**
Fish	- BRUVs - Trawling - Acoustic cameras - Acoustic telemetry - Echosounders - ROV	Abecasis et al., (2018) Bicknell et al., (2016) Griffin et al., (2016) Jones et al., (2019) Ligtvoet et al., (1995) Martignac et al., (2015) Sparre & Venema, (1998) Unsworth et al., (2014)
Benthos	- Drop-down video - Towed video - Hydroacoustics (SSS, MBES) - Benthic grab sampling - Aerial and visual surveys (Intertidal) - ROV	Bicknell et al., (2016) Brown et al., (2011) Eleftheriou, (2013) Davies et al., (2020) Murfitt et al., (2017) Sheehan et al., (2010) Sheehan et al., (2016)
Marine Mammals	- Aerial and visual surveys - Acoustic telemetry - Passive acoustics (PAM) - Acoustic cameras - Autonomous underwater sound recorders	Aniceto et al., (2018) Abecasis et al., (2018) Colefax et al., (2017) Hodgson et al., (2017) Merchant et al., (2014) Verfuss et al., (2018) Wiggins & Hildebrand (2007)

1.6. Scope for Further Research

With extractive survey methods damaging the baseline environment and limited by their proximity to seabed infrastructure and sensitive habitats, the use of static, non-extractive equipment such as underwater cameras, are considered a viable alternative. One of the main challenges to underwater video in North-East Atlantic coastal waters are high current velocities influenced by large tidal ranges in turn increasing water turbidity and reducing underwater visibility (Bicknell et al., 2016; Hung et al., 2016; Mallet & Pelletier, 2014; Taylor et al., 2013). Such limitations have not been adequately addressed in past research in this region.

Novel methods to improve underwater image quality may lie in the use of freshwater housings and / or the use of acoustic technology. As it stands, very little published literature exists in relation to freshwater housings. A freshwater housing consists of a lens which is filled with clean freshwater water in front of a camera in order to reduce the path length that light must travel through turbid water (Hitchin et al., 2015). This method has been used regularly in both deep sea and coastal surveys by the marine surveying industry for drop-down camera deployments; however, little research has been conducted using these methods to quantify their ability to assess benthic and fish assemblages in both control and field environments. The integration of bait with this method to attract organisms to an area, potentially makes for a novel sampling method in low visibility environments. Other methods such as imaging sonar may be considered the next step in providing clear footage of low visibility marine environments and have previously been applied to turbid freshwater environments for fish monitoring during migrations. However, in its current form, the ability to identify fauna to species level using imaging sonar is difficult. Refining these methods and / or combining this with existing methods such as digital BRUV footage may be a viable option in gaining accurate information of the coastal environment and to inform conservation and industrial development decisions. Furthermore, the standardisation of such survey methods is essential for comparing survey results both spatially and temporally when interpreting data.

Chapter 2

Consistency is Critical for the Effective Use of Baited Remote Video

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Goldcliff, 2017

Abstract

Baited Remote Underwater Video (BRUV) are popular marine monitoring techniques used for the assessment of motile fauna. Currently, the majority of published studies evaluating these BRUV methods stem from environments in the Southern Hemisphere. This has led to stricter and more defined guidelines for the use of these techniques in these areas in comparison to Northern temperate environments where little or no specific guidance exists. This study explores metadata taken from BRUV deployments around the UK to understand the influence of methodological and environmental factors on the information gathered during BRUV deployments. In total, 39 BRUV surveys accumulating 457 BRUV deployments, from 26 locations around the UK in depths between 4-30m were used in this analysis.

This study identified 88 different taxa from 43 families across all deployments. Whilst taxonomic groups such as Labridae, Gadidae and Gobiidae were represented by a high number of species, species diversity for the Clupeidae, Scombridae, Sparidae, Soleidae, Gasterosteidae and Rajidae families were low with many families absent altogether. Bait type was consistently identified as one of the most influential factors over species richness, relative abundance and faunal assemblage composition. Image quality and deployment duration were also identified as significant influential factors over relative abundance. As expected, habitat observed was identified as an influential factor over faunal assemblage composition in addition to its significant interaction with image quality, time of deployment, bait type and tide type (spring / neap).

Our findings suggest that methodological and environmental factors should be taken into account when designing and implementing monitoring surveys using BRUV techniques. Fluctuations and variations in data may be attributed to methodological inconsistencies and/or environment factors as well as over time and therefore must be considered when interpreting the data.

Keywords: Baited remote underwater video; Coastal biodiversity; Environmental monitoring; Metadata; Temperate habitats; Underwater cameras.

2.1. Introduction

Baited Remote Underwater Video (BRUV) are popular marine monitoring techniques used for the assessment of motile fauna (Whitmarsh et al., 2017). Although these techniques have predominately focussed on fish assemblages (Lowry et al., 2011), they have also been applied to large marine predators including sharks and pinnipeds as well as invertebrates such as Cephalopoda and crustacea (Whitmarsh et al., 2017). They have also been used for length measurements of assemblages, particularly fish (Whitmarsh et al., 2017). Such systems may consist of either one (mono) or two (stereo) cameras which film the area surrounding a bait used to attract motile fauna into the field of view of a camera (Cappo et al., 2006; Mallet & Pelletier, 2014; Wraith et al., 2013). Since the mid- nineties (Ellis & DeMartini, 1995), these methods have been used to assess abundances, diversity and behaviour of motile assemblages (Cappo et al., 2006; Martinez et al., 2011; Priede et al., 1994) and have also been effective in aiding the assessment of metabolic rates (Cappo et al., 2006). They are a cost-effective and safer alternative to other methods such as underwater visual census, remotely operated vehicles or SCUBA divers where issues such as depth, submergence times and potentially dangerous fauna are considered limiting factors to data collection (Esteban et al., 2018; Jones, et al., 2019). They are also considered a much less destructive alternative to extractive survey techniques such as trawling (Davies et al., 2001).

Currently, most published studies evaluating BRUV methods stem from marine environments in the Southern Hemisphere with Australia and New Zealand leading the way in this research (Langlois et al. 2020). Most assessments utilising BRUV methods are undertaken on rocky reef, coral reef and deep-water habitats (Whitmarsh et al., 2017) and in comparison, relatively rarely on coastal soft-sediment habitats, although the following references Borland et al., 2017; Schultz et al., 2019; and Vargas-Fonseca et al., 2016, provide more recent examples of such research. Studies have involved the use of various equipment set ups, bait types and sampling designs in varying environmental conditions.

Defined guidelines for BRUV methodologies in the North Atlantic Region and elsewhere in Europe are currently lacking in comparison to countries such as New Zealand and Australia where vertical and horizontal BRUV guidelines have been published (Haggitt et al., 2014; Langlois et al., et al., 2020). Recent reviews of the protocols associated with BRUV methodologies are now starting to pave the way globally for Findable, Accessible, Interoperable, and Reproducible (FAIR) workflows (Wilkinson et al., 2016) when utilising these techniques (Langlois et al., 2020). Factors such as deployment duration (Unsworth et al., 2014), bait type (Dorman et al., 2012; Hannah & Blume, 2014; Harvey et al., 2007; Jones et al., 2020), time of day (Bassett & Montgomery, 2011; Birt et al., 2012), tidal currents (Taylor et al., 2013) and habitat type (Langlois et al., 2020; McLean et al., 2016) may influence information gathered for species richness, abundances and faunal assemblage composition (Grimmel et al., 2020). Within New Zealand for example, guidelines for BRUV deployments include important aspects such as descriptions of sources of bias (with suggestions of minimising and avoidance), sampling design, equipment, field deployment, data management, abundance and size estimates and data analysis (Haggitt et al., 2014).

This standardisation of methods is vital in monitoring biodiversity in a target area to ensure that comparisons between years or to other study locations is comprehensible and replicable in the future (Costello et al., 2017; Taylor et al., 2013). However, monitoring method guidelines are defined by the geographical area and policy areas which they serve (Turrell, 2018). Due to different biological (e.g., species and habitats) and environmental (e.g., hydrodynamics, sediments, topography) parameters present at different locations globally, a 'one-method suits all' approach may not be possible. Establishing and testing guidelines based on existing knowledge is therefore important for monitoring marine assemblages such as fish in different regions. Implementing such guidelines may allow for the effective management of protected areas to assess their effectiveness in conserving target biodiversity as well as allowing for future informed conservation decisions to be made for coastal developments (Murphy & Jenkins, 2010).

This research explores metadata taken from BRUV deployments collected between 2011 and 2018 from various habitats across South/South-West England and Wales, UK. Data was compiled into a database and analysed by REJ to identify what species are recorded and absent using BRUV methods in UK waters as well as explores the influences of methodological and environmental factors on species richness, relative abundance, and faunal composition. The aim of these findings is to provide an insight into influences of BRUV methods used on data collected around the North-Atlantic Region, and provide a platform for the development of stricter, more consistent guidelines for the deployment of BRUVs.

2.2. Methods and Materials

2.2.1. Database Compilation

Data used for this research were taken from the archives of the following institutions: Swansea University; Ocean Ecology Limited; and Bournemouth University. This data was then supplemented with additional data collected in the UK during this PhD (Chapters 3 and 4) between October 2017 and August 2018. In total, 39 BRUV surveys accumulating 457 BRUV deployments, from 26 locations around the UK in depths approximately between 4-30m (Fig. 1) were compiled into one database for analysis.

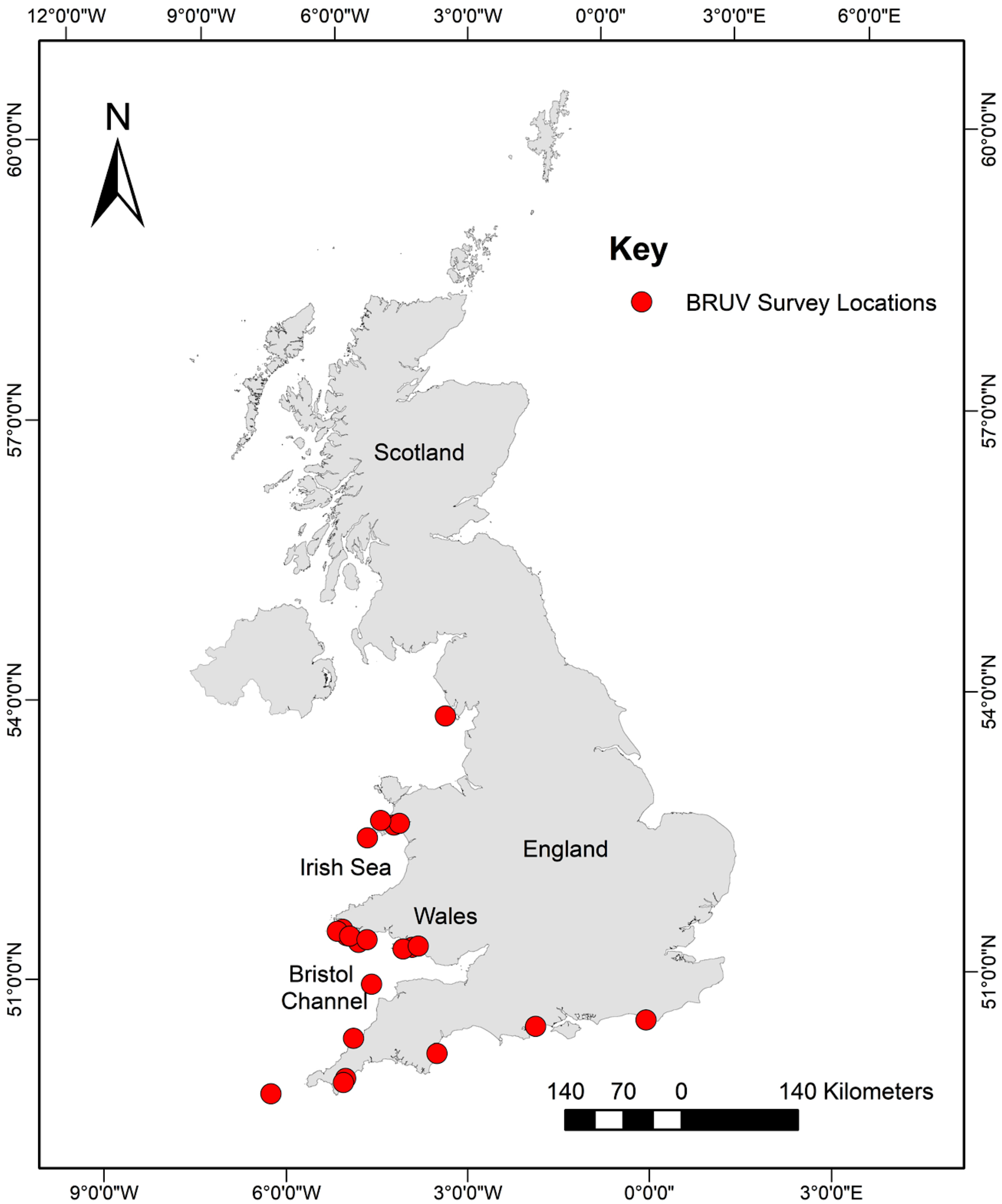

Figure 1: Map showing the locations of the BRUV surveys which form the database used in this analysis.

For this study, relative abundance referred to the maximum number (*MaxN*) of individuals of a family or species present in any one frame of the video recorded. This measure has been extensively used in BRUV research (Whitmarsh et al., 2017) to avoid repeated counts of individuals (Grimmel et al., 2020; Priede et al., 1994).

The following key metadata were extracted from each deployment where available: Habitat observed; Time of deployment; Bait type; Duration of deployment (min); Depth (m); Image quality; Tidal state / type; Species richness; and Relative abundance (*MaxN*).

The BRUV deployments targeted a variety of benthic habitats commonly found around the UK's coastal waters including seagrass beds, sand, mixed course sediments and kelp beds and also used a variety of bait types including mackerel, squid, sardines, fish meal and prawn. Where access to the raw video footage was possible, deployments were categorised by time of day using the following criteria: Dawn, Day, Evening, Night. Deployments undertaken in complete daylight or darkness were categorised as day and night respectively with deployments undertaken during the transitioning period from night to day and vice versa categorised as dawn and evening respectively (Table 1).

Table 1: List of metadata used during this study

Image Quality	**Tide**
Excellent (n= 52)	Mid (n= 115)
Good (n=106)	Neap (n= 95)
Poor (n=133)	Spring (n= 106)
Unusable (n= 19)	N/A (n= 141)
N/A (n= 147)	**Tidal State**
Habitat	High to Low (Ebb) (n= 74)
Artificial Reef (n= 25)	Low to High (Food) (n= 76)
Chalk Reef (n= 2)	N/A (n= 307)
Kelp (n= 42)	**Slack**
Midwater (n= 33)	No (n= 103)
Mixed Coarse Sediment (n= 37)	Yes (n= 47)
Mussel Beds (n= 2)	N/A (n= 307)
Rocky Reef (n= 25)	**Duration (mins)**
Sand (n= 158)	10 (n= 1)
Seagrass (n= 130)	20 (n= 50)
N/A (n= 3)	30 (n= 3)
Time of Day	60 (n= 342)
Dawn (n=16)	90 (n= 2)
Day (n= 290)	100 (n= 2)
Evening (n=42)	120 (n= 34)
Night (n= 19)	140 (n= 2)
N/A (n= 90)	150 (n= 1)
Bait	180 (n= 8)
Crab (n= 16)	240 (n= 6)
Mackerel (n= 244)	360 (n= 6)
None (n= 35)	
Oily Fish Meal and Oils (n= 39)	
Prawn (n= 11)	
Sardines (n= 3)	
Squid (n= 34)	
N/A (n= 75)	

2.2.2. Image Quality Criteria

Defining the quality of BRUV footage is an important aspect when assessing the quality of information gathered by a deployment. Poor image quality may reduce the accuracy of identifying mobile species or render the footage unusable if deemed necessary. Table 2 presents the four categories used to determine BRUV image quality for this analysis with Fig. 2 providing examples of these categories from the raw video footage compiled for this review.

Table 2: Image quality criteria categories, code and description for BRUV footage in compiled database.

Image Category	**Image Code**	**Description**
Excellent	3	Can clearly see the bait plus over 1m into the distance.
Good	2	Can clearly see the bait and maximum 1m into the distance.
Poor	1	Can see the bait only.
Unusable	0	Unable to see bait and fish not clearly visible for identification purposes.

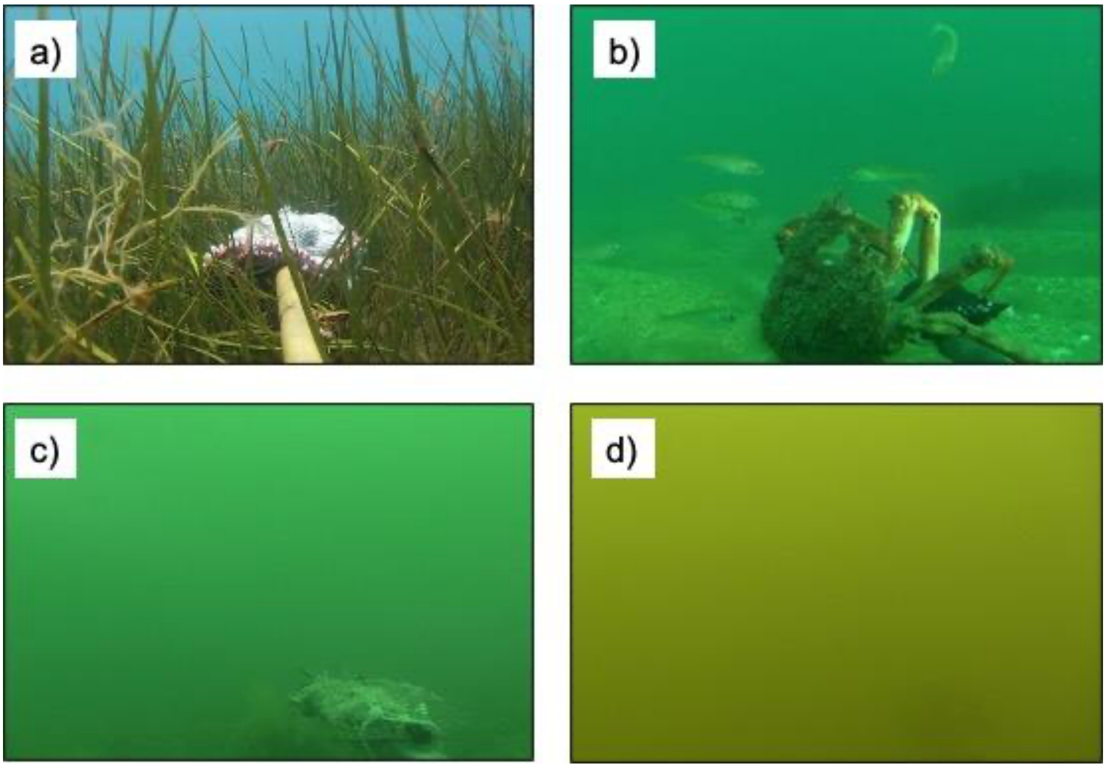

Figure 2: Examples of the image quality criteria used in developing the BRUV database. a) Excellent b) Good c) Poor d) Unusable.

2.2.3. Video Analysis

The majority of raw footage used in this analysis had already been processed for *MaxN* based on previous analytical methods used by Unsworth et al. (2014). Where footage was unprocessed, the same methodology was used. All fish assemblages and motile benthic macrofauna likely to be monitored in coastal habitats using BRUV methods were included in this analysis. Taxa were identified to the highest level possible depending on the visibility of distinguishable features. Organisms were identified as unknown if turbidity levels affected confidence of identification.

Prior to analysis, raw footage was compressed from Advanced Video Coding High Definition (AVCHD) format (standard format for digital recordings and high-definition video camcorders) to Audio Video Interleave (AVI) format using Xilisoft Video Converter Ultimate (http://www.uk.xilisoft.com/) for use in the specialist SeaGIS software Event Measure (www.seagis.com.au).

2.2.4. Data Analysis

Univariate analyses were conducted using RStudio (V4.0.0) and Minitab (Version 11). Significant results were considered $P \leq 0.05$ and all means reported ± 1 Standard Error (SE). Where data were unobtainable, cells were left blank. The following categorical predictors were assessed; habitat observed, image quality, time of day, bait type, tide type (spring/neap), tidal state (high to low, low to high) and slack tide (yes or no). Depth was not used in this analysis due to the low number of BRUV deployments recording it as metadata. Categorical predictors were coded (1,0) with reference levels for the categorical predictors with more than two levels as follows: Broad scale habitat = sand, image quality = excellent, time of day = day, bait type = none, spring/neap tide = mid. Duration of deployment was included as a continuous predictor in this assessment.

Species richness and relative abundance (*MaxN*) were square root transformed prior to analysis to reduce variance heterogeneity. Generalized linear models using a Poisson regression (*family = poisson, link = log*) were fitted to evaluate the influence of BRUV categorical predictors. Prior to running the models, we examined whether there was multicollinearity between any of the predictors based on the variance inflation factor (VIF) using the '*car*' package in R. An aliased test was also carried out to identify which variables were linearly dependent on others (subsequently causing perfect multicollinearity). Based on this test, the slack tide predictor variable was removed as it was shown to be highly correlated with tidal state for both species richness and relative abundance. Once removed, the remaining seven predictors all presented VIF values <3 (Zuur et al., 2010). A base model was initially created using the seven remaining predictors. A stepwise regression following a sequential replacement was then used to find the subset of variables resulting in the best performing model. A chi-square goodness of fit test was used to determine how well the theoretical distribution fit the empirical distribution.

Multivariate analysis on faunal assemblage composition was undertaken on square-root transformed species data using PRIMER-e v7 plus PERMANOVA+ software (Clarke & Gorley, 2007). Bray-Curtis dissimilarity matrices (including a dummy variable to account for deployments with no fauna) were created prior to conducting statistical analyses and visualised using non-metric multi-dimensional scaling (nMDS) plots. Following this, a two-way permutational multivariate analysis of variance (PERMANOVA) (Anderson, 2017) was used to test for interactions between habitat observed and the remaining categorical predictors on faunal assemblage composition. These tests were based on 9999 unrestricted permutations of the raw data with significant results considered $P \leq 0.01$ to account for low sample sizes for some parameters assessed. Pairwise comparisons were subsequently run on the significant interactions between the habitat observed and the categorical predictors identified by the PERMANOVA. Two-way similarity percentages (SIMPER) analyses were used to identify the main species recorded on the BRUVs responsible for any differences between habitat observed and the significant interactions between the remaining categorical predictors identified in the PERMANOVA.

2.3. Results

2.3.1. General Observations

Out of the 457 BRUV deployments used in this assessment, 16 were subject to extreme low visibility (4%) and categorised as unusable, 13 toppled into the sediment due to strong currents (3%) and 5 failed due to a camera fault (1%) totalling 34 failed deployments.

Access to raw footage of 147 deployments were unavailable (32%). Image quality of these were therefore classified as N/A. Of the remaining deployments, 52 were considered excellent image quality (12%), 106 considered good image quality (23%), 133 considered poor image quality (29%) and 19 were considered unusable (4%), 16 of which were due to excessive low visibility conditions where the bait was not visible. Duration of deployments all ranged from 10 minutes to 360 minutes; 75% of BRUV deployments were 60 minutes.

Nine habitats were targeted across the 457 deployments; 158 BRUV deployments were located on sand (35%), 130 in seagrass (28%), 42 in kelp (9%), 37 on mixed coarse sediments (8%), 33 in midwater (7%), 25 on artificial reefs (6%), 25 on rocky reefs (6%), 2 on chalk reefs (<1%) and 2 on mussel beds (<1%). Habitat data were unavailable for 3 deployments (<1%) and were therefore classed as N/A. For time of day, 290 BRUV deployments were undertaken during daylight hours (64%), 19 were undertaken at night (4%), 16 were undertaken at dawn (4%) and 42 were undertaken in the evening (9%). Times of day were unavailable for 90 deployments; these were therefore classified as N/A (19%).

Across all deployments, seven bait types were utilised: 244 used mackerel (53%), 39 oily fish meal (9%), 35 no bait (8%), 34 used squid (7%), 16 crab (4%), 11 prawn (2%), and 3 sardines (1%). Data were unavailable for 75 deployments and were therefore classified N/A (16%).

For tide type, 115 were undertaken during mid tides (25%) 106 BRUV deployments during spring tides (23%), and 95 during neap tides (21%). Deployment dates were unavailable for 141 deployments, these were therefore classified as N/A (31%). The tidal state was running high to low (ebb) for 74 deployments (16%), low to high (flood) for 76 deployments (17%) with 307 classified as N/A (67%). Out of the 457 deployments, 47 were undertaken on slack tide (10%) with 103 not (23%) and 307 classified as N/A (67%).

In total, 88 different taxa from 43 families (fish, molluscs and crustacea) were recorded throughout the 39 BRUV surveys (Supplementary information Table 1). Lesser spotted dogfish (*Scyliorhinus canicula)* was recorded within 113 BRUV deployments across all surveys with the two-spotted goby *Gobiusculus flavescens* also recorded in a high number of deployments at 93. Out of the 43 families recorded across all surveys, those highest represented by different species were

Labridae, Gadidae and Gobiidae with six species each. A number of families were notably represented by a lower number of species, these included Clupeidae, Scombridae, Sparidae, Soleidae, Gasterosteidae and Rajidae with one species each. Cryptic (morphologically indistinguishable) species such as those from the Syngnathidae were difficult to identify to species level. Pelagic and mid-water species were not recorded in high numbers during the 457 BRUV deployments used in this research.

2.3.2. Species Richness

Following a stepwise regression analysis, only one predictor, bait type was identified in the best performing generalized linear model *(family = poisson)* for species richness during BRUV deployments with an R^2 value of 0.26 and AIC value of 328.64 (Table 3, Fig. 3). A chi-square goodness of fit test for this model returned $P = 1.00$ suggesting that there is no evidence that the data does not follow a Poisson distribution.

Observations of the coefficients within the bait predictor (Supplementary Material, Section B, Table 1) showed that oily fish meal and fish oils (1.160, $P =$ <0.001) and mackerel (0.830, $P = 0.006$) had the largest positive influences over species richness in BRUV deployments compared to unbaited deployments (Fig. 3d).

Table 3: Best performing models assessing what predictors have the most influence over species richness during BRUV deployments based on AIC values. Note N/A has been excluded from this statistical analysis.*

Predictors	AIC	AICc	R^2
Base model: All 7 predictors	346.05	352.45	0.35
Bait type	**328.64**	**329.34**	**0.26**
Bait type + Tidal state	330.62	331.57	0.26
Bait type + Tidal State + Image Quality	331.43	332.99	0.30
Broad habitat + Bait type + Tidal State	333.06	334.97	0.31
Bait type + Tidal State + Spring / Neap	334.23	335.79	0.26
Habitat Observed + Bait type + Tidal State + Duration	334.31	336.62	0.32
Habitat Observed + Bait type + Tidal State + Spring / Neap + Duration	334.69	336.60	0.28
Habitat Observed + Bait type + Tidal State + Image Quality	335.44	338.20	0.33
Habitat Observed + Bait type + Tidal State + Spring / Neap	336.82	339.58	0.31
Habitat Observed + Bait type + Tidal State + Image Quality + Duration	336.82	340.07	0.34
Habitat Observed + Bait Type + Tidal State + Time of Day	338.79	342.04	0.31
Duration + Tidal State	339.68	339.87	0.11
Habitat Observed + Bait Type + Tidal State + Time of Day + Duration	340.11	343.89	0.32
Habitat Observed + Bait Type + Tidal State + Time of Day +Image Quality	341.06	345.43	0.34
Habitat Observed + Bait Type + Tidal State + Time of Day + Image Quality + Duration	342.51	347.50	0.35

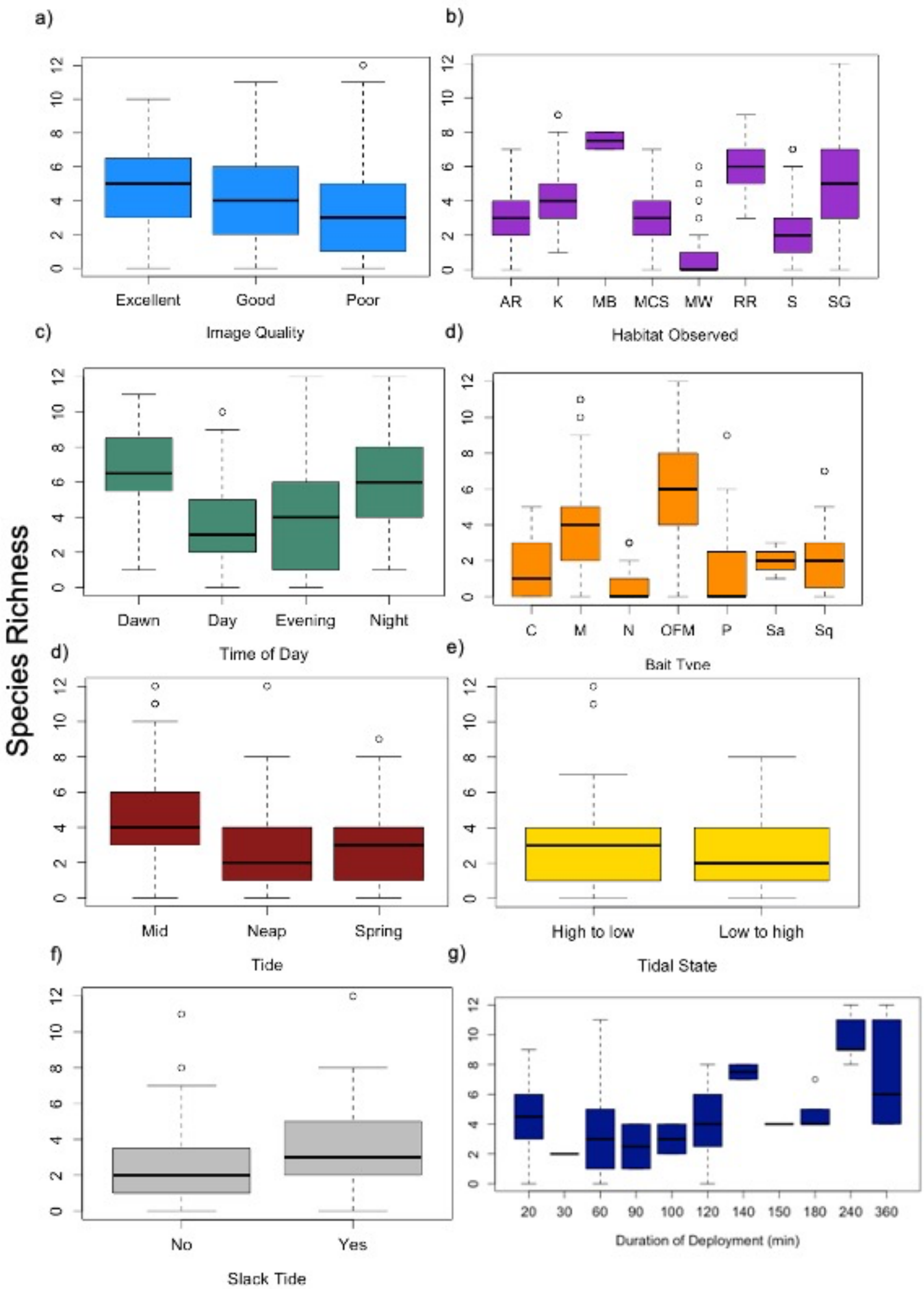

Figure 3: Boxplot (box ranging from first to third quartile and highlighting median value, whiskers extending to 1.5 the interquartile distance with circles indicating outliers) comparing species richness recorded for a) image quality b) broad habitat observed c) time of day d) bait type e) Spring / Neap tides f) Tidal State g) Slack tide h) duration of deployment. AR = Artificial Reef, K = Kelp, MB = Mussel Beds, MCS = Mixed Coarse Sediment, MW = Midwater, RR = Rocky Reef, S = Sand, SG = Seagrass, C = Crab, M = Mackerel, OFM = Oily fish meal and fish oils, N = No bait, P = Prawn, Sa = Sardines, Sq = Squid. Note N/A and unusable deployments have been excluded from this figure.*

2.3.3. Relative Abundance

Following a stepwise regression analysis, three predictors; bait type, image quality and deployment duration were included in the best performing generalized linear model *(family = poisson)* for relative abundance during BRUV deployments with an R^2 value of 0.36 and an AIC value of 416.61 (Table 4). A chi-square goodness of fit test for this model returned $P = 0.849$ suggesting that there is no evidence that the data does not follow a Poisson distribution.

Table 4: Best performing models assessing what predictors have the most influence over relative abundance (MaxN) during BRUV deployments based on AIC values. Note N/A has been excluded from this statistical analysis.*

Predictors	AIC	AICc	R^2
Base model: All 7 predictors	426.66	433.05	0.41
Bait type + Image Quality + Duration	**416.61**	**418.16**	**0.36**
Bait type + Image Quality + Duration + Tidal State	418.61	420.52	0.36
Bait type + Image Quality + Duration + Tidal State + Spring / Neap	420.35	423.11	0.38
Bait type + Image Quality + Duration + Tidal State + Habitat Observed	421.65	424.90	0.38
Bait type + Image Quality + Duration + Tidal State + Time of Day	422.56	425.81	0.37
Bait type + Image Quality + Duration + Tidal State + Spring / Neap + Habitat Observed	423.25	427.62	0.40
Bait type + Image Quality + Duration + Tidal State + Time of Day + Spring / Neap	423.61	427.97	0.39

Observations of the coefficients for the three predictors are presented in the Supplementary Material, Section B, Table 2. For the image quality predictor, coefficients showed that deployments recording a poor image quality had a larger negative influence (-0.894, $P = 0.032$) compared to deployments of excellent image quality (Fig. 4a). For the bait predictor, coefficients again showed that oily fish meal and fish oils (0.992, $P = 0.003$) and mackerel (0.970, $P = <0.001$) had the largest positive influences over relative abundance BRUV deployments compared to unbaited deployments (Fig. 4d). Duration of deployment was also found to have a positive effect over relative abundance ($P = 0.003$) (Fig. 4h).

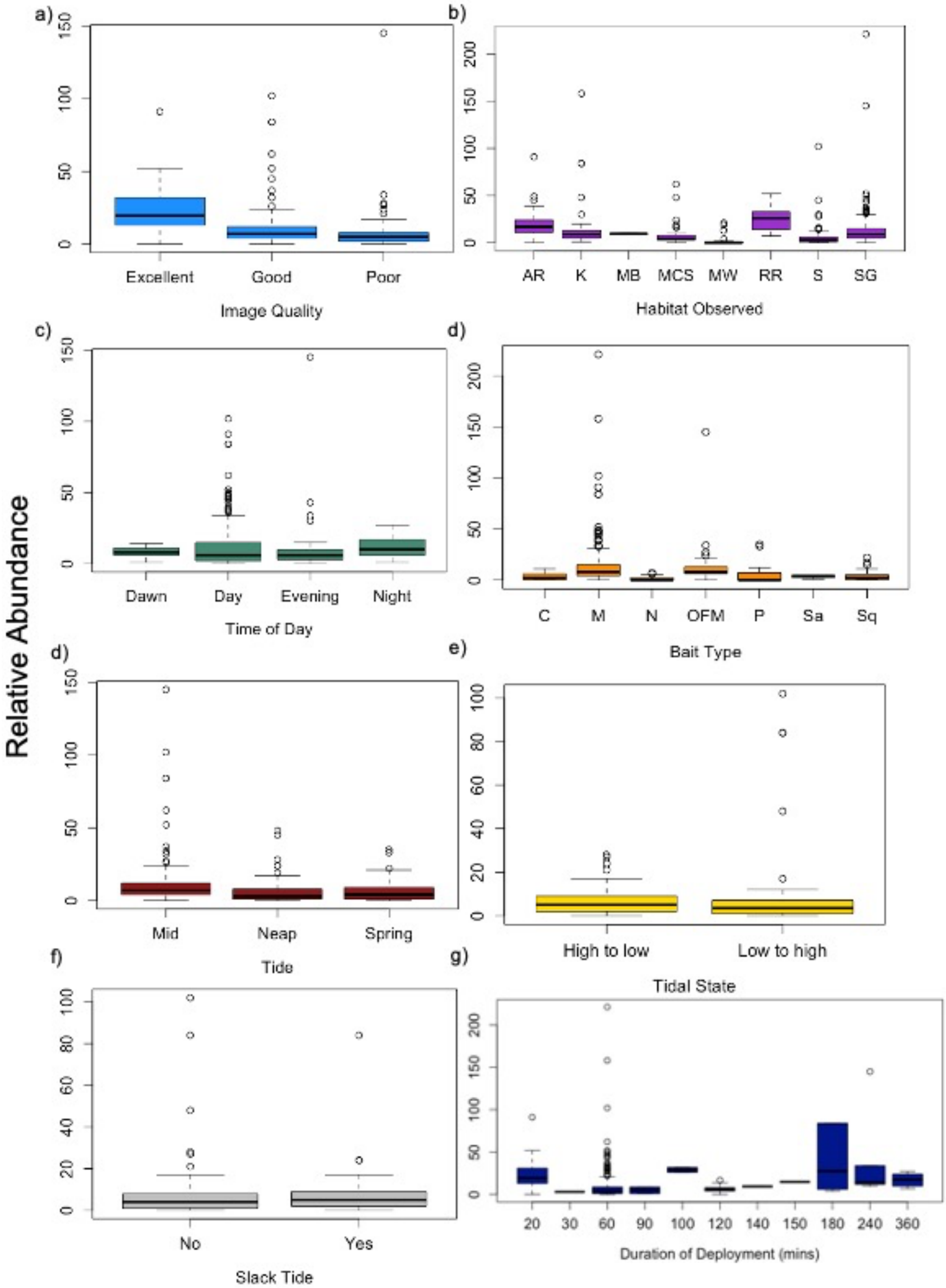

Figure 4: Boxplot (box ranging from first to third quartile and highlighting median value, whiskers extending to 1.5 the interquartile distance with circles indicating outliers) comparing relative abundance recorded for a) image quality b) broad habitat observed c) time of day d) bait type e) Spring / Neap tides f) Tidal State g) Slack tide h) duration of deployment. AR = Artificial Reef, K = Kelp, MB = Mussel Beds, MCS = Mixed Coarse Sediment, MW = Midwater, RR = Rocky Reef, S = Sand, SG = Seagrass, C = Crab, M = Mackerel, OFM = Oily fish meal and fish oils, N = No bait, P = Prawn, Sa = Sardines, Sq = Squid. Note N/A and unusable deployments have been excluded from this figure.*

2.3.4. Faunal Assemblage Composition

A PERMANOVA test of faunal assemblage composition identified significant influences of deployment time of day, bait type and tide (spring / mid / neap) on faunal assemblage composition and their interaction with habitat observed (Table 5). In addition to this, a significant interaction was also present between the habitat observed and image quality. No significant effects or interactions were observed for tidal state and slack tide on faunal assemblage composition across BRUV deployments (Fig. 5; Table 5).

Pairwise comparisons (Supplementary Material, Section C, Table 1) for the interaction between habitat observed and image quality identified significant differences in faunal assemblage composition between poor and good image qualities within sand (t = 1.8104, P = 0.003) and seagrass habitats (t = 2.896, P = <0.001). A SIMPER analysis (Supplementary Material, Section D, Table 1) identified abundances of the following taxa Gobiidae, Paguridae and *Scyliorhinus canicula* as the highest contributors (cumulative 30.70%) to differences in results from poor and good image qualities. Furthermore, as expected, unidentifiable individuals were recorded in higher abundances in poor image qualities also contributing to these differences.

For the interaction between habitat observed and deployment time of day, pairwise comparisons presented significant differences in faunal assemblage composition between day and evening deployments (t = 2.0880, P = <0.001) within sand habitat. Further differences were also identified between day and evening (t = 2.0439, P = <0.001), day and dawn (t = 2.4133, P = <0.001), day and night (t = 2.2477, P = <0.001) and dawn and night (t = 3.3442, P = <0.001) BRUV deployments within seagrass habitats. Differences between day and night (t = 2.0131, P = 0.003) within kelp habitats were also observed. A SIMPER analysis identified higher abundances of the following species *Atherina presbyter, Gobiusculus flavescens* and Ammodytidae as the main contributors to differences in results in faunal composition.

Pairwise comparisons for the interaction between habitat observed and bait type identified significant differences in faunal composition between mackerel and unbaited deployments within mixed coarse sediments (t = 1.7949, P = 0.001), sand (t = 2.9881, P = <0.001), seagrass (t = 2.3400, P = <0.001) and midwater habitats

(t = 2.3687, P = <0.010). Differences were also identified between mackerel and squid (t = 2.0463, P = <0.001) mackerel and crab (t = 1.9546, P = 0.001) squid and no bait (t = 2.1969, P = <0.001) and crab and no bait (t = 2.0300, P = 0.009) deployments within sand habitats. Differences in faunal assemblage composition between mackerel and oily fish meal were also observed within seagrass (t = 3.8963, P = <0.001) and kelp (t = 2.3184, P = <0.001) habitats. Furthermore, differences between oily fish meal and no bait (t = 2.4533, P = <0.001) prawn (t = 2.1780, P = <0.001) and squid (t = 1.9675, P = 0.001) were observed in seagrass habitats. A SIMPER analysis identified abundances of Gobiidae, *Scyliorhinus canicula*, Paguridae and *Merlangius merlangius* as the species most contributing differences in results between bait types.

Except for midwater habitats, pairwise comparisons presented a significant interaction for faunal assemblage composition between habitat observed and tide (spring / mid / neap) (Table 5). Within mixed coarse sediment and seagrass habitats, differences in composition were observed between all tide types (Supplementary Material, Section C, Table 1). Within sand habitats, differences in faunal composition were observed between spring and mid (t = 2.3531, P = <0.001) and neap and mid tides (t = 2.2792, P = <0.001). Similarly, differences between spring and mid tides were observed within kelp habitats (t = 2.0015, P = 0.007). A SIMPER analysis identified abundances of Gobiidae, *Scyliorhinus canicula*, Paguridae and *Merlangius merlangius* as the species most contributing differences in results between tides.

*Table 5: PERMANOVAs for faunal assemblage composition assessing the influence of the six categorical predictors during BRUV deployments and their interaction with habitat observed. *Note* N/A has been excluded from this statistical analysis* ***Bold values P ≤0.01***.

Source	*df*	MS	*Pseudo-F*	*P(Perm)*	*Unique Perms*
Habitat Observed * Image Quality					
Habitat Observed	6	19518	9.969	**<0.001**	9806
Image Quality	2	2361.7	1.2012	0.1965	9897
Hab * Image	5	5241.2	2.6656	**<0.001**	9838
Residual	270	1966.2			
Total	283				
Habitat Observed * Time of Day					
Habitat Observed	6	25071	12.65	**<0.001**	9830
Time of Day	3	4899.5	2.4721	**<0.001**	9858
Hab * Time	6	4212.4	2.1254	**<0.001**	9816
Residual	317	1981.9			
Total	332				
Habitat Observed * Bait Type					
Habitat Observed	7	25553	14.634	**<0.001**	9809
Bait Type	7	6663.6	3.8161	**<0.001**	9801
Hab * Bait	8	6049.2	3.4643	**<0.001**	9807
Residual	325	1746.2			
Total	347				
Habitat Observed * Tide (Spring / Neap)					
Habitat Observed	5	14915	7.5186	**<0.001**	9847
Tide	2	6924.2	3.4906	**<0.001**	9903
Hab * Tide	8	6849.6	3.4530	**<0.001**	9820
Residual	267	1983.7			
Total	282				
Habitat Observed * Tidal State					
Habitat Observed	4	10272	5.2581	**<0.001**	9885
Tidal State	1	2601.9	1.3319	0.2148	9921
Hab * State	3	1721.7	0.88133	0.6547	9896
Residual	119	1953.5			
Total	127				
Habitat Observed * Slack Tide					
Habitat Observed	4	14867	7.6298	**<0.001**	9862
Slack Tide	1	2831.9	1.4533	0.1518	9928
Hab*Slack	3	2584.9	1.3266	0.1177	9915
Residual	119	1948.5			
Total	127				

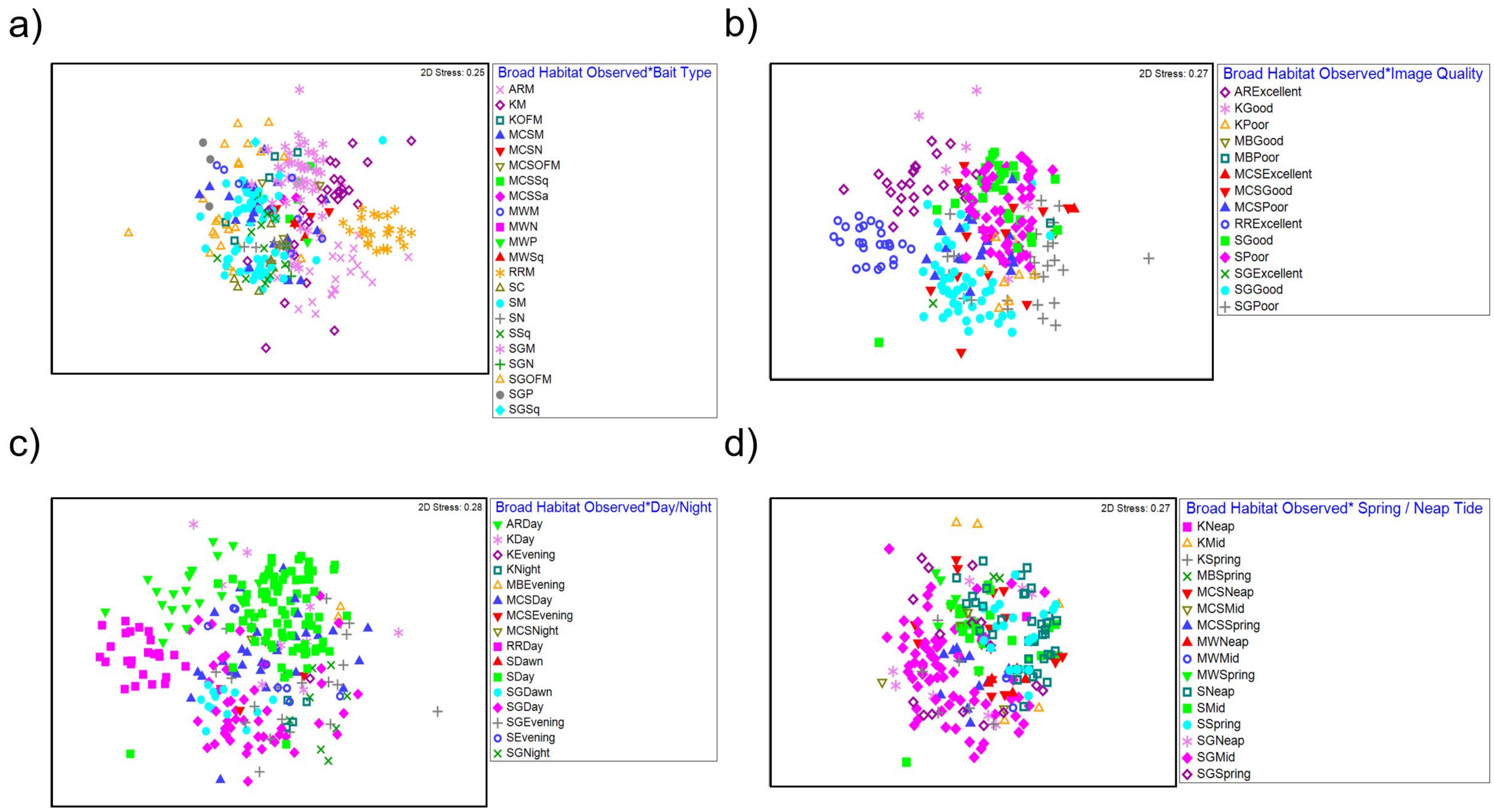

Figure 5: nMDS plot for interactions between a) broad habitat observed and bait type b) broad habitat observed and image quality c) broad habitat observed and time of day and d) broad habitat observed and spring / neap tide. AR = Artificial Reef, K = Kelp, MCS = Mixed Coarse Sediment, MW = Midwater, RR = Rocky Reef, S = Sand, SG = Seagrass, C = Crab, M = Mackerel, OFM = Oily fish meal and fish oils, N = No bait, Sa = Sardines, Sq = Squid, P = Prawn.

2.4. **Discussion**

This study provides a unique quantitative assessment of the methodological and environmental factors influencing information collected using BRUV systems in a northern temperate environment. It provides an important validation of recent reviews of BRUV protocols (Langlois et al., 2020) and will help direct sampling design when implementing BRUV assessments in the North-Atlantic Region.

Our study identified that BRUV techniques are very good tools for sampling certain fish taxonomic groups such as Labridae, Gadidae and Gobiidae. However, our findings suggest that these techniques may be less suitable for sampling families such as Clupeidae, Scombridae, Sparidae, Gasterosteidae and Rajidae. Furthermore, morphologically indistinguishable species (cryptic) also become lost when implementing these tools. Seasons should be considered when interpreting data from BRUV deployments, especially across years as these may also influence the species present as well as relative abundances in any one deployment (Sherman et al., 2020). Of the nine habitats targeted during the 457 BRUV deployments, only 7% were conducted in midwater habitats. With the underrepresentation of pelagic species across surveys, further research into the implementation of midwater BRUVs in the UK would give a better insight into their applicability for monitoring these species. As it stands, most BRUV systems have been designed to target demersal fish species (Whitmarsh et al., 2017), manipulating them to be deployed in the water column may improve their use in targeting pelagic / mid water fish assemblages.

Bait type was consistently identified as the most influential factor over species richness, relative abundance, and faunal assemblage composition. Out of the 457 BRUV deployments considered in this study, 53% used mackerel as bait suggesting that this is a favoured bait in the UK. Previous BRUV studies globally have also favoured oily fish bait types (Whitmarsh et al., 2017) and have also found it to be the best performing when undertaking experimental comparisons to unbaited deployments (Bernard & Götz, 2012; Dorman et al., 2012; Hannah & Blume, 2014; Wraith et al., 2013). Our findings also identify similar patterns with mackerel and oily fish meal having a significant positive influence over species richness and relative abundance in UK coastal waters compared to unbaited deployments. Furthermore, the amount of bait has also previously been noted in past research as influencing diversity and abundance recordings during BRUV deployments and wider techniques such as traps utilising bait (Cyr & Sainte-Marie, 1995; Hardinge et al., 2013; Miller, 1983). Any methodological inconsistencies in BRUV sampling designs must therefore be considered when implementing these methods with regards to the type and amount of bait used. The standardisation of bait use across the UK is a key factor for recording consistent ecological data over time, especially for monitoring and comparison surveys which span years over a specific area (Jones et al., 2020). Changes to bait type used may influence diversity, abundance and composition data recorded. In addition to bait type, deployment duration was also found as having an influence over relative abundance. Similar results have been identified through past research conducted in coastal habitats in the North- Atlantic Region (Unsworth et al., 2014) where minimum deployment times of 1 hour and 2 hours are required to sample 66% and 83% of fish species respectively.

Image quality had a significant negative influence on data collection with only 52 (12%) deployments classed as excellent quality in comparison to 133 (29%) deployments classified as poor quality. The low number of excellent quality images in the study was attributed to the dynamic environments associated with the UK. For instance, large tidal ranges, wave energy (Pattiaratchi & Collins, 1984) and seabed currents (Heathershaw & Langhorne, 1988) all influence large amounts of sediment transport in the water column (Pattiaratchi & Collins, 1984) in turn reducing underwater visibility. Determining when BRUV footage is useable or not is important in understanding the quality and accuracy of data recorded. At present, there are no strict guidelines on what can be classified as a useable BRUV deployment. In this study, deployments where the camera system toppled into the sediment obscuring field of view, were subject to high levels of turbidity or had a fault during deployment was classified as a failed deployment. The classification of high turbidity levels reducing image quality was measured by visualising whether the bait was visible during the video recording. Previous camera studies have found that increased turbidity levels can greatly reduce data accuracy (Mallet & Pelletier, 2014; O'Byrne et al., 2018). A potential solution to reducing the impacts of image quality when comparing BRUV deployments in low visibility environments could be to standardise the field of view when using stereo BRUVs or other low technology methods. This would allow for high-quality and low-quality images to be more comparable. For example, if the lowest useable visibility is 1m, excluding videos with visibility under that and standardise everything else to 1m by only analysing fish that are within 1m of cameras (relative abundance and species richness) in all footage (including high-quality images).

In our study, lower abundances of benthic prey species such as Gobiidae in poor quality images were identified as influencing differences in faunal assemblage composition within sand and seagrass habitats. In contrast to this, scavenging species, such as *Scyliorhinus canicula* and Paguridae were recorded in high abundances in lower quality images suggesting these are more likely to approach the bait during deployments in high turbidity. When analysing poor quality image footage, we must consider that scavenging species are more likely to approach the bait compared to smaller benthic prey species which may be located at a distance from the bait but still attracted to the wider plume (Harvey et al., 2007; Whitmarsh et al., 2018). Recent improvements in BRUV image clarity using clear liquid optical chambers (Jones et al., 2019) also provide a practical alternative in low visibility conditions, expanding the working window for BRUV methods.

The habitats targeted for BRUV deployments included both soft sediments habitats such as sand and seagrass as well as hard substrates such as rocky and artificial reefs. Contrary to initial thoughts, habitat was not identified as the most important factor influencing species richness and relative abundance during this study. However, as expected, faunal assemblage composition was heavily influenced by the habitat in which the BRUV was deployed in. Tide type and deployment time of day were also observed as having an effect over faunal composition. Past studies have identified similar effects of diurnal and tidal variation on assemblage abundances and composition in habitats such as mud and sandflats associated with estuarine environments (Morrison et al., 2002), saltmarsh creeks (Hampel et al., 2003) as well as tropical tidal flats (Reis-Filho et al., 2011). Observations or measurements of currents and tidal state during BRUV deployments should be recorded where possible as metadata as these can affect the bait plume area and potentially result in different conclusions when comparing to other datasets if these factors are not considered (Taylor et al., 2013). It was noted during this study that of the 34 failed deployments recorded, 24 (71%) occurred during spring tides where the camera footage was unable to be analysed either due to toppling into the sediment or subject to high levels of turbidity. Suspended sediment matter tends to be greater during spring tides compared to neap and mid tides limiting underwater visibility, especially around soft sediment coastal and estuarine environments (Allen et al., 1980; Grabemann et al., 1997; Uncles, 2010). Furthermore, tidal currents also tend to be stronger during spring tides (Gonzalez-Santamaria et al., 2013) increasing the likelihood of the camera system toppling. It is therefore recommended that deploying BRUVs during spring tides should be avoided.

Although, this study provides a unique overview of BRUV methods in the North-Atlantic region, it has highlighted the need for comprehensive metadata to be obtained during these surveys for BRUV datasets to be more comparable. During this research, it became apparent that the level of detail in the metadata attributed to each BRUV deployment varied. For example, depth, bait weight, tidal state, season, water temperature, and approximate distance to other habitats should be included in metadata records but was not something consistently recorded in these datasets. Past research has identified that the depth of a BRUV deployment can affect the fish assemblages sampled using BRUV methods (Bond et al., 2018). Furthermore, deploying BRUVs within soft-sediment habitats in close proximity to other habitats such as reefs or seabed infrastructure can influence halo effects (Bond et al., 2018). Fetterplace, (2017) has suggested that BRUVs should be deployed a minimum of 200m from reef habitats when sampling soft-sediment communities as a standardised means of avoiding such halo effects (Schultz et al., 2012). For this research, assumptions were made that all bait types used were the same weight for comparisons to be made. Although past research has suggested that bait weight may not always influence relative abundance, species richness or faunal assemblage composition recorded in BRUV deployments in the North – Atlantic region (Jones et al. 2020), this information is still an aspect which should be recorded as metadata and remain consistent when designing surveys using these tools.

2.4.1. Conclusions

Our findings give an insight into methodological and environmental factors which should be considered when designing and implementing BRUV techniques and have highlighted the need for comprehensive and consistent metadata to be collected during each survey for accurate temporal and spatial data comparisons. Fluctuations and variations in data may be attributed to methodological inconsistencies and/or environmental factors as well as over time or due to anthropogenic influences. All these factors must be considered when analysing and interpreting BRUV data.

Although BRUV techniques are a repeatable, cost-effective, non-destructive, widely used method, a full evaluation into whether they suitable or designed for the assemblages being targeted must be undertaken. The quality and state of the environment in which they are being deployed in must also be considered prior to conducting surveys using these tools.

Recommended guidelines for the implementation of BRUVs in the North-East Atlantic region are described at the end of this thesis in the General Discussion.

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Supplementary Information

Section A – Taxa List

*Table 1: Full list of taxa recorded through BRUV sampling around UK coastal waters and their presence in different habitats, arrival times and times of MaxN (min). SG = Seagrass, K = Kelp, S = Sand, MCS = Mixed Coarse Sediments, MW = Midwater, MB = Mussel Beds, AR = Artificial Reef, RR = Rocky Reef. *Note- Arrival times and time of MaxN have been recorded where access to raw footage / data is available.*

		Habitat									
Family	**Species**	**SG**	**K**	**S**	**MCS**	**MW**	**MB**	**AR**	**RR**	**First Arrival Time (mins) (± 1SE)**	**Time of *MaxN* (mins) (± 1 SE)**
Ammodytidae	-	x	x	x	x					46.15 ± 11.71	52.06 ± 10.90
Ammodytidae	*Ammodytes tobianus*		x							-	-
Ammodytidae	*Hyperoplus lanceolatus*	x		x						-	-
Anguillidae	*Anguilla anguilla*	x								27.03 ± 2.99	27.03 ± 2.99
Atherinidae	-	x								158.59 ± 56.45	168.78 ± 61.42
Atherinidae	*Atherina presbyter*	x	x	x	x					52.74 ± 12.56	75.23 ± 15.36
Balistidae	*Balistes capriscus*	x		x					x	31.82 ± 0.00	31.82 ± 0.00
Belonidae	*Belone belone*	x				x				126.59 ± 0.00	126.59 ± 0.00
Blenniidae	*Lipophrys pholis*		x							-	-
Blenniidae	*Parablennius gattorugine*		x						x	-	-
Callionymidae	*Callionymus lyra*	x				x				151.09 ± 51.21	151.09 ± 51.21
Cancridae	-			x						-	-

Cancridae	*Cancer pagurus*		x	x	x		x			45.73 ± 13.79	60.20 ± 11.82
Clupeidae	*Sprattus sprattus*		x	x	x					27.13 ± 13.27	41.67 ± 4.74
Cottidae	-	x								19.88 ± 0.00	190.88 ± 0.00
Cottidae	*Taurulus bubalis*	x	x							-	-
Congridae	*Conger conger*	x								-	-
Dasyatidae	*Dasyatis pastinaca*	x		x						-	-
Gadidae	-	x	x	x	x		x			39.54 ± 7.98	51.48 ± 8.90
Gadidae	*Gadus morhua*	x	x							23.15 ± 14.95	23.15 ± 14.95
Gadidae	*Merlangius merlangus*	x		x	x	x	x			35.55 ± 6.00	53.96 ± 9.06
Gadidae	*Pollachius pollachius*	x	x	x					x	29.93 ± 5.78	43.77 ± 7.88
Gadidae	*Pollachius virens*	x			x	x				30.85 ± 8.54	37.13 ± 6.85
Gadidae	*Trisopterus luscus*	x	x	x	x	x		x	x	23.77 ± 4.94	31.97 ± 5.18
Gadidae	*Trisopterus minutus*	x		x	x	x	x			31.66 ± 8.34	67.46 ± 12.27
Gasterosteidae	*Spinachia spinachia*	x	x		x					87.76 ± 20.11	112.68 ± 20.01
Gastropoda*	-	x		x	x					24.79 ± 7.94	35.47 ± 8.15
Gobiidae	-	x	x	x	x					17.60 ± 4.18	29.49 ± 7.64
Gobiidae	*Gobius niger*	x	x		x					9.18 ± 0.00	31.8 ± 0.00
Gobiidae	*Gobius paganellus*	x	x							-	-
Gobiidae	*Gobiusculus flavescens*	x	x		x	x		x	x	19.34 ± 2.85	26.06 ± 2.69
Gobiidae	*Pomatoschistus microps*	x				x				-	-
Gobiidae	*Pomatoschistus minutus*	x	x		x			x	x	12.19 ± 4.75	36.37 ± 14.82
Gobiidae	*Pomatoschistus pictus*		x							-	-

Goneplacidae	*Goneplax rhomboides*			x						-	-
Inachidae	*Macropodia*			x						46.06 ± 0.00	46.06 ± 0.00
Labridae	-	x	x	x	x					31.56 ± 4.09	37.56 ± 4.51
Labridae	*Centrolabrus exoletus*	x	x							14.38 ± 7.21	14.39 ± 7.21
Labridae	*Crenilabrus melops*	x	x							24.12 ± 14.85	27.05 ± 14.64
Labridae	*Ctenolabrus rupestris*	x	x	x	x				x	70.46 ± 48.67	70.46 ± 48.67
Labridae	*Labrus bergylta*	x	x	x	x	x		x	x	47.48 ± 11.81	53.48 ± 12.17
Labridae	*Labrus mixtus*	x	x						x	-	-
Labridae	*Symphodus melops*		x					x	x	-	-
Lotidae	-	x								238.44 ± 0.00	238.44 ± 0.00
Lotidae	*Ciliata mustela*	x							x	215.53 ± 33.36	215.53 ± 33.36
Majidae	*Maja squinado*	x	x	x	x		x	x		47.28 ± 8.34	53.91 ± 8.41
Moronidae	*Dicentrarchus labrax*		x	x				x		52.71 ± 0.00	52.71 ± 0.00
Mugilidae	*Chelon labrosus*			x				x		32.65 ± 0.00	32.65 ± 0.00
Mugilidae	-	x								131.39 ± 119.83	176.39 ± 74.83
Mullidae	*Mullus surmuletus*	x		x	x				x	18.10 ± 5.78	22.82 ± 10.50
Nephropidae	-			x	x					17.13 ± 8.41	21.35 ± 9.28
Nephropidae	*Homarus gammarus*	x	x	x	x			x	x	66.30 ± 15.78	66.76 ± 15.70
Octopodidae	-						x			41.28 ± 37.49	54.78 ± 23.99
Paguridae	-	x		x	x		x	x		26.86 ± 4.94	38.59 ± 6.33
Palaemonidae	*Palaemon serratus*	x			x			x		25.41 ± 6.28	28.48 ± 6.13
Pectinidae	-				x					58.03 ± 0.00	58.03 ± 0.00
Phocidae	*Halichoerus grypus*	x	x							30.29 ± 18.52	34.76 ± 18.19
Pleuronectidae	-	x		x						39.13 ± 9.41	48.25 ± 9.40
Pleuronectidae	*Limanda limanda*	x		x						15.28 ± 0.00	15.28 ± 0.00

Pleuronectidae	*Pleuronectes platessa*	x	x	x		x				82.99 ± 38.65	82.99 ± 38.65
Portunidae	-	x	x	x	x					22.00 ± 7.58	37.37 ± 12.16
Portunidae	*Carcinus maenas*	x			x			x		20.36 ± 3.80	30.24 ± 4.12
Portunidae	*Necora puber*	x	x		x				x	23.16 ± 9.71	23.69 ± 9.45
Rajidae	*Raja clavata*	x		x		x				40.91 ± 0.00	40.91 ± 0.00
Scombridae	*Scomber scombrus*	x								-	-
Scophthalmidae	*Scophthalmus rhombus*	x								-	-
Scyliorhinidae	-	x	x	x	x		x			58.24 ± 12.08	66.98 ± 13.58
Scyliorhinidae	*Scyliorhinus*	x								18.00 ± 4.90	17.60 ± 4.50
Scyliorhinidae	*Scyliorhinus canicula*	x	x	x	x	x				23.29 ± 2.11	27.74 ± 2.11
Scyliorhinidae	*Scyliorhinus stellaris*	x								120.1 ± 0.00	120.1 ± 0.00
Scyphozoa*	-		x							8.79 ± 0.00	8.79 ± 0.00
Sepiidae	*Sepiola atlantica*	x	x							152.26 ± 52.75	177.45 ± 44.02
Sepiidae	*Sepia officinalis*	x		x				x		26.89 ± 13.29	26.89 ± 13.29
Sepiolidae	-	x								155.14 ± 126.97	174.12 ± 145.95
Soleidae	-			x						27.57 ± 8.22	27.57 ± 8.22
Soleidae	*Solea solea*			x						47.27 ± 20.09	47.27 ± 20.09
Sparidae	-			x						23.93 ± 1.84	23.93 ± 1.84
Sparidae	*Spondyliosoma cantharus*			x				x	x	18.36 ± 13.66	23.16 ± 18.46
Syngnathidae	-	x								125.08 ± 87.75	125.08 ± 87.75
Syngnathidae	*Entelurus aequoreus*	x								3.69 ± 0.50	17.19 ± 13.00
Syngnathidae	*Syngnathus acus*	x			x					158.19 ± 65.52	158.19 ± 65.52
Syngnathidae	*Syngnathus typhle*	x	x							-	-
Trachinidae	*Echiichthys vipera*	x								57.74 ± 24.78	192.74 ± 110.22
Triglidae	-			x						16.36 ± 3.92	16.36 ± 3.92

Triglidae	*Chelidonichthys cuculus*	x								-	-
Triglidae	*Chelidonichthys lucerna*	x		x					x	32.53 ± 7.48	32.75 ± 7.30
Triakidae	*Mustelus mustelus*			x						20.81 ± 2.71	23.50 ± 3.61
Unknown	-	x	x	x	x		x			46.83 ± 10.08	51.21 ± 9.87

* Identified to Class.

Section B – Poisson Regression Coefficients

Table 1: Coefficients from the Poisson model for species richness. Reference levels for the categorical predictors were as follows: Broad scale habitat = sand, time of day = day, bait type = none, image quality = excellent, spring/neap tide = mid. ***Bold values indicate P = ≤0.05.***

Predictor	Coef	SE Coef	Z-Value	P-Values
Constant	-0.348	0.281	-1.24	0.214
Bait Type				
Crab	0.358	0.402	0.89	0.373
Mackerel	0.830	0.299	2.77	**0.006**
Oily fish meal and fish oils	1.160	0.324	3.58	**<0.001**
Sardines	0.672	0.566	1.19	0.235
Squid	0.655	0.343	1.91	0.056

Table 2: Coefficients from the Poisson model for relative abundance (MaxN). Reference levels for the categorical predictors were as follows: Broad scale habitat = sand, time of day = day, bait type = none, image quality = excellent, spring/neap tide = mid. ***Bold values indicate P = ≤0.05.***

Predictor	Coef	SE Coef	Z-Value	P-Values
Constant				
Duration of Deployment	0.00310	0.00104	2.99	**0.003**
Bait Type				
Crab	0.512	0.351	1.46	0.145
Mackerel	0.970	0.267	3.63	**<0.001**
Oily fish meal and fish oils	0.992	0.335	2.96	**0.003**
Sardines	0.811	0.517	1.57	0.117
Squid	0.680	0.307	2.21	**0.027**
Image Quality				
Good	-0.486	0.405	-1.20	0.230
Poor	-0.894	0.418	-2.14	**0.032**

Section C – Multivariate Analysis Pairwise Comparisons

*Table 1: Pairwise comparisons for faunal assemblage composition. *Note* N/A has been excluded from this statistical analysis Bold values P ≤0.01.*

Source	***t***	***P(perm)***	***Unique Perms***
Habitat Observed * Bait Type			
Mixed Coarse Sediment			
Mackerel vs No bait	1.7949	**0.001**	9034
Mackerel vs Squid	1.5046	0.0137	9060
Mackerel vs Sardines	1.2402	0.1187	1762
Mackerel Oily fish meal	1.2225	0.1244	1762
No bait vs Squid	1.036	0.439	126
No bait vs Sardines	1.4385	0.1756	56
No bait vs Oily fish meal	1.6249	0.0344	56
Squid vs Sardines	0.73517	0.7730	41
Squid vs Oily fish meal	1.5668	0.0737	56
Sardines vs Oily fish meal	1.6173	0.2013	10
Sand			
Mackerel vs No bait	2.9881	**<0.001**	9931
Mackerel vs Squid	2.0463	**<0.001**	9943
Mackerel vs Crab	1.9546	**0.001**	9944
No bait vs Squid	2.9169	**0.001**	9624
No bait vs Crab	2.0300	**0.009**	3728
Squid vs Crab	1.3740	0.1222	9870
Seagrass			
Mackerel vs No bait	2.3400	**<0.001**	9802
Mackerel vs Squid	1.5085	0.0175	8304
Mackerel vs Oily fish meal	3.8963	**<0.001**	9924
Mackerel vs Prawn	2.3680	**<0.001**	8259
Squid vs Oily fish meal	1.9675	**0.0010**	3112
Squid vs Prawn	1.5043	0.1059	10
Squid vs No bait	2.0670	0.0273	35
Oily fish meal vs Prawn	2.1780	**<0.001**	3138
Oily fish meal vs No bait	2.4533	**<0.001**	8082
Prawn vs No bait	2.7367	0.0298	35
Kelp			
Mackerel vs Oily fish meal	2.3184	**<0.001**	9780
Midwater			
Mackerel vs No bait	2.3687	**0.010**	191
Mackerel vs Squid	2.0674	0.0237	992
Mackerel vs Prawn	2.2428	0.0115	280
Squid vs Prawn	1.2241	0.3958	6
Squid vs No bait	1.271	0.4742	3
Prawn vs Nothing	1.0000	1.0000	1
Habitat Observed * Image Quality			
Mixed Coarse Sediment			
Poor vs Good	1.3509	0.0594	9923
Poor vs Excellent	1.4802	0.049	22
Good vs Excellent	1.3046	0.1274	16
Sand			
Poor vs Good	1.8104	**0.0025**	9927
Kelp			
Poor vs Good	1.6619	0.0182	5114
Seagrass			
Poor vs Good	2.6896	**<0.001**	9943
Poor vs Excellent	1.223	0.1066	28
Good vs Excellent	0.88415	0.6112	47
Habitat Observed * Time of Day			
Mixed Coarse Sediment			
Day vs Evening	0.71279	0.8853	9012

Day vs Night	1.0467	0.450	33
Evening vs Night	0.9647	0.7963	5
Sand			
Day vs Evening	2.0880	**<0.001**	9918
Day vs Dawn	0.78797	0.7617	83
Evening vs Dawn	0.88683	0.6508	9
Seagrass			
Day vs Evening	2.0439	**<0.001**	9937
Day vs Dawn	2.4133	**<0.001**	9936
Day vs Night	2.2477	**<0.001**	9922
Evening vs Dawn	2.8219	**<0.001**	9930
Evening vs Night	1.3174	0.0701	9927
Dawn vs Night	3.3442	**<0.001**	9865
Kelp			
Day vs Evening	1.0777	0.3623	11
Day vs Night	2.0131	**0.0028**	1001
Evening vs Night	2.3285	0.2026	5
Habitat Observed * Tide (Spring / Neap)			
Mixed Coarse Sediment			
Spring vs Neap	1.8047	**0.0021**	9944
Spring vs Mid	2.1161	**<0.001**	9625
Neap vs Mid	1.7190	**<0.001**	9644
Sand			
Spring vs Neap	1.6237	0.020	9959
Spring vs Mid	2.3531	**<0.001**	9952
Neap vs Mid	2.2792	**<0.001**	9943
Seagrass			
Spring vs Neap	2.2807	**<0.001**	9926
Spring vs Mid	2.3253	**<0.001**	9940
Neap vs Mid	2.1497	**<0.001**	9938
Kelp			
Spring vs Neap	1.7477	0.034	119
Spring vs Mid	2.0015	**0.0066**	792
Neap vs Mid	2.0605	0.0168	56
Midwater			
Spring vs Neap	2.0002	0.0272	550
Spring vs Mid	1.25	0.3137	57
Neap vs Mid	1.8477	0.0282	86

Section D – Multivariate Analysis SIMPER Results

Table 1: SIMPER analysis in groups outlined by PERMANOVA showing the top five organisms which most contributed to the observed differences among broad habitats observed. MSC = Mixed coarse sediment, OFM = Oily fish meal and fish oils.

Species	Av. Abun.	Av. Abun.	Av. Diss.	Diss./SD	Contrib. %	Cum %
Habitat Observed x Bait Type						
Av. Diss.: 87.43	**MCS**	**Sand**				
Gobiidae	0.33	0.40	9.51	0.71	10.88	10.88
Scyliorhinus canicula	0.62	0.34	7.93	0.84	9.07	19.95
Pollachius pollachius	0.61	0.05	7.04	0.47	8.06	28.00
Paguridae	0.11	0.53	6.37	0.58	7.29	35.29
Merlangius merlangius	0.15	0.30	5.57	0.67	6.37	41.66
Av. Diss.: 88.87	**MCS**	**Seagrass**				
Gobiusculus flavescens	0.04	0.61	7.76	0.87	8.73	8.73
Pollachius pollachius	0.61	0.44	7.30	0.84	8.21	16.94
Scyliorhinus canicula	0.62	0.56	6.44	0.87	7.25	24.19
Atherina presbyter	0.08	0.64	5.86	0.56	6.59	30.79
Labridae	0.26	0.22	5.37	0.59	6.04	36.83
Av. Diss.: 94.10	**MCS**	**Kelp**				
Gobiusculus flavescens	0.04	1.04	10.01	0.89	10.63	10.63
Pollachius pollachius	0.61	0.77	8.84	0.83	9.39	20.02
Scyliorhinus canicula	0.62	0.16	6.08	0.93	6.46	26.48
Gobiidae	0.33	0.03	5.46	0.55	5.80	32.29
Labrus bergylta	0.03	0.47	4.68	0.86	4.98	37.27
Av. Diss.:93.55	**MCS**	**Midwater**				
Pollachius pollachius	0.61	0.10	15.03	0.65	16.06	16.06
Scyliorhinus canicula	0.62	0.22	12.49	0.76	13.36	29.42
Trisoperus luscus	0.25	0.11	8.68	0.56	9.28	38.70
Merlangius merlangius	0.15	0.34	8.15	0.73	8.7	47.41
Gobiidae	0.33	0.00	5.09	0.42	5.44	52.85
Av. Diss.: 95.50	**MCS**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	22.36	1.32	23.41	23.41
Maja squinado	0.19	1.16	10.70	1.04	11.21	34.62
Gobiusculus flavescens	0.04	0.90	6.82	0.72	7.14	41.77
Scyliorhinus canicula	0.62	0.00	6.08	0.93	6.37	48.13
Gobiidae	0.33	0.00	5.38	0.52	5.64	53.77
Av. Diss.:98.06	**MCS**	**Rocky reef**				
Ctenolabrus rupestris	0.06	2.89	18.59	3.25	18.96	18.96
Trisopterus luscus	0.25	2.33	15.56	2.45	15.87	34.83
Parablennius gattorugine	0.00	1.24	8.35	2.18	8.52	43.35
Spondyliosoma cantharus	0.00	1.46	8.15	0.88	8.31	51.66
Symphodus melops	0.00	0.71	4.23	1.14	4.31	55.97
Av. Diss.: 92.93	**Sand**	**Seagrass**				
Gobiusculus flavescens	0.00	0.61	8.73	0.87	9.39	9.39

Scyliorhinus canicula	0.34	0.56	7.60	0.85	8.18	17.57
Gobiidae	0.40	0.06	6.56	0.53	7.06	24.63
Merlangius merlangius	0.30	0.45	5.67	0.69	6.10	30.73
Pollachius pollachius	0.05	0.44	5.59	0.88	6.01	36.74
Av. Diss.: 97.00	**Sand**	**Kelp**				
Gobiusculus flavescens	0.00	1.04	11.06	0.89	11.41	11.41
Pollachius pollachius	0.05	0.77	7.97	0.79	8.22	19.63
Gobiidae	0.40	0.03	6.68	0.56	6.89	26.52
Scyliorhinus canicula	0.34	0.16	5.60	0.75	5.77	32.29
Labrus bergylta	0.00	0.47	5.11	0.86	5.27	37.55
Av. Diss.: 94.78	**Sand**	**Midwater**				
Paguridae	0.53	0.00	11.38	0.52	12.00	12.00
Merlangius merlangius	0.30	0.34	10.55	0.82	11.13	23.14
Scyliorhinus canicula	0.34	0.22	9.94	0.74	10.49	33.63
Gobiidae	0.40	0.00	8.00	0.47	8.44	42.07
Mustelus mustelus	0.18	0.00	6.26	0.38	6.60	48.68
Av. Diss.: 96.0	**Sand**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	23.98	1.32	24.95	24.95
Maja squinado	0.12	1.16	12.24	1.00	12.73	37.68
Gobiusculus flavescens	0.00	0.90	7.32	0.72	7.61	45.30
Gobiidae	0.40	0.00	6.65	0.51	6.92	52.22
Paguridae	0.53	0.25	5.93	0.60	6.17	58.39
Av. Diss.: 98.71	**Sand**	**Rocky reef**				
Ctenolabrus rupestris	0.00	2.89	20.56	3.54	20.80	20.80
Trisopterus luscus	0.10	2.33	16.50	2.39	16.69	37.49
Parablennius gattorugine	0.00	1.24	8.89	2.15	8.99	46.48
Spondyliosoma cantharus	0.03	1.46	8.60	0.89	8.70	55.18
Symphodus melops	0.00	0.57	3.94	0.80	3.98	67.89
Av. Diss.: 84.33	**Seagrass**	**Kelp**				
Gobiusculus flavescens	0.61	1.04	9.70	0.92	11.51	11.51
Pollachius pollachius	0.44	0.77	7.24	0.89	8.59	20.10
Scyliorhinus canicula	0.56	0.16	5.67	0.83	6.73	26.82
Atherina presbyter	0.64	0.23	4.95	0.49	5.87	32.70
Labrus bergylta	0.10	0.47	4.32	0.82	5.13	37.82
Av. Diss.: 91.20	**Seagrass**	**Midwater**				
Merlangius merlangius	0.45	0.34	10.86	0.91	11.91	11.91
Scyliorhinus canicula	0.56	0.22	10.83	0.88	11.87	23.78
Gobiusculus flavescens	0.61	0.04	8.74	0.76	9.58	33.36
Pomatoschistus microps	0.15	0.11	6.24	0.42	6.84	40.21
Pollachius pollachius	0.44	0.10	6.22	0.66	6.82	47.03
Av. Diss.: 94.64	**Seagrass**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	21.02	1.27	22.21	22.21
Maja squinado	0.09	1.16	10.77	1.02	11.38	33.59
Gobiusculus flavescens	0.61	0.90	9.45	0.94	9.98	43.58
Scyliorhinus canicula	0.56	0.00	6.00	0.78	6.34	49.91
Pollachius pollachius	0.44	0.00	4.65	0.86	4.91	54.2
Av. Diss.: 93.94	**Seagrass**	**Rocky reef**				

Ctenolabrus rupestris	0.00	2.89	18.28	3.09	19.46	19.46
Trisopterus luscus	0.24	2..33	12.55	1.69	13.36	32.82
Parablennius gattorugine	0.00	1.24	7.89	2.06	8.40	41.22
Spondyliosoma cantharus	0.00	1.46	7.75	0.87	8.25	49.47
Gobiusculus flavescens	0.61	0.04	5.27	0.92	5.61	55.07
Av. Diss.: 95.72	**Kelp**	**Midwater**				
Gobiusculus flavescens	1.04	0.04	11.98	0.82	12.51	12.51
Merlangius merlangius	0.00	0.34	9.39	0.81	9.81	22.32
Pollachius pollachius	0.77	0.10	9.05	0.73	9.46	31.78
Scyliorhinus canicula	0.16	0.22	6.96	0.84	7.27	39.05
Labrus bergylta	0.47	0.04	5.35	0.82	5.59	44.64
Av. Diss.: 92.17	**Kelp**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	20.61	1.25	22.36	22.36
Gobiusculus flavescens	1.04	0.90	10.83	0.95	11.75	34.11
Maja squinado	0.14	1.16	10.11	0.99	10.97	45.08
Pollachius pollachius	0.77	0.00	6.51	0.73	7.06	52.14
Pomatoschistus minutus	0.29	0.40	4.65	0.59	5.04	57.18
Av. Diss.: 87.09	**Kelp**	**Rocky reef**				
Ctenolabrus rupestris	0.39	2.89	15.40	2.07	17.10	17.10
Trisopterus luscus	0.10	2.33	14.21	2.13	15.78	32.88
Spondyliosoma cantharus	0.00	1.46	7.63	0.86	8.47	41.36
Parablennius gattorugine	0.04	1.24	7.58	1.92	8.42	49.78
Gobiusculus flavescens	1.04	0.04	6.68	0.92	7.42	57.19
Av. Diss.: 98.99	**Midwater**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	25.10	1.24	25.36	25.36
Maja squinado	0.00	1.16	14.66	0.87	14.81	40.17
Merlangius merlangius	0.34	0.00	9.39	0.81	9.48	49.65
Gobiusculus flavescens	0.04	0.90	8.05	0.74	8.13	57.78
Scyliorhinus canicula	0.22	0.00	5.83	0.83	5.89	63.68
Av. Diss.:95.77	**Midwater**	**Rocky Reef**				
Ctenolabrus rupestris	0.00	2.89	21.15	2.85	22.09	22.09
Trisopterus luscus	0.11	2.33	15.06	1.72	15.72	37.81
Parablennius gattorugine	0.00	1.24	9.14	1.89	9.55	47.36
Spondyliosoma cantharus	0.00	1.46	8.70	0.86	9.08	56.44
Merlangius merlangius	0.34	0.00	7.01	0.85	7.32	63.76
Av. Diss.: 88.00	**Art. Reef**	**Rocky Reef**				
Ctenolabrus rupestris	0.00	2.39	17.88	2.95	20.32	20.32
Spondyliosoma cantharus	2.87	1.46	14.47	1.32	16.44	36.76
Trisopterus luscus	0.04	2.33	14.17	2.13	16.10	52.87

Parablennius gattorugine	0.00	1.24	7.72	2.01	8.77	61.64
Maja squinado	1.16	0.00	7.18	1.28	8.16	69.80
Av. Diss.: 96.31	**Mackerel**	**No Bait**				
Gobiidae	0.21	0.00	11.99	0.47	12.45	12.45
Scyliorhinus canicula	0.41	0.09	10.33	0.71	10.73	23.18
Paguridae	0.16	0.12	8.75	0.52	9.08	32.36
Merlangius merlangius	0.28	0.00	7.41	0.58	7.69	39.95
Unknown	0.06	0.12	5.76	0.44	5.99	45.93
Av. Diss.: 85.75	**Mackerel**	**Squid**				
Paguridae	0.16	0.44	10.69	0.69	12.47	12.47
Gobiidae	0.21	0.13	9.30	0.52	10.85	23.32
Scyliorhinus canicula	0.41	0.45	9.12	0.80	10.64	33.95
Mustelus mustelus	0.03	0.33	8.26	0.61	9.64	43.59
Merlangius merlangius	0.28	0.09	6.95	0.61	8.11	51.70
Av. Diss.:80.95	**Mackerel**	**Sardines**				
Pollachius pollachius	0.32	1.05	15.93	1.21	19.68	19.68
Scyliorhinus canicula	0.41	1.00	9.03	0.83	11.15	30.83
Gobiidae	0.21	0.00	7.81	0.56	9.65	40.47
Homarus gammarus	0.08	0.00	5.91	0.54	7.30	47.77
Labridae	0.11	0.00	5.85	0.43	7.23	55.00
Av. Diss.:91.75	**Mackerel**	**OFM**				
Unknown	0.06	0.93	7.28	1.11	7.93	7.93
Atherina presbyter	0.22	0.70	7.05	0.74	7.69	15.62
Gobiusculus flavescens	0.49	0.16	6.65	0.81	7.74	22.86
Scyliorhinus canicula	0.41	0.13	5.20	0.78	5.67	28.54
Pollachius pollachius	0.32	0.31	4.77	0.87	5.20	33.74
Av. Diss.: 94.34	**Mackerel**	**Prawn**				
Merlangius merlangius	0.28	0.75	15.62	1.36	16.56	16.56
Callionymus lyra	0.03	0.47	9.39	1.54	9.95	26.51
Pomatoschistus microps	0.02	0.44	8.43	1.49	8.93	35.44
Belone belone	0.01	0.00	6.44	0.27	6.83	42.26
Scomber scombrus	0.00	0.42	6.29	0.55	6.67	48.93
Av. Diss.: 90.92	**Mackerel**	**Crab**				
Paguridae	0.16	1.01	16.67	0.97	18.33	18.33
Gobiidae	0.21	0.23	13.27	0.58	14.59	32.93
Scyliorhinus canicula	0.41	0.08	8.97	0.68	9.87	42.79
Merlangius merlangius	0.28	0.00	6.87	0.59	7.56	50.35
Mustelus mustelus	0.03	0.08	4.29	0.41	4.72	60.59
Av. Diss.: 92.54	**Squid**	**No Bait**				
Paguridae	0.44	0.12	22.73	0.82	24.56	24.56
Mustelus mustelus	0.33	0.00	18.44	0.75	19.93	44.49
Scyliorhinus canicula	0.45	0.09	11.49	0.54	12.42	56.91
Unknown	0.06	0.12	8.01	0.46	8.65	65.56
Pollachius pollachius	0.23	0.12	6.75	0.29	7.30	72.86
Av. Diss.: 49.28	**Squid**	**Sardines**				
Pollachius pollachius	0.23	1.05	17.00	1.21	34.50	34.50
Trisopterus luscus	0.18	0.33	12.34	1.13	25.05	59.55
Trisopterus minutus	0.09	0.00	5.71	0.48	11.58	71.13
Av. Diss.: 91.97	**Squid**	**OFM**				
Scyliorhinus canicula	0.45	0.19	11.47	1.43	12.47	12.47
Pomatoschistus microps	0.13	0.00	7.07	0.92	7.69	20.16
Merlangius merlangius	0.09	0.45	6.79	0.74	7.39	27.55
Unknown	0.06	0.93	6.73	0.98	7.32	34.86
Spinachia spinachia	0.00	0.54	5.13	0.69	5.58	40.44

Av. Diss.: 91.60	**Squid**	**Prawn**				
Pollachius pollachius	0.23	0.00	39.02	0.81	42.60	42.60
Trisopterus luscus	0.18	0.00	19.51	0.50	21.30	63.90
Labrus bergylta	0.00	0.13	17.78	0.49	19.41	83.32
Av. Diss.: 83.88	**Squid**	**Crab**				
Paguridae	0.44	1.01	26.92	1.05	32.09	32.09
Mustelus mustelus	0.33	0.08	16.91	0.81	20.16	52.26
Scyliorhinus canicula	0.45	0.08	9.37	0.53	11.17	63.42
Gobiidae	0.13	0.23	5.66	0.66	6.75	70.17
Av. Diss.: 98.28	**OFM**	**No Bait**				
Pomatoschistus microps	0.00	0.14	11.68	0.86	11.88	11.88
Unknown	0.93	0.12	8.71	1.09	8.86	20.75
Belone belone	0.00	0.10	8.33	0.96	8.47	29.22
Spinachia spinachia	0.54	0.00	6.68	0.58	6.80	36.02
Scyliorhinus canicula	0.19	0.09	5.96	0.78	6.06	42.08
Av. Diss.: 87.01	**OFM**	**Sardines**				
Pollachius pollachius	0.31	1.05	15.87	1.02	18.23	18.23
Scyliorhinus canicula	0.19	1.00	15.40	0.95	17.70	35.94
Spinachia spinachia	0.54	0.00	15.40	0.95	17.70	53.64
Labridae	0.20	0.00	8.39	1.30	9.65	63.29
Gobiusculus flavescens	0.16	0.00	6.16	0.66	7.08	70.37
Av. Diss.: 92.44	**OFM**	**Prawn**				
Merlangius merlangius	0.45	0.75	12.24	1.91	13.24	13.24
Pomatoschistus microps	0.00	0.44	8.94	2.79	9.67	22.91
Callionymus lyra	0.10	0.47	8.90	2.11	9.63	32.54
Scomber scombrus	0.00	0.42	7.93	0.68	8.58	41.12
Pleuronectes platessa	0.00	0.33	6.66	1.27	7.20	48.32
Av. Diss.: 68.12	**Sardines**	**No Bait**				
Scyliorhinus canicula	1.00	0.09	25.65	1.03	37.66	37.66
Pollachius pollachius	1.05	0.12	21.30	1.17	31.27	68.93
Trisopterus luscus	0.33	0.10	14.20	0.93	20.84	89.77
Av. Diss.: 87.62	**Prawn**	**No Bait**				
Labrus bergylta	0.13	0.00	40.00	0.80	45.65	45.65
Merlangius merlangius	0.75	0.00	12.00	1.12	13.69	59.35
Callionymus lyra	0.47	0.00	7.90	1.07	9.02	68.37
Scomber scombrus	0.42	0.00	6.34	0.49	7.24	75.61
Av. Diss.: 93.17	**Crab**	**No Bait**				
Paguridae	1.01	0.12	39.84	1.18	42.76	42.76
Gadidae	0.08	0.07	13.60	0.46	14.60	57.36
Unknown	0.08	0.12	10.56	0.50	11.34	63.42
Sepia officianalis	0.08	0.00	4.38	0.33	4.71	70.17

Habitat Observed x Image Quality

Av. Diss.: 91.66	**MCS**	**Sand**				
Scyliorhinus canicula	0.60	0.34	10.59	0.75	11.55	11.55
Pollachius pollachius	0.62	0.05	9.86	0.66	10.76	22.31
Paguridae	0.10	0.53	7.47	0.60	8.15	30.46
Gobiidae	0.32	0.39	6.38	0.56	6.97	37.42
Homarus gammarus	0.20	0.06	5.38	0.40	5.87	43.30
Av. Diss.: 88.70	**MCS**	**Seagrass**				
Gobiusculus flavescens	0.15	0.81	7.00	0.77	7.89	7.89
Pollachius pollachius	0.62	0.42	6.82	0.76	7.69	15.58
Atherina presbyter	0.08	0.70	5.89	0.52	6.64	22.22
Scyliorhinus canicula	0.60	0.36	5.82	0.77	6.56	28.78
Labridae	0.28	0.33	4.56	0.58	5.14	33.92
Av. Diss.: 85.97	**MCS**	**Kelp**				
Ammodytidae	0.09	1.44	8.79	0.58	10.22	10.22

Pollachius pollachius	0.62	0.28	7.86	0.82	9.14	19.36
Scyliorhinus canicula	0.60	0.46	7.47	0.90	8.69	28.05
Unknown	0.05	0.64	7.38	0.99	8.58	36.63
Atherina presbyter	0.08	0.53	6.40	0.68	7.45	44.07
Av. Diss.: 92.13	**MCS**	**Mussel Beds**				
Trisopterus minutus	0.14	1.41	11.55	4.50	12.54	12.54
Scyliorhinidae	0.08	1.21	9.69	2.81	10.52	23.06
Merlangius merlangius	0.17	1.21	8.85	2.37	9.60	32.66
Octopodidae	0.00	1.00	8.24	5.15	8.95	41.61
Paguridae	0.10	1.00	7.87	3.48	8.55	50.15
Av. Diss.: 90.87	**MCS**	**Art. Reef**				
Gadidae	0.21	0.00	29.36	5.25	32.31	32.31
Spondyliosoma cantharus	0.00	287	12.55	1.36	13.81	46.12
Merlangius merlangius	0.17	0.00	12.15	5.25	13.38	59.50
Gobius niger	0.04	0.00	7.02	5.25	7.72	67.22
Paguridae	0.10	0.25	6.48	3.06	7.14	74.35
Av. Diss.: 93.99	**MCS**	**Rocky Reef**				
Gadidae	0.21	0.00	24.28	7.05	24.28	24.28
Ctenolabrus rupestris	0.05	2.89	11.62	4.51	11.62	35.90
Merlangius merlangius	0.17	0.00	10.05	7.05	10.05	45.96
Trisopterus luscus	0.24	2.33	9.36	2.71	9.36	5.32
Gobius niger	0.04	0.00	5.80	7.05	5.80	61.12
Av. Diss.: 94.61	**Sand**	**Seagrass**				
Gobiusculus flavescens	0.00	0.81	6.69	0.68	7.07	7.07
Paguridae	0.53	0.08	6.27	0.61	6.63	13.70
Atherina presbyter	0.03	0.70	6.14	0.51	6.49	20.19
Unknown	0.18	0.36	5.90	0.67	6.23	26.42
Merlangius merlangius	0.30	0.42	5.60	0.64	5.92	32.34
Av. Diss.: 91.96	**Sand**	**Kelp**				
Unknown	0.18	0.64	9.11	1.06	9.90	9.90
Ammodytidae	0.00	1.44	8.46	0.56	9.20	9.20
Scyliorhinus canicula	0.34	0.46	8.33	0.74	9.06	28.16
Atherina presbyter	0.03	0.53	7.66	0.70	8.33	36.50
Scyliorhinidae	0.12	0.36	6.43	0.62	7.00	43.49
Av. Diss.: 85.99	**Sand**	**Mussel Beds**				
Trisopterus minutus	0.02	1.41	12.64	4.78	14.70	14.70
Scyliorhinidae	0.12	1.21	10.66	2.66	12.40	27.09
Merlangius merlangius	0.30	1.21	9.45	2.15	10.99	38.09
Octopodidae	0.00	1.00	8.97	5.08	10.43	48.51
Unknown	0.18	1.00	8.24	2.71	9.59	58.10
Av. Diss.: 88.14	**Seagrass**	**Kelp**				
Ammodytidae	0.25	1.44	10.04	0.57	11.39	11.39
Atherina presbyter	0.70	0.53	6.61	0.60	7.50	18.90
Gobiusculus flavescens	0.81	0.07	6.11	0.77	6.93	25.83
Scyliorhinus canicula	0.36	0.46	4.63	0.77	5.25	31.08
Unknown	0.36	0.64	4.58	0.74	5.19	36.27
Av. Diss.: 86.57	**Seagrass**	**Mussel Beds**				
Trisopterus minutus	0.10	1.41	9.32	3.12	10.76	10.76
Merlangius merlangius	0.42	1.21	7.12	1.59	8.22	18.98
Scyliorhinidae	0.18	1.21	7.02	2.26	8.11	27.09

Octopodidae	0.00	1.00	6.88	4.07	7.95	35.04
Paguridae	0.08	1.00	6.56	2.90	7.58	42.62
Av. Diss.: 93.99	**Seagrass**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	14.78	1.37	15.72	15.72
Centrolabrus exoletus	0.06	0.00	8.50	4.32	9.04	24.76
Scyliorhinus canicula	0.36	0.00	8.50	4.32	9.04	33.80
Maja squinado	0.14	1.16	6.75	1.36	7.18	40.99
Gobiusculus flavescens	0.81	0.90	6.04	1.71	6.42	47.41
Av. Diss.: 85.65	**Seagrass**	**Rocky Reef**				
Ctenolabrus rupestris	0.00	2.89	13.48	4.59	15.74	15.74
Centrolabrus exoletus	0.06	0.00	6.76	6.05	7.89	23.63
Scyliorhinus canicula	0.36	0.00	6.76	6.05	7.89	31.52
Trisopterus luscus	0.25	2.33	6.08	1.48	7.09	38.61
Spondyliosoma cantharus	0.00	1.46	6.03	0.87	7.04	45.65
Av. Diss.: 84.04	**Kelp**	**Mussel Beds**				
Trisopterus minutus	0.00	1.41	10.14	4.84	12.07	12.07
Merlangius merlangius	0.00	1.21	8.55	3.53	10.17	22.24
Ammodytidae	1.44	0.00	7.33	0.51	8.73	30.96
Octopodidae	0.00	1.00	7.17	4.84	8.53	39.49
Paguridae	0.00	1.00	7.17	4.84	8.53	48.02
Av. Diss.: 88.00	**Art. Reef**	**Rocky Reef**				
Ctenolabrus rupestris	0.00	2.89	17.88	2.95	20.32	20.32
Spondyliosoma cantharus	2.87	1.46	14.47	1.32	16.44	36.76
Trisopterus luscus	0.04	2.33	14.17	2.13	16.10	52.87
Parablennius gattorugine	0.00	1.24	7.72	2.01	8.77	61.64
Maja squinado	1.16	0.00	7.18	1.28	8.16	69.80
Av. Diss.: 87.76	**Poor**	**Good**				
Gobiidae	0.19	0.33	10.81	0.50	12.32	12.32
Paguridae	0.30	0.27	9.91	0.56	11.29	23.61
Scyliorhinus canicula	0.39	0.38	6.22	0.73	7.09	30.70
Unknown	0.36	0.12	5.05	0.54	5.76	36.45
Merlangius merlangius	0.33	0.25	4.94	0.56	5.63	42.09
Av. Diss.: 89.50	**Poor**	**Excellent**				
Gadidae	0.13	0.11	16.41	1.01	18.33	18.33
Merlangius merlangius	0.33	0.05	7.47	1.10	8.35	26.68
Scyliorhinus canicula	0.39	0.05	5.30	1.51	5.92	32.60
Centrolabrus exoletus	0.00	0.03	4.46	1.06	4.98	37.59
Paguridae	0.30	0.15	4.10	1.00	4.58	42.16
Av. Diss.: 73.44	**Good**	**Excellent**				
Gadidae	0.16	0.11	8.36	0.64	11.38	11.38
Centrolabrus exoletus	0.03	0.03	5.97	1.54	8.13	19.50
Scyliorhinus canicula	0.38	0.05	5.06	1.32	6.88	26.39
Merlangius merlangius	0.25	0.05	4.47	0.83	6.09	32.48
Chelidonichthys lucerna	0.00	0.02	4.43	1.64	6.03	38.50

Habitat Observed x Time of Day

Av. Diss.: 92.59	**MCS**	**Sand**				
Pollachius pollachius	0.62	0.04	10.42	0.64	11.25	11.25
Scyliorhinus canicula	0.60	0.28	10.11	0.76	10.92	22.17
Gobiidae	0.32	0.31	7.99	0.55	8.63	30.79
Paguridae	0.10	0.42	7.13	0.58	7.70	38.50
Homarus gammarus	0.20	0.05	4.55	0.38	4.91	43.41
Av. Diss.: 91.21	**MCS**	**Seagrass**				
Gobiusculus flavescens	0.15	0.77	9.33	0.86	10.23	10.23
Atherina presbyter	0.08	0.83	9.06	0.55	9.93	20.17
Ammodytidae	0.09	0.66	7.03	0.43	7.70	27.87
Pollachius pollachius	0.62	0.40	6.88	0.78	7.55	35.42
Scyliorhinus canicula	0.60	0.29	6.08	0.84	6.67	42.08
Av. Diss.: 86.84	**MCS**	**Kelp**				
Ammodytidae	0.09	1.44	10.69	0.52	12.31	12.31
Scyliorhinus canicula	0.60	0.46	7.24	0.87	8.33	20.64
Pollachius pollachius	0.62	0.28	7.19	0.73	8.28	28.92
Maja squinado	0.19	0.46	6.53	0.99	7.52	35.45
Scyliorhinidae	0.08	0.36	5.59	0.60	6.44	42.88
Av. Diss.: 92.86	**MCS**	**Mussel Beds**				
Trisopterus minutus	0.14	1.41	11.42	4.74	12.29	12.29
Merlangius merlangius	0.17	1.21	9.81	3.21	10.57	22.86
Octopodidae	0.00	1.00	8.07	4.74	8.69	31.56
Paguridae	0.10	1.00	8.07	4.74	8.69	40.25
Scyliorhinidae	0.08	1.21	7.85	1.71	8.46	48.71
Av. Diss.: 96.69	**MCS**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	23.63	1.29	24.44	24.44
Maja squinado	0.19	1.16	12.22	0.96	12.64	37.08
Gobiusculus flavescens	0.15	0.90	7.60	0.74	7.87	44.95
Pollachius pollachius	0.62	0.00	6.49	0.60	6.71	51.66
Scyliorhinus canicula	0.60	0.00	6.21	0.79	6.42	58.08
Av. Diss.: 95.63	**MCS**	**Rocky Reef**				
Ctenolabrus rupestris	0.05	2.89	19.76	3.11	20.66	20.66
Trisopterus luscus	0.24	2.33	14.86	1.96	15.54	36.20
Parablennius gattorugine	0.00	1.24	8.74	2.08	9.14	45.34
Spondyliosoma cantharus	0.00	1.46	8.43	0.88	8.82	54.15
Symphodus melops	0.00	0.71	4.40	1.13	4.60	58.75
Av. Diss.: 96.97	**Sand**	**Seagrass**				
Gobiusculus flavescens	0.00	0.77	10.59	0.82	10.92	10.92
Atherina presbyter	0.12	0.83	10.42	0.53	10.74	21.67
Ammodytidae	0.04	0.66	7.87	0.41	8.11	29.78
Paguridae	0.42	0.06	4.89	0.54	5.04	34.82
Scyliorhinus canicula	0.28	0.29	4.59	0.66	4.73	39.55
Av. Diss.: 92.81	**Sand**	**Kelp**				
Ammodytidae	0.04	1.44	11.26	0.49	12.14	12.14
Scyliorhinus canicula	0.28	0.46	9.70	0.81	10.46	22.59
Scyliorhinidae	0.10	0.36	7.55	0.60	8.13	30.72
Maja squinado	0.10	0.46	7.40	0.99	7.97	38.69
Unknown	0.14	0.64	6.43	0.68	6.93	45.62
Av. Diss.: 99.40	**Sand**	**Mussel Beds**				
Trisopterus minutus	0.01	1.41	14.09	5.70	14.18	14.18

Merlangius merlangius	0.24	1.21	12.13	3.51	12.21	26.38
Scyliorhinidae	0.10	1.21	11.92	4.83	11.99	38.38
Octopodidae	0.00	1.00	9.96	5.70	10.02	48.40
Paguridae	0.42	1.00	9.96	5.70	10.02	58.43
Av. Diss.:96.99	**Sand**	**Art. Reef**				
Spondyliosoma cantharus	0.03	2.87	27.42	1.30	28.27	28.27
Maja squinado	0.10	1.16	16.15	0.86	16.65	44.92
Gobiusculus flavescens	0.00	0.90	8.33	0.72	8.59	53.50
Paguridae	0.42	0.25	7.10	0.60	7.32	60.82
Symphodus melops	0.00	0.37	4.28	0.54	4.42	65.24
Av. Diss.: 99.11	**Sand**	**Rocky Reef**				
Ctenolabrus rupestris	0.01	2.89	22.68	3.22	22.88	22.88
Trisopterus luscus	0.08	2.33	18.41	2.35	18.57	41.46
Parablennius gattorugine	0.00	1.24	9.88	1.99	9.97	51.52
Spondyliosoma cantharus	0.03	1.46	9.31	0.89	9.40	60.82
Symphodus melops	0.00	0.71	4.88	1.13	4.93	65.75
Av. Diss.: 89.68	**Seagrass**	**Kelp**				
Ammodytidae	0.66	1.44	12.73	0.63	14.19	14.19
Atherina presbyter	0.83	0.53	7.62	0.52	8.49	22.68
Gobiusculus flavescens	0.77	0.07	7.61	0.81	8.49	22.68
Scyliorhinus canicula	0.29	0.46	5.55	0.81	6.19	37.36
Maja squinado	0.12	0.46	4.69	0.95	5.23	42.59
Av. Diss.: 82.42	**Seagrass**	**Mussel Beds**				
Trisopterus minutus	0.08	1.41	8.95	2.45	10.86	10.86
Scyliorhinidae	0.14	1.21	7.46	1.97	9.05	19.92
Merlangius merlangius	0.33	1.21	7.00	1.61	8.50	28.41
Octopodidae	0.00	1.00	6.93	3.39	8.41	36.83
Paguridae	0.06	1.00	6.13	1.98	7.43	44.26
Av. Diss.: 93.95	**Seagrass**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	19.89	1.30	21.18	21.18
Gobiusculus flavescens	0.77	0.90	9.73	0.98	10.36	31.54
Maja squinado	0.12	1.16	9.64	1.01	10.26	41.80
Atherina presbyter	0.83	0.00	7.46	0.49	7.94	49.74
Ammodytidae	0.66	0.00	5.84	0.40	6.21	55.96
Av. Diss.: 96.25	**Seagrass**	**Rocky Reef**				
Ctenolabrus rupestris	0.02	2.89	17.23	3.28	17.90	17.90
Trisopterus luscus	0.08	2.33	18.41	2.35	18.57	41.46
Parablennius gattorugine	0.00	1.24	7.53	2.21	7.83	39.57
Spondyliosoma cantharus	0.00	1.46	7.48	0.87	7.78	47.34
Gobiusculus flavescens	0.77	0.04	5.84	0.86	6.07	53.41
Av. Diss.: 83.94	**Kelp**	**Mussel Beds**				
Labridae	0.28	0.00	13.91	17.64	16.57	16.57
Halichoerus grypus	0.09	0.00	11.35	17.64	13.53	30.09
Trisopterus minutus	0.00	1.41	11.35	17.64	13.53	43.62
Merlangius merlangius	0.00	1.21	9.76	3.36	11.63	55.25

Scyliorhinidae	0.36	1.21	9.63	5.34	11.47	66.71
Av. Diss.:90.35	**Kelp**	**Art. Reef**				
Spondyliosoma cantharus	0.00	2.87	21.10	1.30	23.36	23.36
Ammodytidae	1.44	0.00	9.30	0.49	10.30	33.65
Maja squinado	0.46	1.16	8.45	0.93	9.35	43.01
Scyliorhinus canicula	0.46	0.00	6.50	0.78	7.19	50.20
Gobiusculus flavescens	0.07	0.90	6.45	0.71	7.13	57.34
Av. Diss.: 97.87	**Kelp**	**Rocky Reef**				
Ctenolabrus rupestris	0.07	2.89	17.94	2.92	18.33	18.33
Trisopterus luscus	0.13	2.33	14.79	2.37	15.11	33.43
Parablennius gattorugine	0.00	1.24	7.93	2.15	8.10	41.54
Spondyliosoma cantharus	0.00	1.46	7.80	0.87	7.97	49.50
Ammodytidae	1.44	0.00	7.60	0.49	7.77	57.27
Av. Diss.: 88.00	**Art. Reef**	**Rocky Reef**				
Ctenolabrus rupestris	0.00	2.89	17.88	2.95	20.95	20.32
Spondyliosoma cantharus	2.87	1.46	14.47	1.32	16.44	36.76
Trisopterus luscus	0.04	2.33	14.17	2.13	16.10	52.87
Parablennius gattorugine	0.00	1.24	7.72	2.01	8.77	61.64
Maja squinado	1.16	0.00	7.18	1.28	8.16	69.80
Av. Diss.: 93.49	**Day**	**Evening**				
Atherina presbyter	0.22	0.62	14.40	0.66	15.40	15.40
Ammodytidae	0.23	0.57	6.73	0.37	7.19	22.59
Paguridae	0.25	0.17	6.21	0.43	6.64	29.23
Gobiidae	0.19	0.15	5.91	0.38	6.32	35.56
Scyliorhinus canicula	0.29	0.17	4.90	0.49	5.25	40.80
Av. Diss.: 79.69	**Day**	**Dawn**				
Atherina presbyter	0.22	0.17	6.36	0.55	7.99	7.99
Gobiusculus flavescens	0.31	0.92	6.03	0.93	7.57	15.56
Merlangius merlangius	0.16	0.82	5.63	1.10	7.06	22.62
Gobiidae	0.19	0.06	5.61	0.31	7.04	29.66
Scyliorhinus canicula	0.29	0.69	5.16	0.92	6.48	36.14
Av. Diss.: 87.64	**Day**	**Night**				
Atherina presbyter	0.22	1.44	10.59	1.05	12.08	12.08
Ammodytidae	0.23	0.75	8.20	0.81	9.35	21.44
Unknown	0.11	1.06	7.05	1.72	8.04	29.48
Merlangius merlangius	0.16	0.63	6.16	0.63	7.03	36.51
Gobiusculus flavescens	0.31	0.23	5.66	0.77	6.46	42.97
Av. Diss.: 86.61	**Evening**	**Dawn**				
Gobiusculus flavescens	0.13	0.92	6.08	1.13	7.02	7.02
Atherina presbyter	0.62	0.17	5.93	0.64	6.84	13.86
Trisopterus luscus	0.07	0.74	5.26	1.21	6.08	19.94
Merlangius merlangius	0.19	0.82	5.20	1.17	6.00	25.94
Scyliorhinus canicula	0.17	0.69	5.16	1.23	5.96	31.89
Av. Diss.: 79.51	**Evening**	**Night**				
Atherina presbyter	0.62	1.44	8.15	1.12	10.25	10.25
Ammodytidae	0.57	0.75	7.01	0.82	8.82	19.07
Merlangius merlangius	0.19	0.63	7.00	0.70	8.81	27.87

Unknown	0.40	1.06	6.18	1.10	7.77	35.64
Gadidae	0.27	0.53	5.54	0.80	6.97	42.62
Av. Diss.: 85.15	**Night**	**Dawn**				
Merlangius merlangius	0.82	0.63	7.80	1.26	9.16	9.16
Atherina presbyter	0.17	1.44	7.16	1.21	8.41	17.57
Unknown	0.00	1.06	7.01	3.03	8.23	25.81
Gobiusculus flavescens	0.92	0.23	5.32	1.18	6.25	32.05
Gadidae	0.15	0.53	4.91	0.89	5.77	37.82
Habitat Observed x Spring / Neap Tide						
Av. Diss.: 89.82	**MCS**	**Sand**				
Scyliorhinus canicula	0.60	0.34	10.31	0.71	11.48	11.48
Pollachius pollachius	0.62	0.05	9.83	0.61	10.94	22.42
Paguridae	0.10	0.53	8.52	0.66	9.49	31.91
Gobiidae	0.32	0.39	8.41	0.61	9.37	41.27
Labridae	0.28	0.00	4.90	0.38	5.45	46.72
Av. Diss.: 89.59	**MCS**	**Seagrass**				
Scyliorhinus canicula	0.60	0.40	6.77	0.85	7.56	7.56
Atherina presbyter	0.08	0.62	6.58	0.57	7.34	14.90
Gobiusculus flavescens	0.15	0.72	6.35	0.76	7.09	21.98
Pollachius pollachius	0.62	0.40	6.06	0.70	6.76	28.74
Merlangius merlangius	0.17	0.47	5.03	0.66	5.62	34.36
Av. Diss.: 84.70	**MCS**	**Kelp**				
Unknown	0.05	0.64	7.64	1.02	9.02	9.02
Ammodytidae	0.09	1.44	7.55	0.57	8.92	17.94
Atherina presbyter	0.08	0.53	7.39	0.75	8.73	26.67
Scyliorhinus canicula	0.60	0.46	7.26	0.90	8.57	35.24
Pollachius pollachius	0.62	0.28	7.17	0.77	8.47	43.71
Av. Diss.: 95.19	**MCS**	**Mussel Beds**				
Trisopterus minutus	0.14	1.41	12.70	5.60	13.34	13.34
Merlangius merlangius	0.17	1.21	10.93	3.53	11.48	24.83
Scyliorhinidae	0.08	1.21	10.27	3.13	10.79	35.62
Octopodidae	0.00	1.00	8.98	5.60	9.44	45.05
Paguridae	0.10	1.00	8.98	5.60	9.44	54.49
Av. Diss.: 97.36	**MCS**	**Midwater**				
Pollachius pollachius	0.62	0.10	16.23	0.72	16.67	16.67
Scyliorhinus canicula	0.60	0.22	15.58	0.74	16.00	32.67
Gobiidae	0.32	0.00	8.09	0.47	8.31	40.98
Labridae	0.28	0.00	7.30	0.39	7.50	48.48
Trisopterus luscus	0.24	0.11	6.03	0.46	6.20	54.67
Av. Diss.: 93.06	**Sand**	**Seagrass**				
Gobiusculus flavescens	0.00	0.72	6.62	0.70	7.11	7.11
Atherina presbyter	0.03	0.62	6.58	0.48	7.07	14.19
Scyliorhinus canicula	0.34	0.40	5.75	0.74	6.18	20.37
Merlangius merlangius	0.30	0.47	5.73	0.69	6.16	26.53
Paguridae	0.53	0.06	5.05	0.47	5.42	31.95
Av. Diss.: 93.30	**Sand**	**Kelp**				
Scyliorhinus canicula	0.34	0.46	9.30	0.83	9.96	9.96
Scyliorhinidae	0.12	0.36	9.13	0.78	9.78	19.75
Ammodytidae	0.00	1.44	8.90	0.52	9.54	29.20
Unknown	0.18	0.64	8.03	0.88	8.61	37.89
Paguridae	0.53	0.00	6.36	0.65	6.82	44.71
Av. Diss.: 87.07	**Sand**	**Mussel Beds**				
Trisopterus minutus	0.02	1.41	13.83	4.83	15.88	15.88
Scyliorhinidae	0.12	1.21	11.70	4.28	13.44	29.33

Merlangius merlangius	0.30	1.21	11.64	2.87	13.37	42.69
Octopodidae	0.00	1.00	9.78	4.83	11.23	53.92
Unknown	0.18	1.00	8.45	2.08	9.70	63.63
Av. Diss.: 99.07	**Sand**	**Midwater**				
Paguridae	0.53	0.00	16.50	0.69	16.65	16.65
Gobiidae	0.39	0.00	10.89	0.48	10.99	27.65
Mustelus mustelus	0.18	0.00	9.95	0.44	10.04	37.69
Scyliorhinus canicula	0.34	0.22	8.77	0.55	8.85	46.54
Merlangius merlangius	0.30	0.34	7.46	0.54	7.53	54.07
Av. Diss.: 93.06	**Seagrass**	**Kelp**				
Ammodytidae	0.29	1.44	14.74	0.74	15.83	15.83
Atherina presbyter	0.62	0.53	5.93	0.58	6.37	22.20
Pomatoschistus minutus	0.02	0.43	5.83	1.04	6.26	28.46
Maja squinado	0.13	0.46	5.37	1.08	5.77	34.23
Gobiusculus flavescens	0.72	0.07	5.17	0.74	5.55	39.78
Av. Diss.: 95.30	**Seagrass**	**Mussel Beds**				
Trisopterus minutus	0.08	1.41	0.41	4.04	10.93	10.93
Scyliorhinidae	0.15	1.21	8.83	3.59	9.26	20.19
Merlangius merlangius	0.47	1.21	8.44	2.30	8.86	29.05
Octopodidae	0.00	1.00	7.36	4.04	7.73	36.78
Paguridae	0.06	1.00	7.36	4.04	7.73	44.51
Av. Diss.: 95.72	**Seagrass**	**Midwater**				
Atherina presbyter	0.62	0.00	7.86	0.43	8.21	8.21
Gobiusculus flavescens	0.72	0.04	6.97	0.63	7.28	15.49
Merlangius merlangius	0.47	0.34	6.76	0.59	7.06	22.56
Scyliorhinus canicula	0.40	0.22	5.90	0.57	6.16	28.72
Pomatoschistus microps	0.15	0.11	5.11	0.33	5.34	34.06
Av. Diss.: 81.82	**Kelp**	**Mussel Beds**				
Trisopterus minutus	0.00	1.41	9.78	7.05	11.96	11.96
Merlangius merlangius	0.00	1.21	8.40	3.91	10.27	22.22
Atherina presbyter	0.53	0.00	7.32	1.09	8.94	31.16
Octopodidae	0.00	1.00	6.92	7.05	8.45	39.62
Paguridae	0.00	1.00	6.92	7.05	8.45	48.07
Av. Diss.: 97.95	**Kelp**	**Midwater**				
Scyliorhinus canicula	0.46	0.22	13.77	0.86	14.06	14.06
Scyliorhinidae	0.36	0.00	13.18	0.77	13.45	27.51
Unknown	0.64	0.00	11.42	1.03	11.66	39.17
Atherina presbyter	0.53	0.00	7.89	0.63	8.06	47.23
Ammodytidae	1.44	0.00	7.32	0.50	7.47	54.71
Av. Diss.: 95.40	**Mussel Beds**	**Midwater**				
Trisopterus minutus	1.41	0.03	14.48	3.57	15.18	15.18
Merlangius merlangius	1.21	0.34	13.24	3.52	13.88	29.05
Scyliorhinidae	1.21	0.00	12.24	3.35	12.83	41.89
Octopodidae	1.00	0.00	10.24	3.57	10.73	52.62
Paguridae	1.00	0.00	10.24	3.57	10.73	63.35
Av. Diss.: 88.96	**Spring**	**Neap**				
Paguridae	0.29	0.34	13.20	0.70	14.83	4.83
Gobiidae	0.08	0.42	9.28	0.54	10.43	25.26
Scyliorhinus canicula	0.49	0.25	8.90	0.65	10.00	35.26
Mustelus mustelus	0.05	0.15	7.43	0.43	8.35	43.62
Unknown	0.26	0.18	6.58	0.50	7.39	51.01

Av. Diss.: 90.08	**Spring**	**Mid**				
Scyliorhinus canicula	0.49	0.42	7.89	0.75	8.76	8.76
Merlangius merlangius	0.31	0.42	6.89	0.75	7.64	16.40
Paguridae	0.29	0.12	6.23	0.41	6.92	23.32
Atherina presbyter	0.14	0.46	4.96	0.46	5.51	28.83
Gobiidae	0.08	0.13	4.64	0.29	5.15	33.98
Av. Diss.: 90.36	**Neap**	**Mid**				
Gobiidae	0.42	0.13	9.21	0.51	10.19	10.19
Paguridae	0.34	0.12	8.23	0.53	9.11	19.30
Merlangius merlangius	0.24	0.42	7.07	0.73	7.82	27.13
Scyliorhinus canicula	0.25	0.42	6.91	0.73	7.65	34.78
Scyliorhinidae	0.15	0.16	5.93	0.70	6.56	41.34

Chapter 3

The influence of bait on remote underwater video observations in shallow-water coastal environments associated with the North- Eastern Atlantic

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Three Cliffs Bay, 2018

Author Contribution

REJ led the design of the experimental procedures, experimental set up, deployment locations, fieldwork and data collection; REJ analysed the data and drafted the manuscript. RAG, SJH and RKFU provided advice on the survey design, presentation of results, reviewed and commented on the manuscript with all authors giving final approval for submission.

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Steph Januchowski-Hartley	Swansea University	5
Richard Unsworth	Swansea University	5

We the undersigned agree with the above stated "proportion of work undertaken" for each of the above published peer-reviewed manuscripts contributing to this thesis:

Signed Candidate [redacted] ________________________________

Author 1 [redacted] ________________________________

Author 2 ________________________________

Author 3 ________ [redacted] ______

Author 4 ________ [redacted] ______

Abstract

The use of baited remote underwater video (BRUV) systems for examining and monitoring marine biodiversity in temperate marine environments is rapidly growing, however many aspects of their effectiveness relies on assumptions based on studies from the Southern Hemisphere. The addition of bait to underwater camera systems acts as a stimulus for attracting individuals towards the camera field of view, however knowledge of the effectiveness of different bait types in northern temperate climbs is limited, particularly in dynamic coastal environments. Studies in the Southern Hemisphere indicate that oily baits are most effective whilst bait volume and weight do not impact BRUV effectiveness to any great degree. The present study assesses the influence of four bait types (mackerel, squid, crab and no bait (control)) on the relative abundance, taxonomic diversity and faunal assemblage composition at two independent locations within the North-Eastern Atlantic region; Swansea Bay, UK and Ria Formosa Lagoon, Portugal. Two different bait quantities (50g and 350g) were further trialled in Swansea Bay.

Overall, patterns showed that baited deployments recorded statistically higher values of relative abundance and taxonomic diversity when compared to un-baited deployments in Swansea Bay but not in Ria Formosa Lagoon. No statistical evidence singled out one bait type as best performing for attracting higher abundances and taxonomic diversity in both locations. Faunal assemblage composition was however found to differ with bait type in Swansea Bay, with mackerel and squid attracting higher abundances of scavenging species compared to the crab and control treatments. With the exception of squid, bait quantity had minimal influence on bait attractiveness. It is recommended for consistency that a minimum of 50g of cheap, oily fish such as mackerel is used as bait for BRUV deployments in shallow dynamic coastal environments in the North-Eastern Atlantic Region.

Keywords: Baited remote underwater video; Temperate habitats; Bait type; Bait quantity; Subtidal sediments; Fish assemblages.

3.1. Introduction

Baited remote underwater video (BRUV) can be used as a standardised, non-extractive technique to assess motile fauna, more specifically of fishes and fish assemblages (Cappo et al., 2006). Bait attracts individuals of different species towards the field of view of the recording camera by releasing chemical stimuli including water-soluble proteins into the surrounding water column (Wraith, et al., 2013). The inclusion of bait with underwater cameras has been shown to help with overcoming the problem of low fish counts associated with fish passing un-baited systems by chance (Stobart et al., 2007) and has been utilised in both deep-sea environments (Fleury & Drazen, 2013) and shallow coastal environments (Unsworth et al., 2014).

The type and quantity of bait as well as characteristics of different species and the environment can influence or attract different motile faunal assemblages and can lead to biases in predatory or scavenging species in BRUV surveys (Fleury & Drazen, 2013; Harvey et al., 2007; Yeh & Drazen, 2011). Studies in the southern hemisphere, over coral and rocky reef habitats, found that oily fishes such as those found from the Clupeidae and Scombridae families consistently attracted higher taxonomic diversity and abundances (Dorman et al., 2012; Walsh et al., 2016; Wraith et al., 2013). Equally, the plume emitted can vary depending on the physical characteristics of the bait, such as persistence, quantity, moisture content, soak time, and dispersal area (Dorman et al., 2012). Quantity of bait may also influence faunal abundances; more bait may attract more individuals to the camera (Hardinge et al., 2013). Attraction to BRUVs by different fauna can also be influenced by hunger levels, individual boldness, size of bait plume as well as hydrographic and topographic conditions (Harvey et al., 2007; Taylor et al., 2013). Other considerations include the increased presence of predatory species in the vicinity of the bait potentially altering the behaviour and / or abundance of prey in the presence of bait (Coghlan et al., 2017; Dunlop et al., 2015).

In the North-Eastern Atlantic region, faunal assemblages associated with subtidal sediment habitats have traditionally been sampled using grabs, dredges, towed video cameras (sledge) and trawls (Kaiser et al., 2004). However, such methods can be inappropriate when in close proximity to seabed infrastructure and within or near marine protected areas, because of the methods' destructive and mobile nature (Griffin et al., 2016; Jones et al., 2019). Challenges with sampling these dynamic environments means that many data gaps remain for the motile fauna that inhabit these sediment habitats (Shields et al. 2011). At present, little guidance exists for BRUV deployments in the North Atlantic. With recent methodological improvements in low visibility and dynamic coastal environments (Jones et al., 2019), an opportunity exists to apply these BRUV methods to highly dynamic and regulated systems (Ghazilou et al., 2007) present in the North Atlantic region.

Here, our goal was to establish a method standardisation for BRUV deployments by determining the bait types and quantities that are best suited to shallow coastal environments (<15m) associated with the North-Eastern Atlantic region. This study aims to provide an insight into bait performance and inform BRUV guidelines for future monitoring in the region. We assessed the relative abundance, taxonomic diversity and faunal assemblage composition in relation to the various bait types and quantities in two independent case study areas, Swansea Bay, United Kingdom and Ria Formosa Lagoon, Portugal. For the purpose of this study, quantitative comparisons between these two locations were not made. We hypothesized that large quantities of oily fish treatments would perform best by attracting higher numbers of individuals and species in shallow coastal environments in the North-Eastern Atlantic region based on their performance in other bait studies. We discuss our findings from two independent case studies and provide a recommendation for future BRUV deployments in soft sediment, shallow coastal habitats in this region.

3.2. Methods and Materials

3.2.1. Site Descriptions

Sampling for this study was conducted at two case study locations in the North-Eastern Atlantic region; Swansea Bay, United Kingdom and Ria Formosa Lagoon, Portugal (Fig. 1). Swansea Bay is considered a highly dynamic environment, subject to tidal ranges of 10.5m (Waters & Aggidis, 2016) and large tidal currents. The surveyed habitat type at this study location was subtidal sediment consisting of fine sands with gravel patches which remain homogeneous over a large spatial area. The surveyed area of Ria Formosa Lagoon was also characterised by subtidal soft sediments; however, an increased heterogeneity was present in the wider area with patches of seagrass beds, sandflats and saltmarshes present (Curtis & Vincent, 2005). Intense morphodynamics, strong winds, and tidal ranges up to 3.2m (Ceia et al., 2010) also influence this location. Sampling was conducted in 2018 and 2019; Swansea Bay was sampled in August 2018 and Ria Formosa Lagoon was sampled in May 2019.

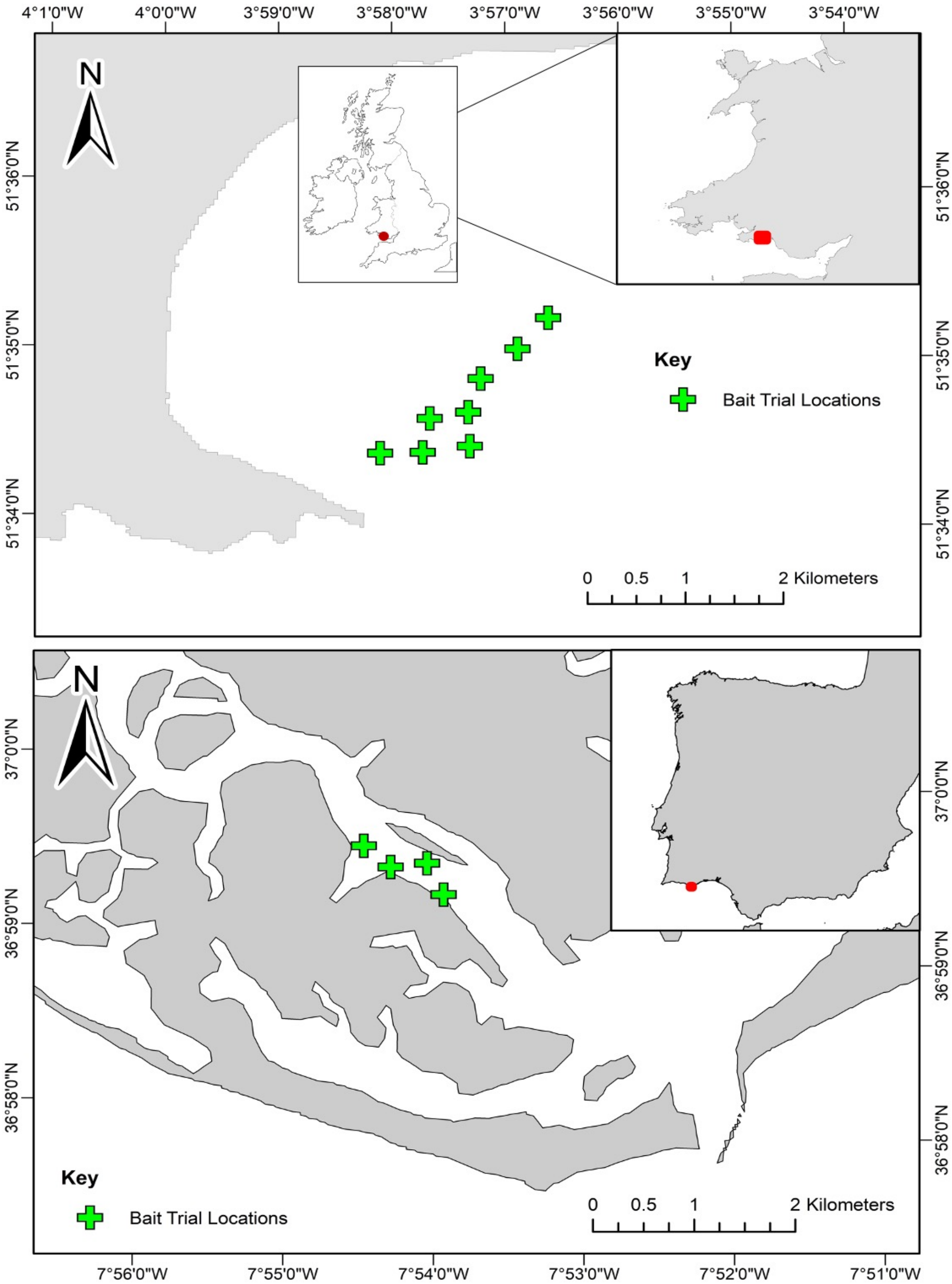

Figure 1: Station locations for the BRUV bait trials in Swansea Bay, United Kingdom (top) and Ria Formosa Lagoon, Portugal (bottom). Stations are positioned a minimum 350m apart.

3.2.2. Experimental Design

In the Swansea Bay case study, we used a two-factor design, considering both bait type (mackerel, squid, crab, and no bait (control)) and weight (350g, 50g, and no bait (control)). In the Ria Formosa Lagoon case study, we used a one-factor design with bait (mackerel, squid, crab and no bait (control)). We were unable to assess differences in bait weights in Ria Formosa Lagoon due to technical difficulties, so we used a single weight of 200g (Fig. 2). The range of bait weights we considered in both Swansea Bay and Ria Formosa case studies are similar to those used in previous studies in the North-Eastern Atlantic Region (Griffin et al., 2016; Peters et al., 2015; Unsworth et al., 2014), but less than those most commonly reported in Australian studies (Whitmarsh et al., 2017). Predation induced bait depletion by high fish abundances have not previously been recorded in these study areas and is unlikely given the composition of species. Therefore, the bait weights used are reasonable given the anticipated types and abundances of species in our two case study areas.

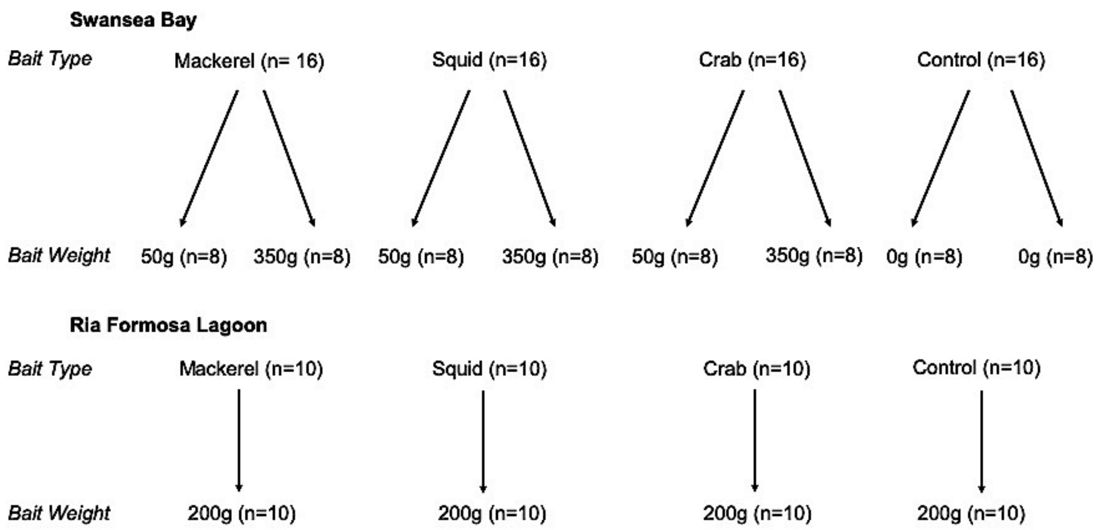

Figure 2: Experimental design of BRUV treatments with replicates in brackets for Swansea Bay and Ria Formosa Lagoon.

We used eight deployment stations in Swansea Bay and four in Ria Formosa Lagoon (Fig. 1). Each station was standardised for depth (3 to 10m) and substrate type (sandy and mixed coarse sediments in Swansea Bay; soft sediments mixed with seagrass in Ria Formosa Lagoon). We deployed our stations during daylight hours (8am to 7pm) allowing an hour between sunrise and sunset to avoid crepuscular variation in assemblages (Myers et al., 2016). Due to the modest bait weights, we determined 350m to be sufficient distance between deployment stations to ensure independence of deployments, avoiding overlap of bait plumes and reducing the likelihood of fish moving between sites during the sampling period based on previous research (Wraith et al., 2013).

In terms of bait types, we wanted to trial species from the following groups; Fish, Crustacean and Mollusca. Such groups have previously been implemented in past bait studies. Atlantic mackerel (*Scomber scombrus*) was used at both in both Swansea Bay and Ria Formosa Lagoon. In Swansea Bay we used Foreign Peeler Crab (*Portunus pelagicus*) and the European Common Squid (*Alloteuthis subulata*); both are commonly used by UK recreational anglers and available at local bait shops. In the Ria Formosa Lagoon, we used similar bait types; these included the Common Shore Crab (*Carcinus maenas*) and the European Squid (*Loligo vulgaris*) which were also widely available in local shops. All bait types were defrosted, chopped into similar sized pieces of approximately 3cm x 3cm and weighed 24 hours prior to sampling and placed into sealed labelled bags to retain contents.

Deployments of each treatment (a bait type and weight in Swansea Bay; a bait quantity of 200g in Ria Formosa Lagoon) were randomly deployed across the eight stations, we ensured no replicate was deployed at the same time. Each treatment was deployed for a period of one hour. A 5mm polyvinyl chloride (PVC) mesh bait bag was used to maximise dispersal, with bait replenished after every deployment.

We retrieved 51 successful deployments from Swansea Bay and 38 from Ria Formosa Lagoon. The following deployments were unsuccessful and not included in subsequent analyses: 4 x mackerel 350g, 1 x mackerel 50g, 2 x crab 50g, 2 x crab 350g, 1 x squid 50g, 1 x squid 350g, 1 x squid 350g and 1 x control. Two mackerel deployments also failed in Ria Formosa Lagoon. Of the failed deployments in Swansea Bay, seven were due to low underwater visibility (bait not visible), two were due to a camera fault and four were due to the BRUV toppling forwards into the sediment during the deployment. The two failed deployments in Ria Formosa were due to low levels of underwater visibility.

3.2.3. Sampling Equipment

The mono-BRUVs used during this study consisted of one Hero 4 GoPro high definition camera (GoPro, San Mateo, CA) in a waterproof housing with a resolution of 1920 x 1080, focal length of 17.2mm and a horizontal field of view of 122.6° (approximately 7.3m widest field of view). This was mounted onto an aluminium frame and weighted with 4kg at the base for stability. A bait pole extended 65cm in front of the camera supporting the 5mm mesh bag containing the bait treatment. Each mono-BRUV system was deployed with a rope attached to a surface buoy to allow for remote deployment and recovery. No artificial light was added to these frames.

3.2.4. Video Analysis

All fish assemblages and motile benthic macro fauna likely to be monitored in coastal habitats using BRUV methods (Jones et al., 2019) were included in this analysis. Raw footage from each BRUV deployment was compressed to Audio Video Interleave format using Xilisoft Video/Media Converter Ultimate (www.uk.xilisoft.com) for the use of the footage in the specialist SeaGIS software Event Measure (www.seagis.com.au). We did not review any deployments where a BRUV had toppled into the sediment restricting field of view, or where the bait bag was not visible because of high levels of turbidity.

We viewed and analysed all footage for maximum number of individuals observed in a single video frame (*MaxN*) over a one-hour deployment. *MaxN* is a measure of relative abundance to avoid repeated counts of individuals (Priede et al., 1994). Taxonomic diversity was calculated from the number of different species entering the camera frame during a one-hour deployment with faunal assemblage composition in each deployment recorded. Where possible, taxa were identified to species level, followed by family level if distinguishable features were not present. Organisms were identified as unknown if turbidity levels affected confidence of identification and not included further in the analysis.

3.2.5. Statistical Analysis

Results for the two locations were analysed as two independent case studies. We conducted all analyses in PRIMER v7 (Clarke & Gorley, 2007). Data were transformed (square root) where appropriate for count data, to reduce variance of heterogeneity.

For Swansea Bay, we assessed both total sample (combined weights under each bait type) and split sample (comparing weights within each bait type). Quantities used for the baited treatments (50g and 350g) differed to those used in the control treatment (0g). A nested design was followed to allow statistical comparisons between the baited treatments and the control. For Swansea Bay, the univariate analysis consisted of a two-factor (bait and weight) permutational multivariate analysis of variance (PERMANOVA+; Anderson, 2017) using a Euclidean resemblance matrix to test for differences in relative abundance and taxonomic diversity between treatments (Table 1). In Ria Formosa Lagoon, a one-factor (bait) PERMANOVA was used.

For the multivariate analysis of faunal assemblage composition in Swansea Bay, a (bait and weight) PERMANOVA using a Bray-Curtis resemblance matrix was used, and a single factor PERMANOVA was used for Ria Formosa Lagoon. Principle coordinates were plotted for both locations in a constrained Canonical Analysis of Principal Coordinates (CAP) to test for differences between groups of significant factors and to visualise patterns in the data that can be hidden in unconstrained Non-metric Multidimensional Scaling plots (Anderson & Willis, 2003; Table 1). A 'leave-one out' cross validation analysis was undertaken to give a statistical measure of the distinctiveness of the groups presented within the CAP plots.

Table 1: Analysis undertaken on the three variables used in the assessment of bait type and bait weight

Variable	Analysis
Swansea Bay	
Relative Abundance (*MaxN*)	Univariate; Two-Factor Nested PERMANOVA, Pairwise Comparison
Taxonomic Diversity	Univariate; Two Factor Nested PERMANOVA, Pairwise Comparison
Faunal Assemblage Composition	Multivariate; Nested PERMANOVA, CAP, SIMPER, PERMDISP
Ria Formosa Lagoon	
Relative Abundance (MaxN)	Univariate; One-Factor PERMANOVA, Pairwise Comparison
Taxonomic Diversity	Univariate; One Factor PERMANOVA, Pairwise Comparison
Faunal Assemblage Composition	Multivariate; Nested PERMANOVA, CAP, SIMPER, PERMDISP

All PERMANOVA tests were based on 9999 unrestricted permutations of the raw data with significant results considered $P<0.05$. Pairwise tests were carried out where appropriate to identify differences between treatments. Where possible, we used an analysis of similarity percentages (SIMPER) to identify the main species recorded on the BRUVs responsible for any differences identified between treatments. A permutational analysis of multivariate dispersions (PERMDISP) was also used to assess differences between bait types and quantities. All means have been reported ±1 Standard Error (SE).

3.3. Results

We identified 130 individuals from 17 taxa recorded in 51 BRUVs in Swansea Bay and 55 individuals from 7 taxa in 38 BRUVs in Ria Formosa Lagoon (Table 2). The greatest number of taxa were recorded using squid and crab bait in Swansea Bay (10) and using mackerel bait in Ria Formosa Lagoon (4) (Table 2). The control (no bait) recorded the lowest number of taxa (3) in Swansea Bay, but squid had the lowest number of taxa in Ria (2) Formosa Lagoon (Table 2).

Table 2: The mean (± 1SE) relative abundance (MaxN) of all taxa sampled during BRUV deployments in Swansea Bay (above) and Ria Formosa Lagoon (below) using different bait treatments (M = Mackerel, S = Squid, C = Crab)

		Mean (±SE) Relative Abundance						
Family	**Species**	**M50g**	**M350g**	**S50g**	**S350g**	**C50g**	**C350g**	**Control (no bait)**
Swansea Bay, South Wales								
Arthropoda								
Paguridae	-	1.71 (±0.87)	0.50 (±0.29)	1.00 (±0.65)	1.14 (±0.51)	2.50 (±0.99)	1.33 (±0.88)	0.29 (±0.19)
Chordata								
Balistidae	*Balistes capriscus*	0.14 (±0.14)	-	-	-	-	-	-
Moronidae	*Dicentrarchus labrax*	-	-	-	0.14 (±0.14)	-	-	-
Gadidae	-	-	-	-	-	-	0.17 (±0.17)	0.21 (±0.15)
Gobiidae	-	-	-	0.14 (±0.14)	0.71 (±0.57)	0.17 (±0.17)	0.50 (±0.50)	-
Majidae	*Maja squinado*	-	-	-	0.14 (±0.14)	-	-	-
Mullidae	*Mullus surmuletus*	-	-	-	-	-	-	0.14 (±0.14)
Pleuronectidae	-	-	-	0.14 (±0.14)		0.17 (±0.17)	-	-
Rajidae	*Raja clavata*	-	-	-	0.14 (±0.14)	-	-	-
Scyliorhinidae	*Scyliorhinus canicula*	0.57 (±0.30)	-	0.14 (±0.14)	0.71 (±0.29)	0.17 (±0.17)	-	-
Sparidae	-	0.29 (±0.18)	-	-	-	-	-	-
Sparidae	*Spondyliosoma cantharus*	0.57 (±0.57)	-	-	-	0.33 (±0.33)	-	-
Triglidae	-	-	0.25 (±0.25)	0.14 (±0.14)	-	-	-	-
Triglidae	*Chelidonichthys lucerna*	0.43 (±0.20)	-	-	0.14 (±0.14)	0.17 (±0.17)	0.33 (±0.33)	-

Table 2: The mean (± 1SE) relative abundance (MaxN) of all taxa sampled during BRUV deployments in Swansea Bay (above) and Ria Formosa Lagoon (below) using different bait treatments (M = Mackerel, S = Squid, C = Crab)

		Mean (±SE) Relative Abundance						
Family	**Species**	**M50g**	**M350g**	**S50g**	**S350g**	**C50g**	**C350g**	**Control (no bait)**
Triakidae	*Mustelus mustelus*	0.43 (±0.20)	1.00 (±0.00)	0.29 (±0.18)	1.57 (±0.30)	0.17 (±0.17)	-	-
Mollusca								
Gastropoda	-	-	-	-	-	0.33 (±0.21)	-	-
Sepiidae	*Sepia officinalis*	-	-	-	-	0.17 (±0.17)	-	-

Family	**Species**	**M200g**	**S200g**	**C200g**	**Control (no bait)**
Ria Formosa Lagoon, Portugal					
Chordata					
Atherinidae	*Atherina presbyter*	0.63 (±0.63)	-	0.10 (±0.10)	-
Sparidae	*Diplodus puntazzo*	-	-	-	0.10 (±0.10)
Mugilidae	-	2.00 (±2.00)	-	-	0.80 (±0.80)
Paguridae	-	0.75 (±0.31)	0.50 (±0.31)	0.50 (±0.17)	0.50 (±0.27)
Portunidae	-	0.13 (±0.13)	-	-	-
Rajidae	-	-	0.10 (±0.10)	-	-
Mollusca					
Gastropoda	-	-	-	0.10 (±0.10)	-

3.3.1. Swansea Bay

3.3.1.1. Relative Abundance

The PERMANOVA test for relative abundance (*MaxN*) showed statistical differences between the four bait type treatments in Swansea Bay ($F_{3,44}$ = 4.5051, P = 0.01; Table 3), but not for the bait quantities within the different bait type treatments ($F_{3,44}$ = 2.948, P = 0.05; Table 3). A pair-wise test identified that the control treatment (no bait) recorded a significantly lower relative abundance compared to all three baited treatments. No differences in relative abundance between the three bait types were observed.

Overall patterns showed that all three baited treatments presented similar mean relative abundances (*MaxN*) captured on camera with 3.29 (±1.03), 3.27 (±0.76) and 3.25 (±1.07) individuals for squid, mackerel and crab respectively (Fig.3a & b). The control (no bait) treatment presented a mean of 0.64 (±0.25) individuals. When splitting the bait treatments by weight, 350g of squid had the highest mean relative abundance (4.71 ±1.74) (Fig. 3a). Smaller quantities (50g) of crab and mackerel recorded similar abundances of 4.17 ±1.47 and 4.14 ±1.06 respectively.

3.3.1.2. Taxonomic Diversity

The PERMANOVA test for differences in taxonomic diversity in Swansea Bay showed statistical differences between the four bait type treatments ($F_{3,44}$ = 7.8532, P = <0.001; Table 3) and the bait quantities within the different bait type treatments ($F_{3,44}$ = 4.5706, P = 0.01; Table 3). A pair-wise test identified that the control treatment (no bait) recorded a significantly lower taxonomic diversity compared to all three baited treatments. No significant differences between the three bait types for taxonomic diversity were observed. A second pairwise test for bait quantities identified that only 50g of squid recorded significantly less taxonomic diversity compared to other baited deployments using 350g (Post hoc: t = 2.59, P = 0.03).

Overall patterns showed that mackerel and squid presented a similar mean taxonomic diversity captured on camera with 2.18 (±0.30) and 2.14 (±0.52) individuals respectively (Fig.3a & b). When splitting each bait type treatment by weight in Swansea Bay, 350g of squid recorded the highest mean taxonomic diversity (3.14 ±0.74) followed by 50g mackerel treatment with a mean of (2.43 ±0.37); Fig. 3a).

Table 3 PERMANOVA of MaxN, Taxonomic Diversity and Faunal Assemblage Composition *in Swansea Bay, South Wales and Ria Formosa Lagoon, Portugal. Bold values P <0.05.*

Source	***df***	**MS**	***Pseudo-F***	***P(perm)***	***Unique Perms***
MaxN					
Swansea Bait Type	3	3.5293	4.5051	**0.01**	9950
Swansea Bait Quantity (Bait Type)	3	2.3094	2.948	0.05	9954
Residual	44	0.7833			
Total	50				
Ria Formosa Lagoon Bait Type	3	1.2029	1.3503	0.2778	9269
Residual	34	0.89082			
Total	37				
Taxonomic Diversity					
Swansea Bait Type	3	2.7439	7.8532	**<0.001**	9949
Swansea Bait Quantity (Bait Type)	3	1.5970	4.5706	**0.01**	9948
Residual	44	0.3494			
Total	50				
Ria Formosa Lagoon Bait Type	3	0.25788	0.75963	0.5297	971
Residual	34	0.33948			
Total	37				
Faunal Assemblage Composition					
Swansea Bait Type	3	4651.7	4.758	**<0.001**	9929
Swansea Bait Quantity (Bait)	3	3038.4	3.039	**0.001**	9928
Residual	44	978.31			
Total	50				
Ria Formosa Lagoon Bait Type	3	563.82	0.78809	0.5969	9933
Residual	34	715.42			
Total	37				

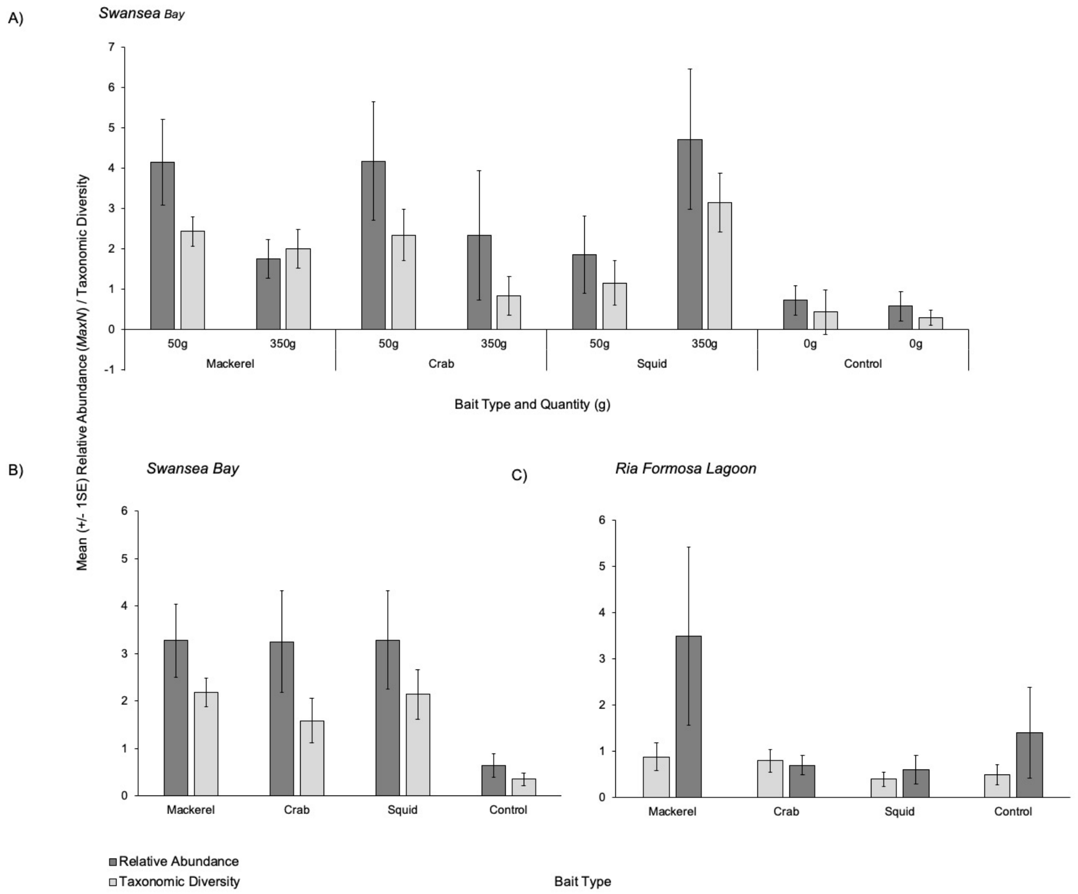

Figure 3: Mean (+/- 1 SE) Relative Abundance / Taxonomic Diversity a) for the different bait and weight treatments in Swansea Bay b) for the combined bait type treatments in Swansea Bay c) for the bait type treatments in Ria Formosa Lagoon.

3.3.1.3. Faunal Assemblage Composition

The PERMANOVA test of faunal assemblage composition in Swansea Bay showed a significant treatment effect for bait type ($F_{3,44}$ = 4.758, P = <0.001; Table 3). A pair-wise test identified that significant differences were present between the control treatment and all three baited treatments. Statistical differences were also identified between crab and the squid and mackerel treatments (Post hoc; t = 1.74, P = 0.03 and t = 1.67, P = 0.04 respectively). No statistical differences were present between mackerel and squid. For bait quantities, statistical differences in faunal composition were present ($F_{3,44}$ = 3.039, P = 0.001; Table 3) between the 50g and 350g treatments within the squid bait treatment (Post hoc; t = 2.04, P = 0.02) only.

The CAP plot for bait type in Swansea Bay (Fig. 4a) showed patterns in bait type identified in the PERMANOVA. Control deployments were separated out from the majority of the squid and mackerel deployments along CAP axis 1 in the negative values with crab deployments occasionally overlapping. Baited deployments ranged into both the positive and negative values of CAP axis 1. With the exception of the control treatment (85.71%), the baited groups had a relatively low 'leave one out' allocation success. Out of the 51 samples, only 21 of the deployments were correctly classified. There was a mis-classification error 58.82%, indicating that overall similar faunal assemblages were sampled using baited deployments when compared to unbaited deployments.

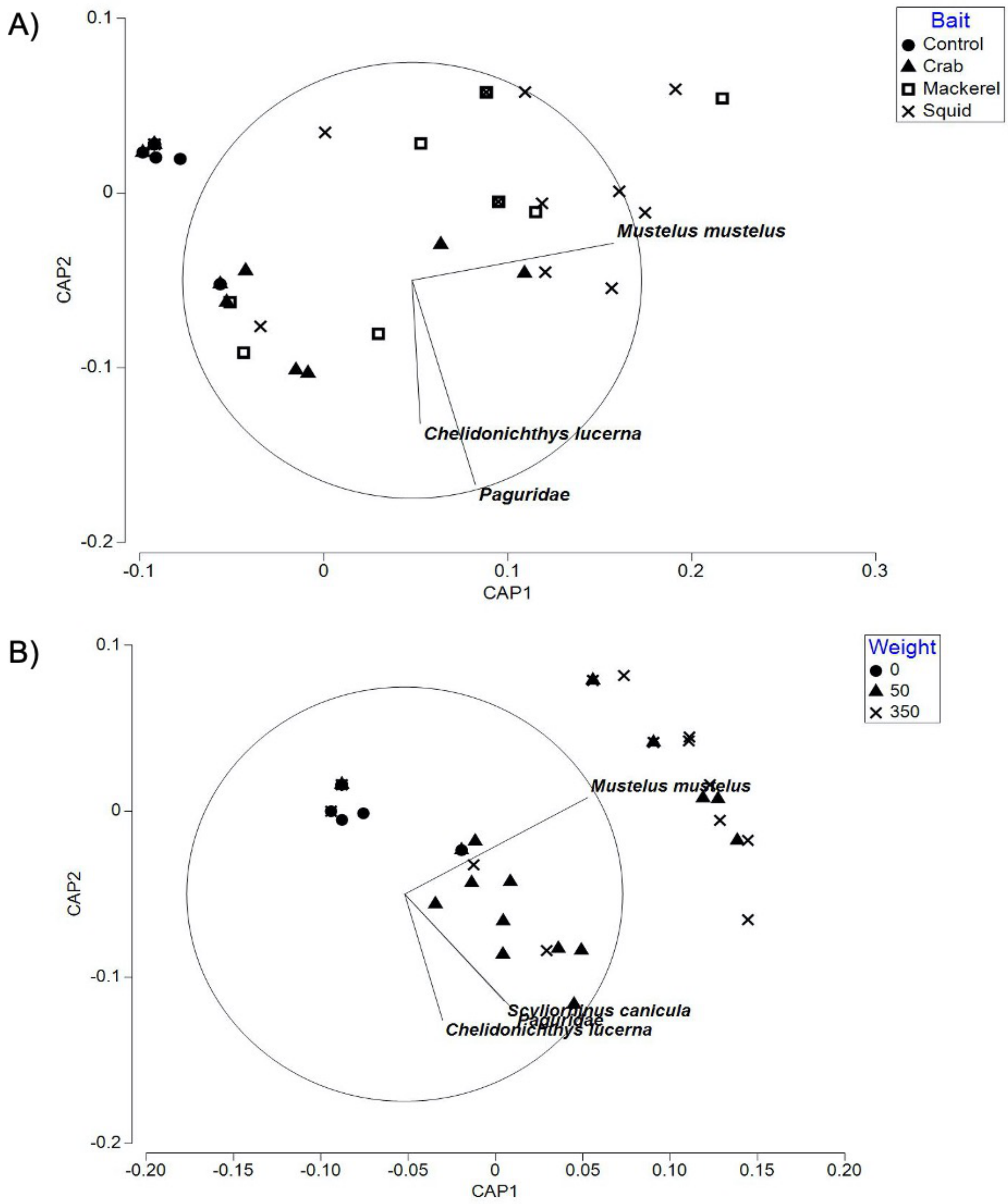

Figure 4: CAP ordination for fish assemblages sampled by BRUV deployments in a) Swansea Bay for the four bait type treatments b) Swansea Bay for the three bait quantity treatments. Vector lines refer to strongly correlated faunal assemblages (>0.6) with the direction and length of line indicating the direction and strength of correlation in relation to the 1st and 2nd CAP axes.

Similarly, the CAP plot for bait quantities (Fig. 4b) also showed the control treatment to be separated out from the majority of the baited treatments along CAP axis 1. All taxa with a correlation greater than 0.6 to either CAP axes were correlated towards the baited treatments. The 'leave one out' allocation showed a good allocation success for the control treatment (85.71%) but a low allocation success for the 50g (45.00%) and 350g (58.82%) treatments. Out of the 51 samples, 31 of the deployments were correctly classified with a mis-classification error of 39.22%.

A test of multivariate dispersions (PERMDISP) between treatments in Swansea Bay identified a statistically significant variation between bait type treatments (bait type; 3,47 $F = 5.207$, $P = 0.01$) and bait quantities (bait weight; 2,48 $F = 12.531$, $P = <0.001$), suggesting a significant spread of these results around the spatial mean. Deployments of the control (34.42 ± 2.96) and 50g of bait treatments were identified to be the most variable (38.40 ± 1.63).

A SIMPER analysis (Table 4) identified higher abundances of Paguridae in crab deployments, Gadidae in control deployments, and *M. mustelus* and *S. canicula* in mackerel and squid deployments as the main organisms responsible for differences between the three bait types and the control treatment.

Table 4 SIMPER analysis in groups outlined by PERMANOVA showing the organisms which most contributed (>70% cumulative contribution) to the observed differences among bait type treatments in Swansea Bay.

Species	**Av. Abun.**	**Av. Abun.**	**Av. Diss.**	**Diss./SD**	**Contrib. %**	**Cum %**
Av. Diss.: 92.37	**Control**	**Crab**				
Paguridae	0.20	1.01	42.21	1.24	45.73	45.73
Gadidae	0.17	0.08	17.06	0.50	18.49	64.22
M. surmuletus	0.10	0.00	5.21	0.27	5.64	69.87
Gobiidae	0.00	0.23	4.77	0.48	5.16	75.03
Av. Diss.: 93.72	**Control**	**Mackerel**				
M. mustelus	0.00	0.64	33.91	0.96	35.42	35.42
Paguridae	0.20	0.77	25.50	1.06	26.63	62.06
C. lucerna	0.00	0.27	8.22	0.56	8.58	70.64
Av. Diss. :95.36	**Control**	**Squid**				
M. mustelus	0.00	0.75	31.21	1.04	32.46	32.46
Paguridae	0.20	0.70	24.36	0.97	25.33	57.79

Table 4 SIMPER analysis in groups outlined by PERMANOVA showing the organisms which most contributed (>70% cumulative contribution) to the observed differences among bait type treatments in Swansea Bay.

Species	**Av. Abun.**	**Av. Abun.**	**Av. Diss.**	**Diss./SD**	**Contrib. %**	**Cum %**
S. canicula	0.00	0.39	14.55	0.60	15.13	72.91
Av. Diss.: 85.44	**Crab**	**Squid**				
Paguridae	1.01	0.70	25.47	1.06	29.89	29.82
M. mustelus	0.08	0.75	21.05	0.86	24.66	54.48
S. canicula	0.08	0.39	10.66	0.57	12.48	66.97
Gobiidae	0.23	0.29	6.42	0.68	7.52	74.48
Av. Diss.: 84.03	**Crab**	**Mackerel**				
Paguridae	1.01	0.77	23.30	1.13	27.94	27.94
M. mustelus	0.08	0.64	23.05	0.79	27.64	55.58
C. lucerna	0.20	0.27	7.740	0.64	9.28	64.86
S. canicula	0.08	0.31	6.390	0.62	7.67	72.52

3.3.2. Ria Formosa Lagoon

3.3.2.1. Relative Abundance

The PERMANOVA test for relative abundance in the Ria Formosa Lagoon showed no statistical differences between the four bait treatments (Fig. 3c) ($F_{3,37} = 1.3503$, $P = 0.2778$; Table 3). Overall patterns show that mackerel recorded the highest relative abundance (*MaxN*) with a mean of 3.50 (±1.93) individuals. Squid had the lowest mean relative abundance (0.60 ±0.31).

3.3.2.2. Taxonomic Diversity

The PERMANOVA test for differences in taxonomic diversity showed no statistical differences between the four bait types ($F_{3,37} = 0.75963$, $P = 0.5297$; Table 3). Similar to mean relative abundance, mackerel recorded the highest taxonomic diversity in Ria Formosa Lagoon with 0.88 (±0.30) taxa. The squid treatment recorded the lowest taxonomic diversity with a mean of 0.40 (±0.16) taxa.

3.3.2.3. Faunal Assemblage Composition

The PERMANOVA test on relative abundance (*MaxN*) showed no significant treatment effect of bait type on faunal composition ($F_{3,37} = 0.78809$, $P = 0.5969$;

Table 3). The CAP plot for bait type (Fig. 5) confirms the PERMANOVA results with no clear distinction between treatments along CAP axes.

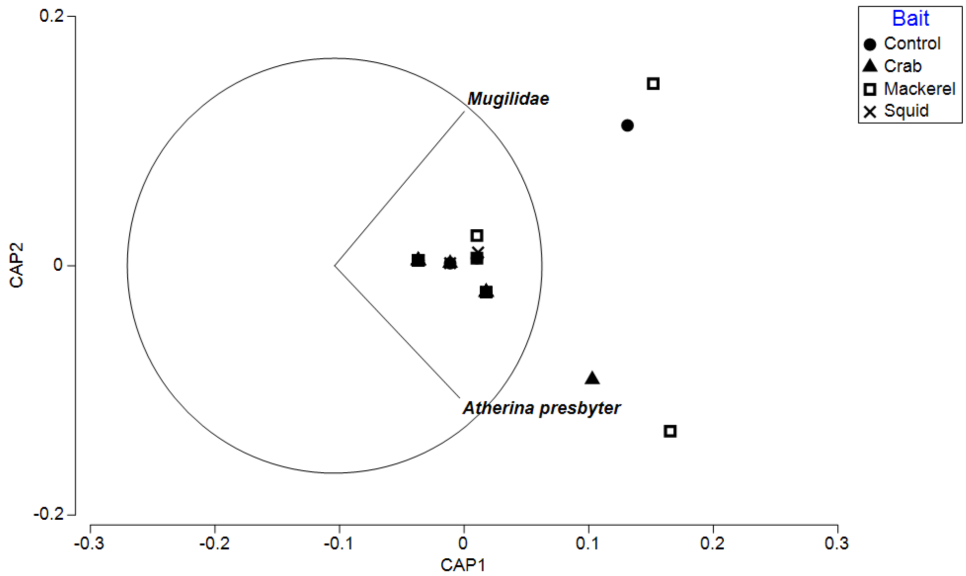

Figure 5: CAP ordination for fish assemblages sampled by BRUV deployments in Ria Formosa Lagoon for the four bait types. Vector lines refer to strongly correlated faunal assemblages (>0.6) with the direction and length of line indicating the direction and strength of correlation in relation to the 1st and 2nd CAP axes.

The 'leave one out' allocation presented a low allocation success (0.00%) for all bait treatments with the exception of the squid treatment (60.00%). Out of the 38 samples, only 10 of the deployments were correctly classified with a misclassification error of 73.68%. The PERMDISP test between bait types showed no statistically significant variation between treatments (bait type: 3,34 $F = 2.018$, $P = 0.301$). A SIMPER analysis was not conducted for Ria Formosa Lagoon as no statistical differences between bait types were identified by the PERMANOVA.

3.4. Discussion

This study found that baited deployments, on average, attracted higher relative abundance and taxonomic diversity of marine fauna compared to unbaited deployments, but that these were not found to be statistically different between bait types. In Swansea Bay, statistically higher relative abundances and taxonomic diversity were recorded for baited deployments compared to unbaited deployments. Faunal assemblage analysis also in Swansea identified differences in composition between baited and unbaited treatments as well as between the crab treatment and the mackerel and squid treatments. We discuss these findings and wider implications for future studies using BRUV deployments to monitor underwater species diversity in the North-Eastern Atlantic Region.

Our findings correspond to previous studies comparing baited and un-baited BRUVs (Bernard & Götz, 2012; Dorman et al., 2012; Hannah & Blume, 2014; Wraith et al., 2013) where the presence of bait has been found to both increase similarity between replicates and detect differences between habitat types (Whitmarsh et al., 2017). The following taxa Paguridae, *M. mustelus, S. canicula* and *C. lucerna* were heavily related to baited deployments. Increased numbers of these scavenging and opportunistic species (Lyle, 1983; Saïdi et al., 2009; Stagioni et al., 2012) also influenced statistical differences between baited and unbaited treatments. Studies applying BRUV methods to deep-sea environments have found similar shifts in community composition when using bait in camera deployments with the abundance of scavenging species greater in the presence of bait (Yeh & Drazen, 2011). Differences in faunal composition were also noted between crab and mackerel and squid baits, however, with further analysis only showing Paguridae as having higher abundances when using crab, mackerel and squid were considered better for faunal coverage.

At both study locations, low numbers of relative abundance and taxonomic diversity were observed for the majority of species, and this could have impacted our results. BRUV performance may have been influenced by variables such as the distribution of species over large spatial areas at both locations. Subtidal soft sediment habitats such as those surveyed in Swansea Bay and Ria Formosa Lagoon provide far less habitat structural heterogeneity when compared to reef type habitats and can remain homogenous over large areas (Syms & Jones, 2004). The distribution of organisms on subtidal soft sediment habitats often depends on factors such as food availability, disturbance and seabed complexity which, in large habitats, may influence a large spatial distribution of individuals (McCormick, 1995; Parsons et al., 2014). Furthermore, underwater visibility was relatively low at both locations which may have limited the relative abundance and diversity of species recorded during deployments. Seven and two of the failed deployments in Swansea Bay and Ria Formosa Lagoon respectively were due to very high levels of turbidity obscuring the bait. The large tidal ranges observed in Swansea Bay alongside its shallow nature equate to a large amount of sediments suspended into the water column (Collins & Banner, 1980). In Ria

Formosa Lagoon high turbidity levels are attributed to agricultural run-off and sewage (Newton & Mudge, 2005).

Relative abundance and taxonomic diversity was lower than expected in the Ria Formosa Lagoon in particular, based on previous monitoring studies using seine netting techniques at this location (Ribeiro et al., 2008). The variability of anthropogenic activity in this instance could have additionally influenced the performance of our BRUV deployments in this study location. There was a notable difference in motorized boat traffic between the two survey locations, with Ria Formosa Lagoon harbouring higher numbers of small motorized vessels over a small spatial area (Correia et al., 2015). Previous studies showed that anthropogenic noise associated with recreational motorized boat activity has the potential to impact fish movements and behaviour (Nichols et al., 2015; Roberts et al., 2016; Slabbekoorn et al., 2010). Further research into the impacts of anthropogenic noise on BRUV performance would provide an interesting insight into this.

Contrary to our hypothesis, more bait (e.g., 350g) did not perform better than less bait, in terms of mean relative abundance observed for baited treatments in Swansea Bay. However, 350g of the squid treatment attracted a significantly higher taxonomic diversity compared to its 50g counterpart during deployments suggesting that a higher quantity of squid is required to gain higher values of diversity compared to other baits. Findings in other coastal studies have found that BRUV deployments with higher bait quantities do not necessarily improve bait performance in deployments (Hardinge et al., 2013). Our finding contradicts previous findings for bait quantities used in traps (Cyr & Sainte-Marie, 1995; Miller, 1983) and plume models in deep sea environments (Sainte-Marie & Hargrave, 1987), where higher bait quantities produced higher relative abundance of scavenging amphipods. This suggests that bait quantity is likely to have a different effect depending on faunal assemblage sampled i.e. swarms of amphipods compared to larger scavengers such as fish or crabs. Taylor et al. (2013) found that, in dynamic coastal environments, bait plume penetration and dispersal could be primarily driven by tidal currents influencing the probability of assemblages locating relevant attractants (Heagney et al., 2007; Hill & Wassenberg, 1999; Stiansen et al., 2010). Although environmental parameters were not measured during this study, our findings support those of previous studies in dynamic coastal environments, which suggest that external environmental factors are more likely to influence bait performance compared to bait quantity.

Various quantities of bait have previously been used in BRUV studies globally ranging from 50g to >2kg (Whitmarsh et al., 2017). Compared to Australian BRUV studies where records show that up to 1kg of bait can be consumed or removed during an hour deployment (Dorman et al., 2012), minimal bait depletion was observed in Swansea Bay for both bait weights tested. This may have also influenced the small differences in relative abundance and taxonomic diversity observed further suggesting that assemblages are equally attracted to the bait throughout the deployment regardless of the bait quantity used (Harvey et al., 2007). In areas where scavenging rates are higher, we expect bait quantity to be more important. Similarly, the majority of 200g bait also remained in the bait bag after all one-hour BRUV deployments in Ria Formosa Lagoon for all three bait types.

3.4.1. Recommendations

Coastal habitats assessed during this research comprised primarily of subtidal soft sediments in dynamic environments less than 10m depth. Although, no one individual bait type provided a statistically higher *MaxN* or taxonomic diversity, smaller quantities of mackerel and crab (e.g.50g), were found to produce similar values of *MaxN* and taxonomic diversity as larger quantities (e.g. 350g) of squid in Swansea Bay. It is recommended for consistency and standardisation that when implementing BRUV methods in these environments, bait use should include locally sourced oily fish such as mackerel. Compared to other bait types such as crustaceans and cephalopods, oily fish is considered a much cheaper alternative and is readily available in both local angling shops and supermarkets. Following methods used in previous studies, best practice for bait is to defrost for at least 24 hours prior to deployments in order to generate a greater aroma and bait plume once in the water (Dorman et al., 2012). To maximise effectiveness, bait should be replenished after each deployment as an increased soak time has been found to reduce bait quality over time (Løkkeborg & Johannessen, 1992). We suggest that for deployments in this region, minimum quantities of 50g are sufficient for attracting organisms to the camera field of view.

3.4.2. Conclusions

Statistically higher relative abundances (*MaxN*) and taxonomic diversity were recorded for baited camera deployments compared to unbaited deployments in Swansea Bay. Statistical differences were also found for faunal assemblage composition between bait types in this study area. Mackerel and squid recorded similar abundance values for scavenging species such as *M. mustelus* and *S. canicula* that were statistically greater than those returned by crab or control treatments where records of these species were minimal. We found no statistical evidence for a single bait type influencing *MaxN*, taxonomic diversity or faunal composition in the Ria Formosa Lagoon study area potentially due to the lower numbers of abundance and diversity recorded in this location.

3.5. Acknowledgements

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Chapter 4

Improving visual biodiversity assessments of motile fauna in turbid aquatic environments

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Author contribution

RAG and Ocean Ecology Ltd designed the CLOC system used for this research and, alongside RKFU, conceived the ideas for this study. REJ led the design of the experimental procedures and experimental set up; REJ and SCR collected the data; REJ analysed the data and led the writing of the manuscript. RAG and RKFU reviewed the manuscript with all authors giving final approval for submission.

Author	**Institution**	**% Contribution**
Robyn Jones	Swansea University	85
Ross Griffin	Ocean Ecology Limited	5
Sam Rees	Swansea University	5
Richard Unsworth	Swansea University	5

We the undersigned agree with the above stated "proportion of work undertaken" for each of the above published peer-reviewed manuscripts contributing to this thesis:

Signed Candidate ______ [redacted] ______________________

Author 1_____________ [redacted] __________

Author 2__________ [redacted] _______

Author 3___________ [redacted] ________

Author 4________ [redacted] _________

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 non-extractive 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 trialled 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 = <0.001). The ability to identify taxa to species level in field conditions was also significantly increased with the addition of a CLOC (P = <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.

Keywords

Baited Remote Underwater Video; Faunal identification; Motile fauna; Turbidity; Underwater visibility; Visual biodiversity assessments.

4.1. Introduction

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 (Smith, 2003; Wilber & Clarke, 2001). The ecological knowledge we have of these environments currently relies heavily on the use of extractive sampling methods (Costello et al., 2017) limiting their capacity to directly observe either the habitat or associated flora and fauna and are increasingly prohibited in areas covered by Marine Protected Area management. Such areas are critically important for biodiversity, fisheries, energy and ecosystem services such as carbon storage (Levin et al., 2001; Meybeck, 1993). Coastal turbid environments are often areas of the world characterised by rapid population expansion, resource exploitation and energy development due to their dynamic nature (Mélin & Vantrepotte, 2015).

Marine renewable energy development in sectors such as offshore wind, tidal and wave energy are an example of resource exploitation in these dynamic areas and are considered a key objective in many countries (Inger et al., 2009). The impacts on fish 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; Griffin et al., 2016; Lindholm et al., 2015; Unsworth et al., 2014). 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 (Gill, 2005; Heiskanen et al., 2016; Pelc & Fujita, 2002; Sheehan et al., 2010).

Non-destructive 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 & Pelletier, 2014). Acoustic methods can be used as an alternative in this instance; however, these techniques are also susceptible to backscatter in areas of high turbulence (Evans & 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 (Cappo et al., 2006; Watson et al., 2005; 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 (Bicknell et al., 2016; Davies et al., 2001; Mallet & Pelletier, 2014). 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.

4.2. Materials and Methods

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 vison of a fish community for measuring length) 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 analysed 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).

4.2.1. The CLOC-BRUV System

A custom-built frustum stainless-steel frame of the dimensions L170cm (diagonal length) x W64cm x H93cm designed by Ocean Ecology Ltd. and fabricated by R. W. Davis & Son Ltd (Gloucester, UK) (Fig.1) was used for the CLOC-BRUV deployments.

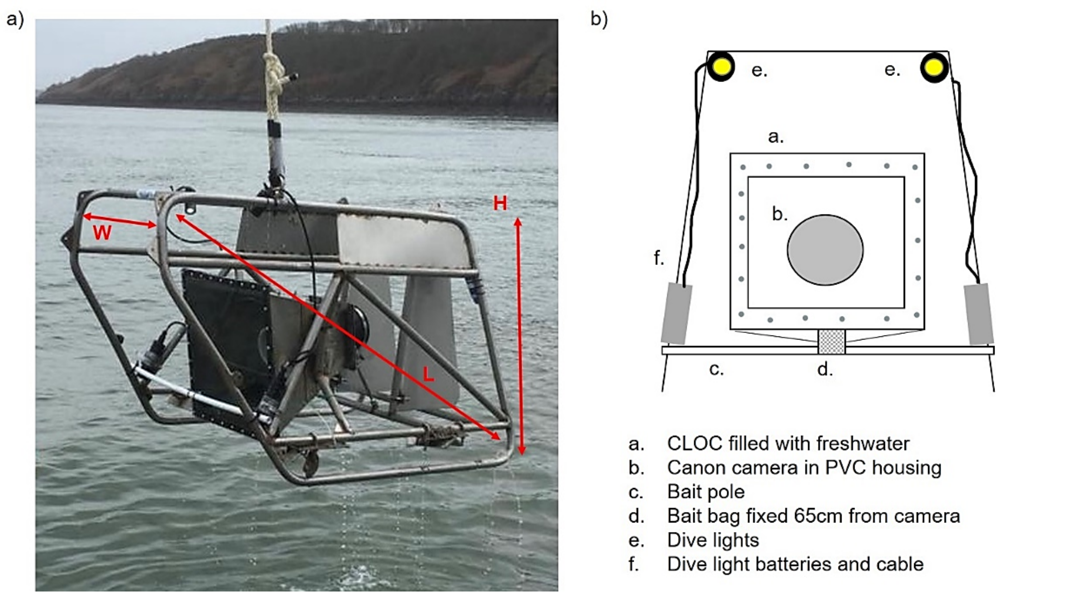

Figure 1: a) Image of CLOC frame during field deployments b) Simple schematic of CLOC system set up for field deployments including camera, bait and light positioning.

Mounted on the centre of the frame, the CLOC, fitted with a clear square polycarbonate lens and filled with approximately 75L of freshwater, (L61cm x W61cm x H60cm) 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 80kg. When full, this weight increased to 155kg. 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 150mm-165mm Drainage Coupling DC165. Fins were positioned at the back of the frame for orientating with the prevailing current and a customised rubber gasket, lined with silicone grease, was bolted between the polycarbonate lens and the metal components of the CLOC to keep watertight.

The camera had a 20x HD Video Lens offering a 35mm equivalent of 26.8mm–576mm, 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 stabilisation were disabled during deployments (Unsworth et al., 2014). Video data was recorded on to internal Secure Digital (SD) cards. A bait pole was fixed parallel to the CLOC at a distance of 65cm from the camera and approximately 30cm from the floor in order for the bag to be comfortably in the field of view of the camera. A 5mm PVC mesh bait bag was positioned in the centre 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 (Unsworth et al., 2014; Wraith et al., 2013); therefore, Atlantic Mackerel *Scomber scombrus* was used as bait. Approximately 250g 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.

4.2.2. The Stereo-BRUV System

The stereo-BRUV system was based on the same set up used by Unsworth et al., (2014). A custom-built galvanised steel frame of the dimensions L80cm x W50cm x H50cm was used for these deployments (Fig. 2).

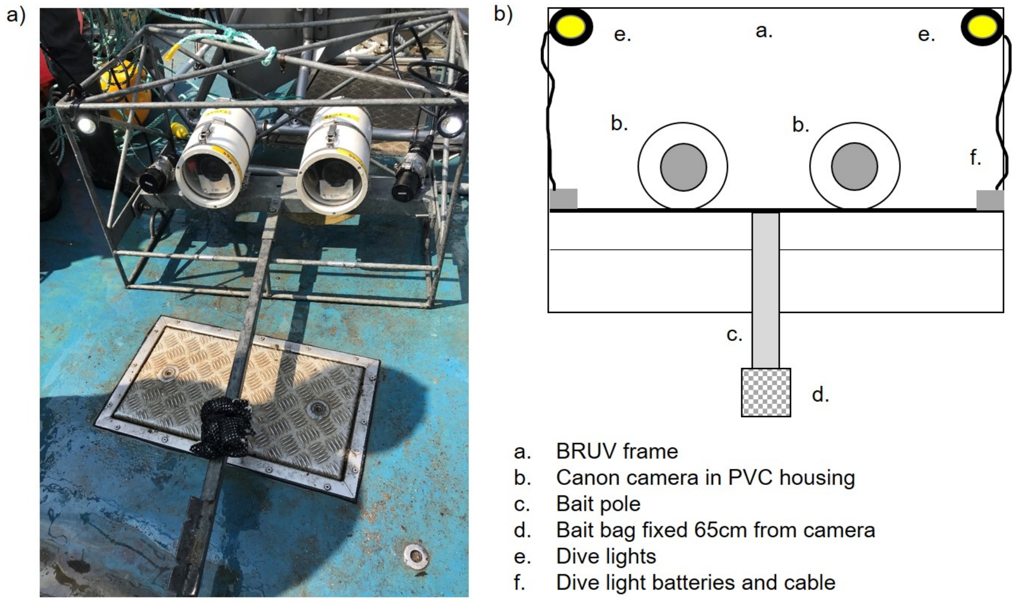

Figure 2: a) Image of stereo housing system during field deployments b) Simple schematic of stereo-BRUV system set up for field deployments including camera, bait and light positioning.

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 30cm. Focal length was set to infinity (∞) and face detection and tracking, and image stabilisation were disabled during deployments (Unsworth et al., 2014) with a horizontal field of view of 45.5°. Video data was recorded on to internal SD cards. A 65cm bait pole with a 5mm mesh bag was mounted in front of the cameras, approximately 15cm to the floor. Approximately 250g 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 40cm compared to 20cm 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.

4.2.3. Laboratory Trails

Trials comparing the efficiency of a CLOC in a low visibility environment were carried out under controlled conditions. One cylindrical tank (r= 0.75m, h= 1.5m) was filled with approximately 1.78m^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*; Lesser Spotted Dogfish *Scyliorhinus canicula)* were placed 65cm from the Canon HFG model camera in both systems. Diluted *Chlorella sp.* algae was added to the tank in approximately 5 litre batches to reduce water visibility from over 1m (0μg^{-1} *Chlorella sp.* per 100ml) to 0.25m (3.4μg^{-1} *Chlorella sp.* per 100ml). This was calculated by filtering 100ml 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 >1m to 0.25m.

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.

4.2.4. Field Trials

Comparative BRUV deployments were undertaken at four locations across the UK (Fig. 3) in areas of varying underwater visibility ranging from 3.5 to 7.5m (Table 1).

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	51º 42.761' N	5º 06.798' W	3	3
Freshwater West, Milford Haven	51º 39.767' N	5º 05.265' W	3	3
Aberavon, Swansea Bay	51º 35.153' N	3º 50.854' W	3	3
St Anthony, Falmouth	50º 08.570' N	5º 01.220' W	4	4

For each location, an area of 200m x 200m 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 deployed simultaneously within a distance of 50m of each other for a period of one hour during daylight hours (8am-6pm) based on previous camera comparison methods (Logan et al., 2017; Unsworth et al., 2014). A distance of 50m 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 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.

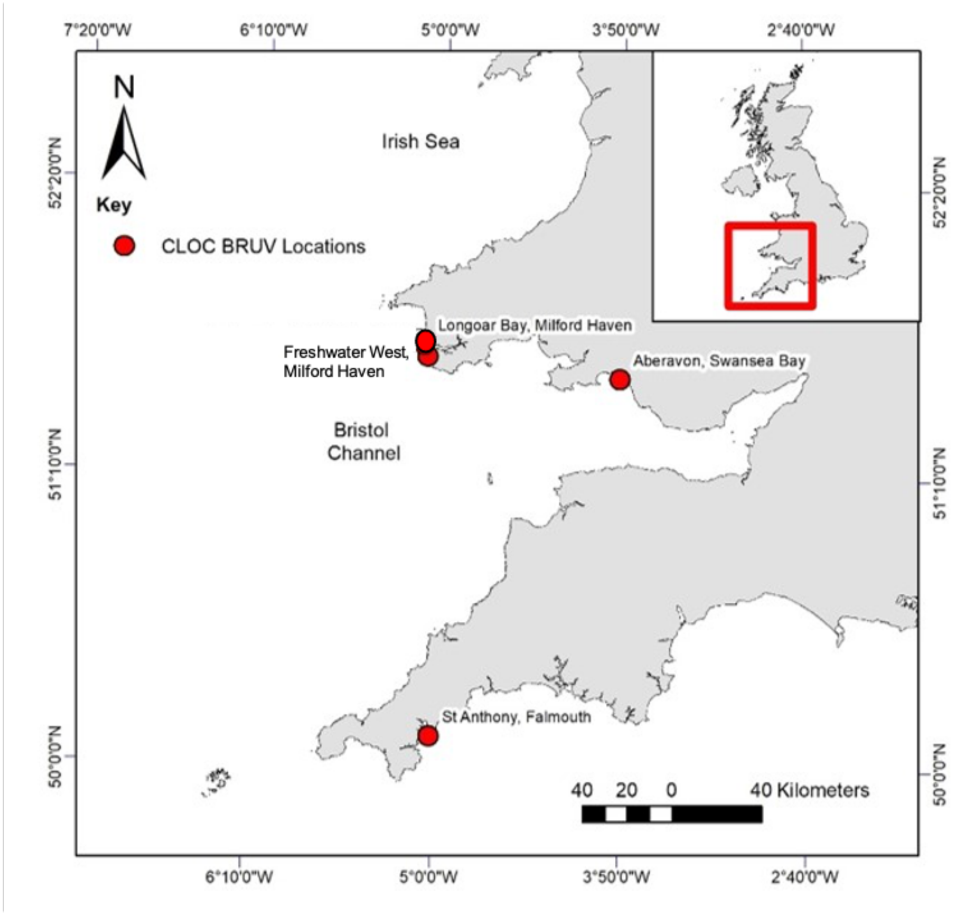

Figure 3: Map showing the four locations of CLOC and comparison stereo-BRUV deployments taken around the South Wales Coast and South West England (UK).

4.2.5. 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 totalled 22 images across the eight visibility levels generated (Table 2).

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

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

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 UK. Raw footage was compressed from Advanced Video Coding High Definition (AVCHD) format (standard format for digital recordings and high-definition video camcorders) to Audio Video Interleave (AVI) 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 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. *MaxN* is a measure of relative abundance to avoid repeated counts of individuals. Analysis of footage started once the camera system was positioned on the seabed and stopped once the camera was lifted off the seabed during recovery. Sediment settling times once the camera systems hit the seabed were also recorded.

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

4.2.6. Statistical Analysis

Summary data is presented as means ± one 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 was Arcsine transformed (for percentage data) 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 & 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 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.

4.3. Results

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.

4.3.1. 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.

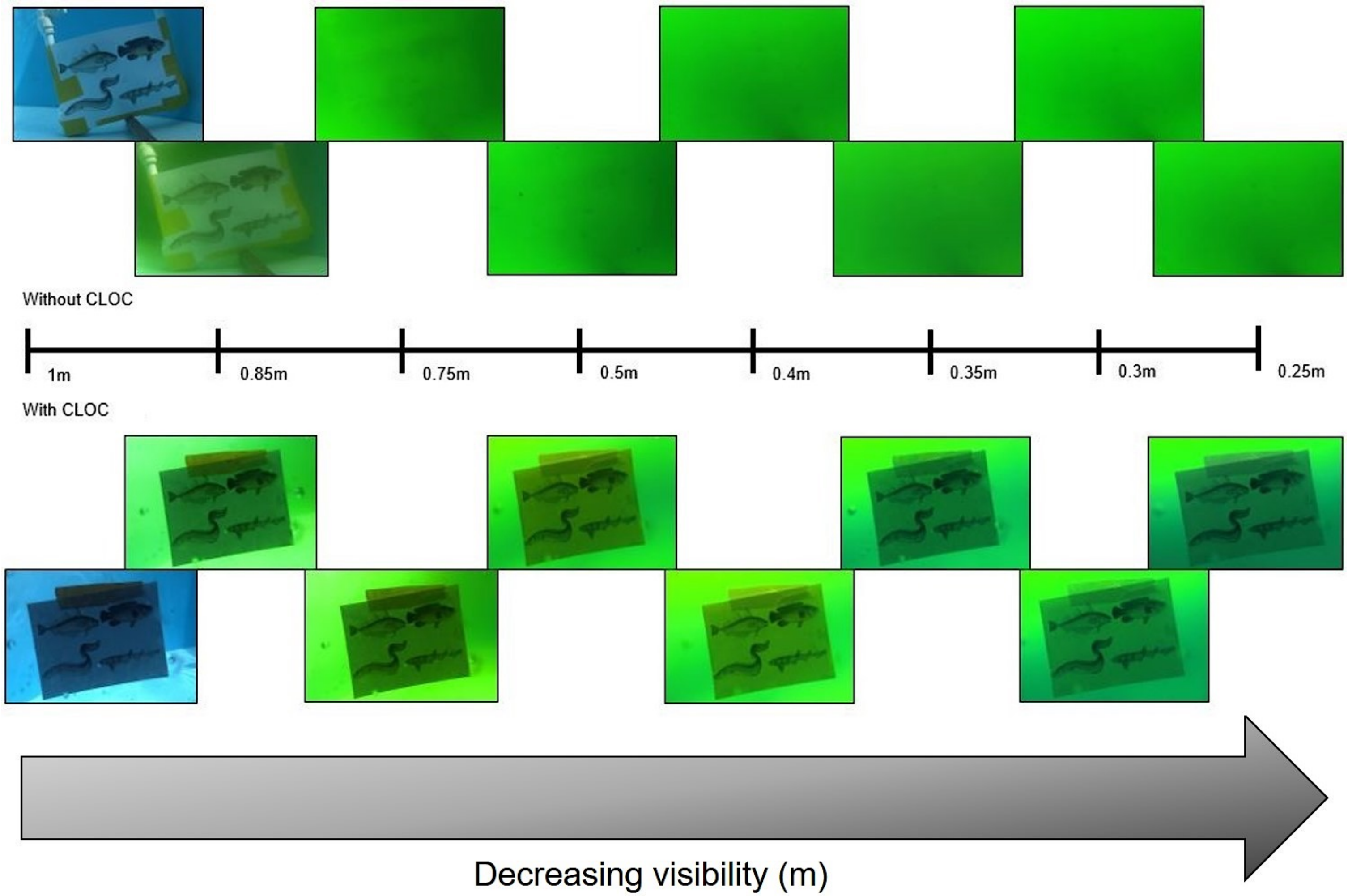

Figure 4: Observations taken from the recordings of simultaneous deployments of the two BRUV systems across the eight different underwater visibilities measured under controlled conditions using a Secchi Disk. Images of the four fish species are positioned 65cm from the camera in both BRUV system set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

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 = <0.001, Ballan wrasse H_1 = 26.41, P = <0.001, Conger eel H_1 = 14.64, P = <0.001, Lesser spotted dogfish H_1 = 26.40, P= <0.001) (Fig. 5). In perfect visibility conditions i.e., no algae added into the tank, the ability to identify features without a CLOC was not affected for all four fish species.

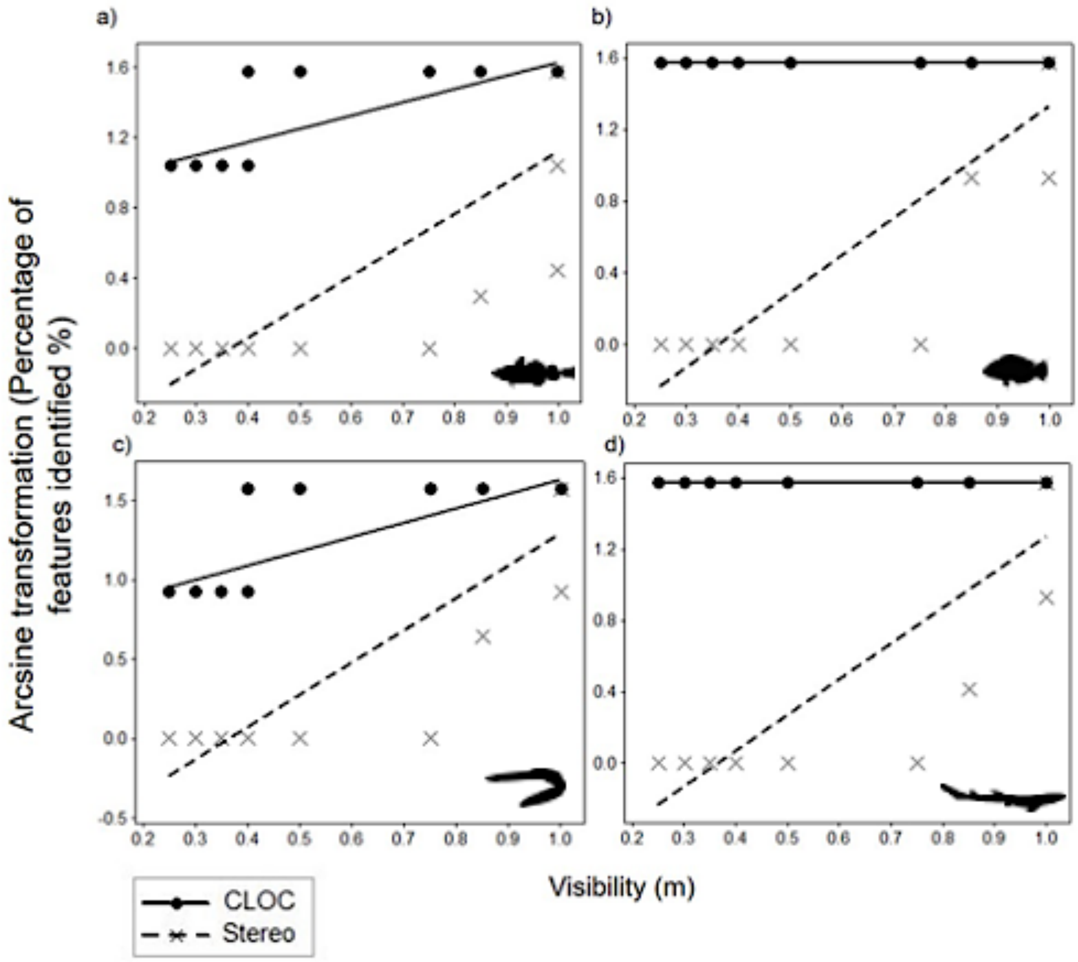

Figure 5: Percentage of features (Arcsine transformed) successfully identified with a CLOC and without a CLOC (Stereo) for a) Whiting b) Ballan wrasse c) Conger eel d) Lesser spotted dogfish across eight Secchi Disk visibility readings (m) in a controlled environment.

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 were reduced to 90% (0.05 ± 1SE) and 87% (0.07 ± 1SE) respectively at an underwater visibility of 0.4m. At an underwater visibility of 0.35m, 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 1m after algae was added when using the stereo-BRUV system. When visibility was reduced to 0.85cm, 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 0.75cm and below, the average ability to identify features for all four fish species was zero.

4.3.2. 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 65cm enhanced when using the CLOC (Fig. 6). For instance, at an underwater visibility of 3.5m, a lesser spotted dogfish was clearly visible when utilising 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 re-suspension 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 seconds compared to 22.8 ±4.2 seconds for the CLOC-BRUV system.

Figure 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 (3.5m) in Swansea Bay, South Wales. An individual is also present in the imagery without the CLOC; however, distinguishing features are difficult to determine.

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

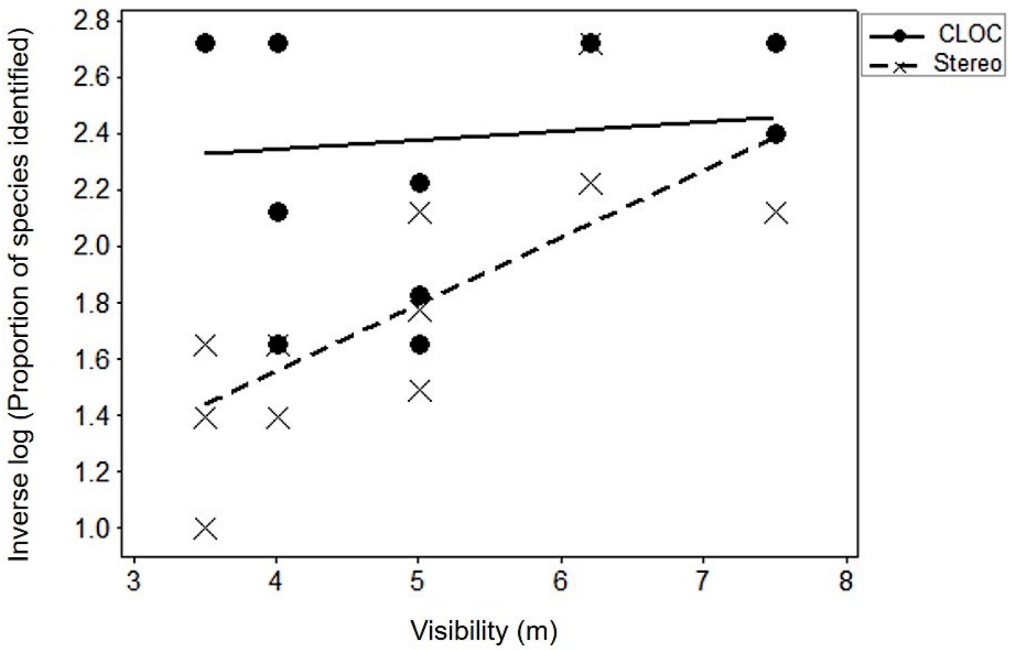

Figure 7: Proportions of species (after undergoing and Inverse log transformation) identified from the taxa recorded during comparative camera system deployments with and without a CLOC across four locations of varying underwater visibility (m) taken from the South Wales and South-West England coastlines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

Comparisons between the species richness and arrival times of taxa into the camera field of view for both camera systems are 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 hour as standard BRUV systems.

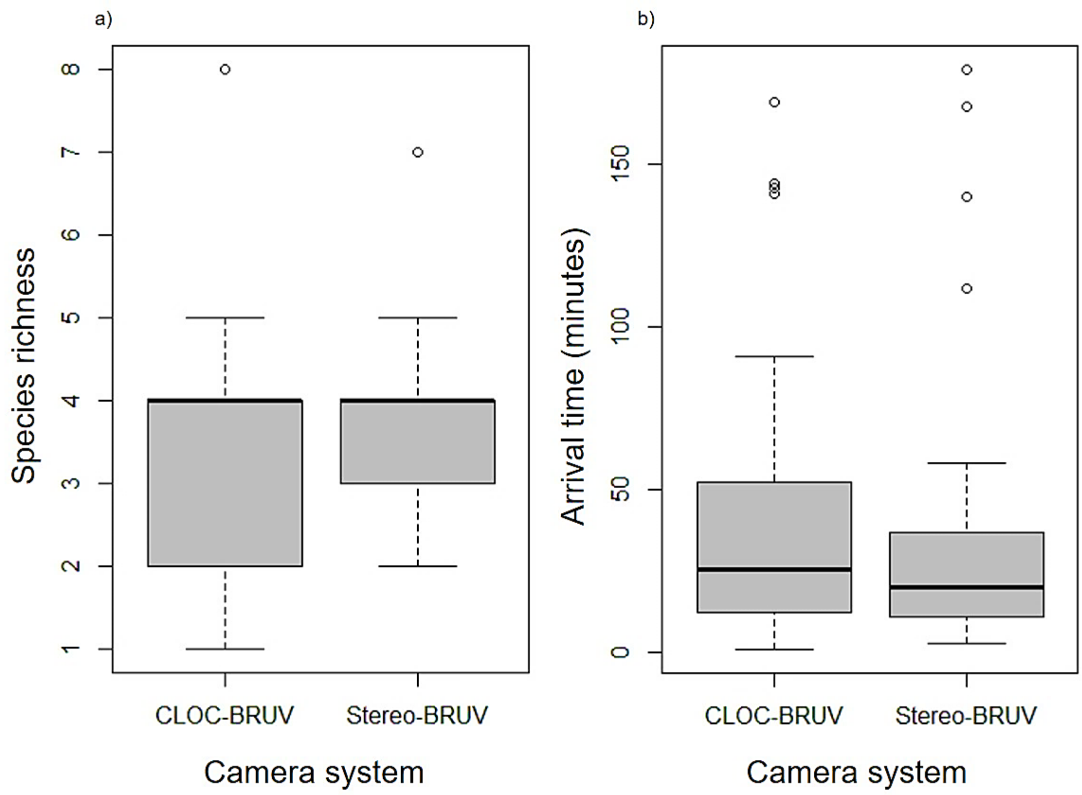

Figure 8: Boxplot (box ranging from first to third quartile and highlighting median value, whiskers extending to 1.5 the interquartile distance with circles indicating outliers) showing the a) species richness b) arrival times of taxa into the camera frame for the two camera systems stereo-BRUV and CLOC-BRUV.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

4.4. 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.

4.4.1. Feasibility

Underwater cameras are a common and non-destructive 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 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 organisations 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.

4.4.2. Alternative Camera Methods

Other methods of visualising 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 & 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 (Brown et al., 2011; Mackinson et al., 2002; Martignac et al., 2015; McClatchie et al., 2000).

4.4.3. 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 habitats 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 customisable 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 4m 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 seconds.

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.

4.4.4. 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^{I}), Simpson's Index of Diversity (λ) and Species Evenness (J') (Gray, 2000). As species are usually the interest when trying to characterise an area, acquiring good quality footage allowing for the successful identification of taxa to the species level is required whilst minimising 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 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 minimising 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.

4.5. Acknowledgements

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Chapter 5

Improving benthic biodiversity assessments in turbid aquatic environments

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Author Contribution

RJ, RU and RG conceived the ideas and designed the methodology; RJ collected and analysed the data; JH developed the BIIGLE label tree for image analysis; RJ led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

Author	**Institution**	**% Contribution**
Robyn Jones	Swansea University	80
Richard Unsworth	Swansea University	5
Jon Hawes	CEFAS	5
Ross Griffin	Ocean Ecology Limited	10

We the undersigned agree with the above stated "proportion of work undertaken" for each of the above published peer-reviewed manuscripts contributing to this thesis:

Signed Candidate ___ [redacted] _________

Author 1________ [redacted] __________

Author 2_________ [redacted] _________

Author 3_______ [redacted] ___________

Author 4__________ [redacted] ____________

Abstract

Biodiversity in turbid aquatic environments is commonly assessed using extractive sampling methods that damage the seabed. Underwater cameras equipped with Clear Liquid Optical Chambers (CLOCs) for the assessment of seabed habitats and species are a non-extractive alternative and have been applied in turbid environments globally. A CLOC is a body of clear liquid positioned in front of a camera to reduce the scattering of light that would otherwise occur when passing through the turbid water it displaces. Here, we test and quantify the effectiveness of CLOC for marine benthic biodiversity assessments over gradients of increasing turbidity.

The addition of a CLOC to a conventional benthic camera system significantly enhanced the quality of information gathered. Images acquired using the CLOC system consistently recorded statistically higher values of image quality (49% increase based on the clarity of the image), percentage seabed visible within the drop-down frame (34% increase) and European Nature Information System habitat level identification (49% increase). Furthermore, it was found that the 'annotation success' of taxa were found to increase between individual experts in the presence of a CLOC. A reduced sampling effort was also identified when using a CLOC. Taxonomic richness increased by 27% when comparing the same number of image stills collected with and without the CLOC.

By reducing the limitations of underwater visibility previously attributed to underwater cameras, this concept extends the potential for use of non-destructive survey techniques and allows for future users to collect robust information of an area, making better informed management decisions.

Keywords: Benthic habitat assessments; Biodiversity; Underwater imagery; Environmental management; Dropdown video; Turbidity

5.1. Introduction

Coastal environments comprise some of the most of productive and biodiverse ecosystems on the planet (Costanza et al., 1997). Understanding and assessing the biodiversity in these regions is essential for their sustainable management and the resources they supply to humanity (Carstensen, 2014; Heery et al., 2017). In many coastal areas this need is compromised by high turbidity, necessitating the widespread use of extractive and destructive biodiversity and habitat sampling techniques, even in sites of high conservation value (Orpin et al., 2004). Improved methods for non-destructively understanding habitats, biodiversity and species of conservation concern in turbid environments are needed to enhance their conservation management (O'Byrne et al., 2018).

A key element of such assessments is the characterisation and monitoring of the benthic environment (Howell et al., 2010). Benthic surveys identify and enumerate species present to understand this distribution of habitats and biological communities. They are undertaken by regulators, industry, academic scientists and NGOs to manage, understand and reduce impacts (Cordes et al., 2016; Leeney et al., 2014; Maclean et al., 2014).

Traditional sampling techniques for seabed habitat classification and assessment of benthic communities are destructive in nature, they include trawling, dredging and sediment sampling (Eleftheriou, 2013). Although underwater cameras are used as a non-destructive alternative for benthic assessments (Bicknell et al., 2016; Bethoney & Stokesbury, 2018) high levels of turbidity from either suspended sediments or high planktonic concentrations greatly reduce the quality of the imagery obtained restricting their use (Jones et al., 2019).

The quality of information that can be derived from seabed imagery such as species composition and habitat type, relies heavily on the underwater visibility in the areas of interest, ranging from deep sea to coastal environments. This can lead to ambiguity and mis-interpretations of data by analysts (Collin et al., 2011; Underwood & Chapman, 2013), resulting in potential management and conservation implications. Various methods have been trialled to address this problem, notably the use of acoustic imaging and Clear Liquid Optical Chamber (CLOC) technologies (Griffin et al., 2020; Jones et al., 2019). Otherwise known as 'freshwater lenses' or 'clearwater boxes', CLOCs use a body of clean freshwater in front of a camera sensor in a custom-built frame with an interface of non-concave glass or Perspex between the freshwater and seawater. This addition of clean liquid reduces the scattering of light that would otherwise occur when passing through the turbid water it displaces without limiting the field of view (Hitchin et al., 2015). Documented uses of the 'freshwater lens' concept stretch as far back as 1997 where underwater visual inspections of submarine structures by divers utilised equipment such as clear-water masks and plastic bags used as a clear water lens (Fang, 1997). Furthermore, ecological uses of this concept also originated around the same time through the use of handheld 'illuminated underwater seagrass viewers' used when diving in highly turbid waters in Australia (Coles et al., 1997). Since then, such systems have been developed for use in various aquatic environments and operate under a range of different designs. More recent documented uses of this concept include the use of clear water boxes for underwater bridge inspections (Browne et al., 2010) with additional applications expanding the working window for the use of these lenses to compact cameras such as GoPros (Sexton Underwater Products, 2017).

This research expands on our existing findings on the use of CLOC systems for improving the identification of species during baited remote underwater video deployments (Jones et al., 2019) and applies the same concept to the collection of seabed imagery. The present study tests the relative benefits of CLOC usage over conventional camera systems by comparing the quality and detail of information gathered during CLOC and non-CLOC benthic camera deployments in reduced underwater visibilities ≥0.5m and across five habitat types.

5.2. Materials and Methods

5.2.1. Study Sites

Seabed imagery was collected at two locations in South Wales, UK over a gradient of increasing turbidity on soft substrate habitats and an area of biogenic reef formed by the genus of polychaete worms *Sabellaria* in their reef form. These reefs are protected by a variety of European conservation legislation and policies, most notably as Annex-I features under the Habitats Directive (Council Directive 92/43/EEC). The inclusion of *Sabellaria* habitats in this study provided an opportunity to establish if these reef qualifying attributes could be assessed more readily from imagery acquired using a CLOC compared to without.

The two sites sampled during this study were Swansea Bay and The Milford Haven Waterway, South Wales, UK (Fig. 1). Swansea Bay is a highly dynamic area subject to large tidal ranges of up to 10.5m (Waters & Aggidis, 2016) and large tidal currents leading to high levels of suspended sediments and reduced underwater visibility. The Milford Haven Waterway is the largest ria-estuary in the UK. It is a busy shipping channel for passenger ferries and for the petrochemical industry. Strong currents and swell are present at the mouth of the estuary becoming more sheltered further inland with tidal ranges up to 6.1m The Haven Waterway also harbours a number of protected marine habitats; the Pembrokeshire Marine Special Area of Conservation is designated for the features of estuaries and shallow inlets and bays to name a few. (Carey et al., 2015; Pembrokeshire Coast National Park 2013).

5.2.2. Experimental Design

The sampling design was based on a stratified random approach (Noble-James et al., 2017) whereby sampling stations were positioned within a number of key seabed habitat types based on existing European Nature Information System (EUNIS) broad-scale habitat (BSH) maps (European Marine Observation and Data Network, 2018) and previous sediment grab information taken from the Tidal Lagoon Swansea Bay Environmental Survey (Titan Environmental Surveys Limited, 2014). In total, 25 stations (Table 1) were targeted across the two study sites in South Wales (Fig.1) in depths <15m. These included infralittoral sediments such as fine sand, fine gravel and intertidal *S. alveolata* reefs in Swansea Bay and a mixture of rock and soft sediment types including exposed infralittoral rock and sediments dominated by macroalgae in The Milford Haven Waterway. Seabed imagery was collected at each station during two independent drop-down camera deployments, one with and one without a CLOC. Due to only one camera system being available, CLOC deployments at each station were conducted first followed by non-CLOC deployments the next day.

A drop-down method of deployment was used for both systems taking a single observation (image still) per drop once the frame was fully positioned and stable on the seabed (Hitchin et al., 2015). A radius of 10m was used around each station to determine whether the camera deployment was within an acceptable sampling distance from the station. Positioning was determined by the onboard vessel GPS equipment. To avoid sampling the same area, the frame was winched up 2m with the vessel allowed to drift for 2-5 seconds before being redeployed. If the vessel drifted out of the 10m station radius, the frame was winched back up on to the deck whilst the vessel repositioned. For each station sampled with and without the CLOC, a minimum of 5 minutes of video and 10 image stills were collected at each station. In addition to this, at each station one water sample (950ml) was taken 1m above the seabed using a niskin bottle for Total Suspended Sediment (TSS) analysis prior to dropping the cameras. One LaMotte Secchi Disk drop with a calibrated line was also taken as a measure of vertical underwater visibility (Lee et al., 2015). In total, two water samples and secchi drops were therefore collected at each station to account for CLOC and non-CLOC deployments.

Table 1 BSH EUNIS classifications and number of image stills collected during CLOC and non-CLOC deployments at Swansea Bay and Milford Haven, UK.

Survey Location	**Station**	**BSH EUNIS Classification**	**CLOC Stills**	**No-CLOC Stills**
Swansea Bay	1	A5.1 Sublittoral coarse sediment	18	14
	2	A5.2 Sublittoral sand	19	11
	3	A5.2 Sublittoral sand	15	10
	4	A5.2 Sublittoral sand	12	12
	5	A5.1 Sublittoral coarse sediment	21	17
	6	A5.1 Sublittoral coarse sediment	13	10
	7	A2.7 Littoral biogenic reefs	12	11
	8	A2.7 Littoral biogenic reefs	11	10
	9	A5.1 Sublittoral coarse sediment	23	12
	10	A5.1 Sublittoral coarse sediment	40	15
	11	A5.1 Sublittoral coarse sediment	23	17
	12	A5.1 Sublittoral coarse sediment	24	17
	13	A5.1 Sublittoral coarse sediment	25	18
	14	A5.1 Sublittoral coarse sediment	26	25
	15	A2.7 Littoral biogenic reefs	40	11
	16	A2.7 Littoral biogenic reefs	53	15
Milford Haven	1	A5.1 Sublittoral coarse sediment	22	17
	2	A5.1 Sublittoral coarse sediment	14	21
	3	A5.1 Sublittoral coarse sediment	12	21
	4	A3.2 Atlantic and Mediterranean moderate energy infralittoral rock	18	26
	5	A3.2 Atlantic and Mediterranean moderate energy infralittoral rock	18	24
	6	A3.2 Atlantic and Mediterranean moderate energy infralittoral rock	20	24
	7	A5.5 Sublittoral macrophyte-dominated sediment	17	23
	8	A5.2 Sublittoral sand	17	16
	9	A5.2 Sublittoral sand	16	16
Total			**529**	**413**

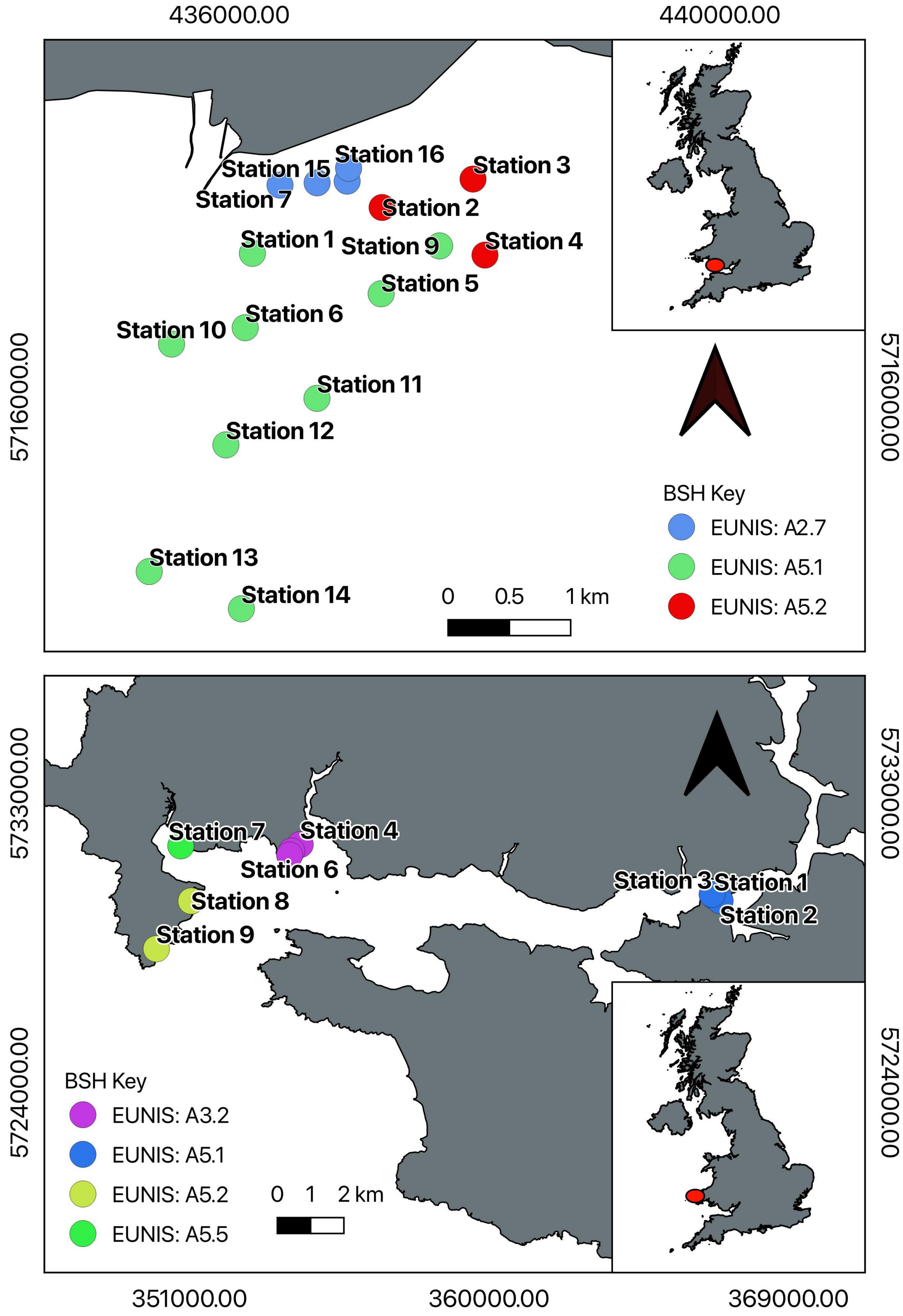

Figure 1: Locations of the 25 stations sampled with and without the CLOC across two locations in South Wales; Swansea Bay (above) and The Milford Haven Waterway (below).

5.2.3. Camera System Set-Up

A custom built frustum stainless-steel frame of the dimensions L170cm (diagonal length) x W64cm x H93cm designed by Ocean Ecology Ltd., fabricated by R. W. Davis & Son Ltd (Gloucester, UK) and was employed during previous work by Jones et al., (2019) was used for this study. The frustrum shaped CLOC (L61cm x W61cm x H60cm) fitted with a clear square polycarbonate lens, filled with approximately 75L of freshwater (tap) and mounted in the centre of the frame using two lateral fixture points. It was then orientated and fixed to a top horizontal bar so the lens was held 10cm above the seabed facing downwards to provide a plan view of the seabed (Fig. 2a, b).

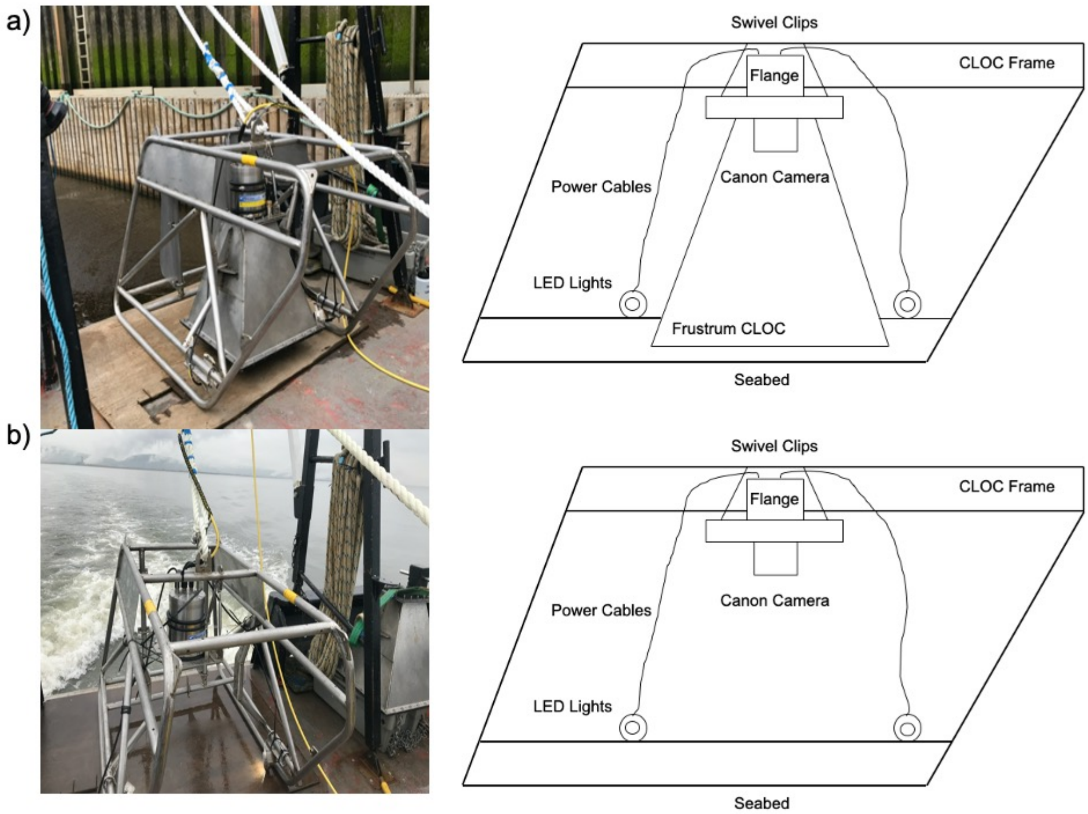

Figure 2: a) Image of benthic CLOC frame during field deployments and simple schematic of benthic CLOC system setup for field deployments b) Image of benthic frame without the CLOC during field deployments and simple schematic of benthic system setup without the CLOC for field deployments.

The weight of the frame and CLOC when empty (i.e. not filled with freshwater) was 80kg. When full, this weight increased to 155kg. A Canon 750D SLR stills camera within a 316 stainless steel underwater housing was mounted into the top of the CLOC using a Flexseal 150mm-165mm Drainage Coupling DC165 allowing for 18 + megapixel image capture. This resulted in the lens of the camera being mounted 65cm above the seabed providing an angle of view of 64° horizontal 46° vertical and 74° diagonal. To calculate the linear field of view the following equation was used:

$$\text{Linear Field of View} = 2\ (\tan\ (\text{Angle of View} / 2) \times \text{Distance to Seabed})$$

The linear field of view was therefore calculated as approximately 81cm x 55cm. A secondary viewfinder video camera (Microsoft LifeCam HD-3000) was also positioned inside the stainless-steel housing in order to record constant high definition footage of each deployment directly on to the topside hard disk drive (HDD). The live video feed was displayed via a 100 m umbilical on the topside computer which also allowed for control of the stills camera using Canon EOS software. Still images were saved onto the camera microSD card and transferred to the topside computer HDD via USB interface after completion of each station.

Fins were positioned at the back of the frame for orientating with the prevailing current and a customised rubber gasket, lined with silicone grease, was bolted between the polycarbonate lens and the metal components of the CLOC to keep watertight (Jones et al., 2019). Two 3,000lm light-emitting diode (LED) lights were mounted at the bottom of the CLOC to illuminate the area of seabed directly below the CLOC and controlled using the topside system to optimise lighting.

For deployments without the CLOC, the same frame was used mirroring the set up described above. The camera housing was attached to the frame at the same elevation and in the same position providing the same field of view (Fig 2c, d).

5.2.4. Image Analysis

Camera System Comparisons

To eliminate observer bias influencing differences between the two datasets, the same individual (REJ) analysed all the images used in this analysis. Images were randomly named and randomised in order prior to analysis to avoid any station knowledge bias. Prior to conducting this study, a power and sample size analysis undertaken in Minitab 11 to derive the optimal sample size for this study identified that a minimum of 322 images were required to detect a difference of more than 0.5 taxa between CLOC and non-CLOC deployments with a power of 90%. In total 942 images were collected across the two study locations; 529 with the CLOC and 413 without the CLOC (Table 1). Image analysis was conducted using the cloud-based Bio-Image Indexing and Graphical Labelling Environment software (BIIGLE 2.0) (Langenkämper et al., 2017). Images were classified and annotated using the Marine Protected Area (MPA) v1 label tree developed by the Centre for Environment, Fisheries and Aquaculture Science (CEFAS). This label tree is based upon the Collaborative and Annotation Tools for Analysis of Marine Imagery (CATAMI) system (Althaus et al., 2015) which enables the user to "nest" higher resolution taxonomies within morphological "classes". When it was possible to identify specific taxa (to family, genus or species level), these taxa were added to the label tree, nesting them under the most appropriate morphological class. Where taxa could not be confidently identified, a second analyst with additional experience in UK benthic assemblages also reviewed the image to ensure the identification was taken to the highest classification level possible.

Still image analysis was a two-part process, with labels being attributed to the images at "Tier 1" (whole image labels – used for standard metadata information) and epibiotic abundance / percentage cover within an image ("Tier 2" labels). Tier 1 labels included: image quality, percentage seabed visible and EUNIS habitats/biotope classification (Parry, 2019). These values were categorised using a numbered criteria defined in Supplementary Information, Section A; Table 1. Percentage gravel (inclusive of boulders, cobbles, shells, granules), sand and mud were recorded and used to determine and assign EUNIS BSHs. Tier 2 annotations involved enumeration of all visible taxa using points for "count" taxa and polygons for ground-covering taxa recorded in square pixels (sqpx). SACFOR was used to assess faunal turf, cf. *Spirobranchus sp* and barnacle cover (Cirripedia) based on guidance taken from Moore et al. (2019). For the purpose of this study, 'faunal turf' referred to hydroids considered <1cm in height (Sheehan et al., 2016). Taxonomically similar species, which could not be distinguished with confidence, were grouped. These included Ascidians, red macroalgae (rhodophyta), brown macroalgae (phaeophyceae) and green macroalgae (chlorophyta).

Sabellaria alveolata Assessment

S. alveolata reef assessments where appropriate, were also undertaken for images in Swansea Bay based on the Jenkins et al. (2018) and Gubbay (2007) reef score criteria (Supplementary information, Section A; Table 2). Elevation and patchiness (percentage cover) were recorded for each image where *S. alveolata* was deemed present to determine whether structures could be considered a reef (scored as low, medium, and high). Elevation of *Sabellaria* reefs during this assessment was non-quantitatively associated with proximity of reef structure to the CLOC lens (approximately 10cm from the seabed). The visibility of tube apertures were noted during this analysis. Confidence scores were also given to each of the two parameters, elevation and patchiness attributed to *S. alveolata* in each image and were recorded as 1 (high), 0.5 (medium), or 0 (low) based on the descriptions given in Supplementary information, Section A; Table 3 derived from Griffin et al., (2020).

Multiple Investigator Assessment

To assess whether the classification of the macrofauna improves in presence of a CLOC, eight benthic scientists with expertise in benthic image assessments in UK waters were blindly given the same ten images to analyse following the same protocol used for the camera system comparisons. Images consisted of comparable CLOC and non-CLOC deployments from the same randomly selected stations across the two survey locations. Investigators used in this analysis included those actively using benthic underwater camera methods and familiar with image analysis procedures (>3 years' experience) in both industry and academia in the UK.

5.2.5. Data Analysis

Summary data are presented as means ±1 standard error (SE). Univariate statistical analysis was conducted using Minitab 11. Deviations from normality were identified prior to analysis due to the size of the data. With this in mind, only P values ≤0.01 were considered significant to minimise the risk of type I error (McDonald, 2009; Underwood, 1997). Multivariate statistical analysis for epibiota assemblage composition was undertaken in PRIMER-e v7 (Clarke & Gorley, 2007) plus PERMANOVA+ (Anderson, 2017) using a Bray-Curtis similarity index using a dummy variable of 1 to handle images with no fauna (Bray & Curtis, 1957). For the purpose of this analysis, each image still has been treated as a 'standalone replicate'.

Camera System Comparisons

After examination of the data distribution, Log10 transformations were undertaken for image quality, percentage seabed visible and EUNIS habitat classification level values. In order to control for TSS and underwater visibility levels, General Linear Models (GLM) were conducted on image quality, percentage seabed visible and EUNIS classification level identified values (Level 1-6). A non-parametric Kruskal-Wallis one-way analysis of variance on ranks was used to test whether differences in taxonomic diversity were present between the images collected using CLOC and non-CLOC methods.

Abundance and epibiota cover (sqpx) were transformed to presence / absence datasets prior to analysis to allow for one complete multivariate analysis to be undertaken. Species accumulation curves were conducted for CLOC and non-CLOC deployments to calculate the total number of taxa observed per sample size for each method (number of images) (Ugland et al., 2003). Images were permutated randomly 9999 times and the mean value of the accumulation curve over all permutations was shown by the UGE index (Canning-Clode et al., 2008; Ugland et al., 2003). A two-way permutational multivariate analysis of variance (PERMANOVA) based on 9999 unrestricted permutations of the raw data was used to test for differences in assemblage composition between images collected for deployment method and BSH type (Anderson et al., 2008). A two-way analysis of similarity percentages (SIMPER) then identified the main species recorded within images responsible for any differences identified between methods and BSH type. Faunal assemblage composition for both factors were visualised using Non-metric multidimensional scaling (nMDS) ordination plots. In the presence of a significant interaction between deployment method and BSH type, additional principle coordinates plotted in a constrained Canonical Analysis of Principal Coordinates (CAP) plot was used to visualise patterns in the data often hidden in unconstrained nMDS plots (Anderson & Willis, 2003).

Sabellaria alveolata Assessment

Statistical analysis was not performed on confidence scores assigned to reef parameters in CLOC and non-CLOC deployments as *S. alveolata* colonies were only visible in one of the non-CLOC images collected.

Multiple Investigator Assessment

Assessment methods used in this section loosely followed those described in Durden et al., (2016) previously used to compare image annotation data. To evaluate the influence of deployment method on ecological univariate diversity indices generated for multiple investigators. Shannon *H'* (e) and Simpson Index of Diversity $1\text{-}\lambda$ were calculated for each investigator for each deployment method using PRIMER-e v7. A Kruskal – Wallis one-way analysis of variance was then used to assess whether significant differences in diversity indices were present between deployment methods for multiple investigators.

'Annotation success' in this study was defined as the classification of a specimen to family level or higher across all images. It was calculated as the number of investigators which similarly identified an organism as fraction of the total number of investigators ($n = 8$). For example, where all eight investigators detected and classified the same taxa when analysing images from the same deployment method, this was classed as 100% annotation success.

5.3. Results

In total, 66 taxa were observed across the two locations; 60 in Swansea Bay and 42 in Milford Haven using the two camera systems (Supplementary Information, Section B; Table 1). The most recorded taxa in Swansea Bay was cf. *Spirobranchus* sp. with 281 entries. The most recorded taxa in Milford Haven was rhodophyta with 115 entries.

5.3.1. Camera System Comparisons

Image Quality

Differences in image quality and the percentage seabed visible for both CLOC and non-CLOC deployments were present when compared to TSS and underwater visibility (Fig. 3 and Fig. 4a, b). Following the criteria in Supplementary Material, Section A, Table 1, the mean image quality values for CLOC deployments across the two survey areas was 4.16 ±0.04 SE with the non-CLOC deployments 2.78 ±0.06 SE. The mean percentage seabed visible values for CLOC deployments across the two survey areas was 3.71 ±0.03 SE and 2.77 ±0.06 SE for the non-CLOC deployments. For both methods, increased TSS and decreased underwater visibility had a negative effect on image quality and percentage seabed visible. Statistical comparisons between CLOC and non-CLOC deployments across varying gradients of TSS demonstrated that there was a significant effect of the CLOC in improving image quality after controlling for the effect of TSS levels ($F_{1,939}$ = 357.38, P = <0.001). Similarly, there was also a significant effect of the CLOC improving the image quality after controlling for the effect of underwater visibility ($F_{1, 939}$=438.69, P = <0.001) (Table 2). The same statistical comparisons also show there was a significant effect of the CLOC in improving the percentage seabed visible within the drop down frame after controlling for the effect of TSS ($F_{1, 939}$= 203.78, P = <0.001) and underwater visibility ($F_{1,939}$= 227.46, P = <0.001) (Table 2).

EUNIS Classification

Differences in EUNIS classification level identified for both CLOC and non-CLOC deployments were also present when compared to TSS and underwater visibility (Fig. 3). The mean EUNIS classification level identified for CLOC deployments across the two survey areas was 3.87 ±0.03 SE with the non-CLOC deployments 2.59 ±0.07 SE. For both methods, increased TSS and decreased underwater visibility had a negative effect on the EUNIS classification level identified. Statistical comparisons between CLOC and non-CLOC deployments across varying gradients of TSS show there was a significant effect of the CLOC in increasing the EUNIS classification level identified after controlling for the effect of TSS levels ($F_{1, 939}$= 178.60, P = <0.001) (Table 2). Similarly, there was also a significant effect of the CLOC in increasing the EUNIS classification level identified after controlling for the effect of underwater visibility ($F_{1,939}$= 343.43, P = <0.001) (Table 2).

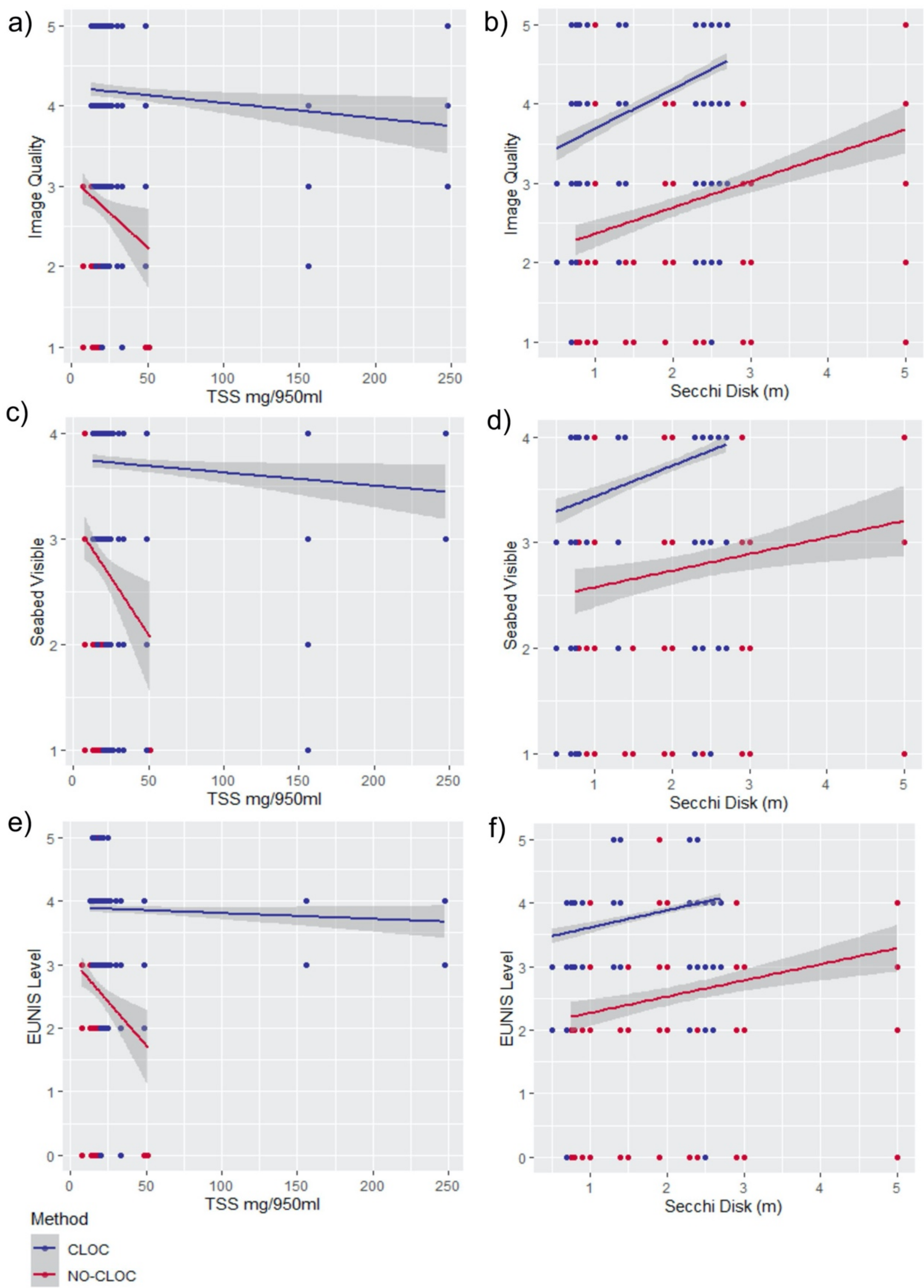

Figure 3: Image comparisons using the numbered criteria shown in Supplementary Material Section A, Table 1, between deployment methods for a) image quality values against TSS mg/950ml b) image quality values against secchi disk depth (m) c) percentage seabed visible values against TSS mg/950ml d) percentage seabed visible values against secchi disk depth (m) e) EUNIS habitat classification level identified against TSS mg/950ml f) EUNIS habitat classification level identified against secchi disk depth (m).

Table 2: *Results of the GLM testing the differences in image quality, seabed visible and EUNIS habitat level identified between CLOC and non-CLOC deployments using TSS and secchi disk readings as covariates.* ***Bold values indicate significant differences (P≤0.01).***

Source	DF	Adjusted MS	*F*	*P*
Image Quality				
TSS (mg/950ml)	1	0.0979	3.26	0.071
Deployment Method	1	10.7195	357.38	**<0.001**
Residual	939	0.0300		
Total	941			
Secchi Disk (m)	1	2.2357	80.66	**<0.001**
Deployment Method	1	12.1598	438.69	**<0.001**
Residual	939	0.0277		
Total	941			
Seabed Visible				
TSS (mg/950ml)	1	0.08976	2.42	0.120
Deployment Method	1	7.55480	203.78	**<0.001**
Residual	939	0.03707		
Total	941			
Secchi Disk (m)	1	0.9149	25.35	**<0.001**
Deployment Method	1	8.23217	227.46	**<0.001**
Residual	939	0.03619		
Total	941			
EUNIS Level				
TSS (mg/950ml)	1	0.03944	1.55	0.214
Deployment Method	1	7.56737	178.60	**<0.001**
Residual	939	0.02550		
Total	941			
Secchi Disk (m)	1	0.99761	40.75	**<0.001**
Deployment Method	1	8.0782	343.43	**<0.001**
Residual	939	0.02448		
Total	941			

Taxonomic Richness and Species Accumulation Curves

Taxonomic richness differed significantly between the two deployment methods (Supplementary material, Section B; Table 1). The CLOC deployments presented a mean of 3.12 ±0.12 SE taxa per image and the non-CLOC deployments presented a mean of 2.15 ±0.12 SE taxa per image (Kruskal Wallis: $H_1 = 36.14$, $P = <0.001$) (Fig. 4c) representing a 45% increase in the presence of a CLOC.

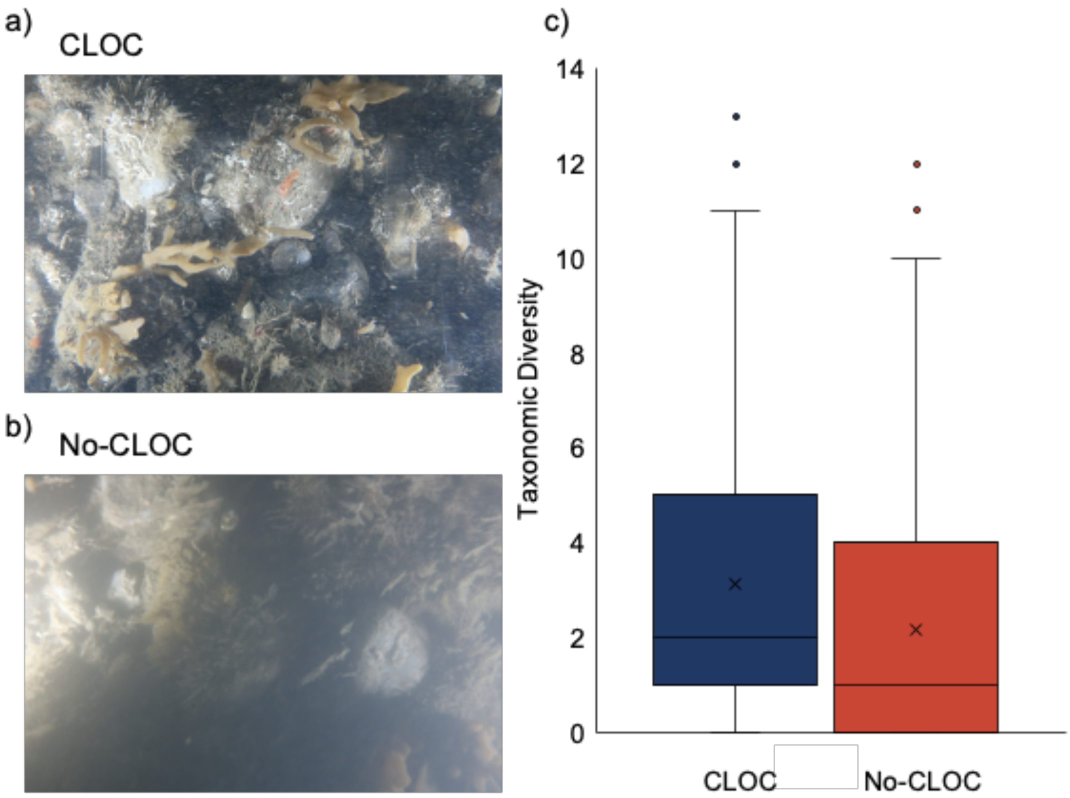

Figure 4: a) Image taken from Station 5 in Swansea Bay using the CLOC at TSS levels of 247.7g/950ml with a secchi disk reading of 0.75m b) Image taken from Station 5 in Swansea Bay without the CLOC at TSS levels of 13.5mg/950ml with a secchi disk reading of 0.75m c) Boxplot (box ranging from first to third quartile and highlighting median value, whiskers extending 1.5 the interquartile distance with points indicating outliers) showing taxonomic diversity observed during CLOC and non-CLOC deployments.

For both camera deployment methods, species count increased with increasing sampling effort (Fig. 5). The total number of species observed at 400 images was 61 for the CLOC deployment and 48 for non-CLOC deployments. At the maximum sample size for the CLOC deployments (529), the number of species observed was 63 with the maximum sample size for non-CLOC deployments (413) still remaining at 48. Taxa count therefore increased by 27% when comparing the same number of image stills collected with and without the CLOC. The CLOC was therefore considered more efficient in observing higher numbers of species in smaller sample sizes (replicates) compared to non-CLOC deployments.

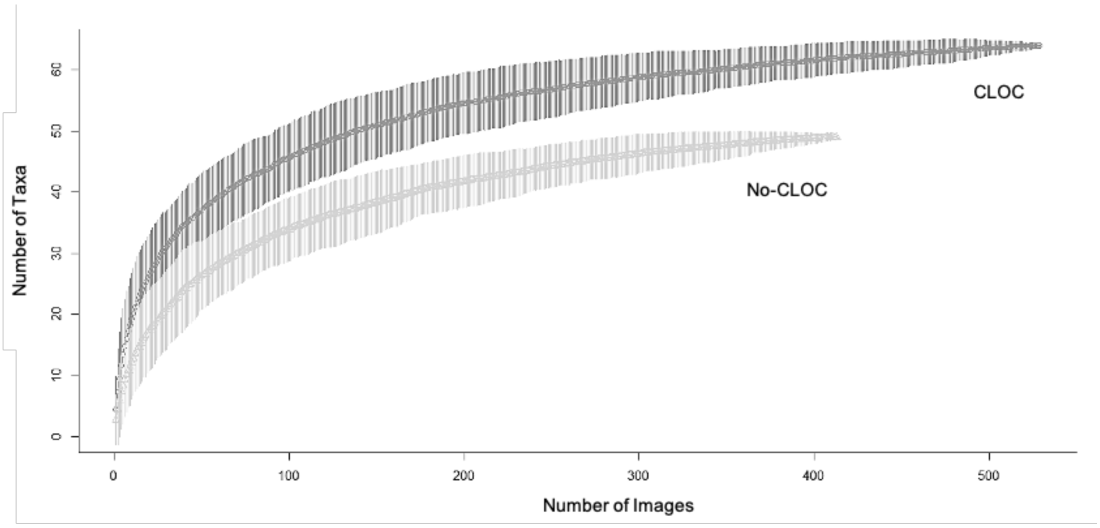

Figure 5: Species accumulation curves for CLOC and non-CLOC deployments using the UGE index (Ugland et al., 2003).

Epibiota Assemblage Composition

Epibiota assemblage composition for CLOC and non-CLOC deployments differed significantly ($F_{1,932}$ = 27.134, P = <0.001; Table 3). Although significant differences were present between the two methods, when visualising patterns on a nMDS plot (Fig. 6a) no distinct method groupings were present. Significant differences were also present between BSH type ($F_{4,932}$ = 71.149, P= <0.001; Table 3) as well as a significant interaction between deployment method and BSH type ($F_{4,928}$ = 18.249, P= <0.001; Table 3). This suggests that the ability to distinguish epibiota assemblages between BSH types is reliant on the deployment method used. When visualising assemblage composition in the nMDS plot by BSH type (Fig. 6b) and the interaction between deployment method a BSH type in the nMDS (Fig. 6c) and CAP plots (Fig. 6d), image still groupings were more distinct. With a stress level of 0.17, these plots may be considered a good representation of the similarities / dissimilarities between image stills (Clarke & Gorley, 2007).

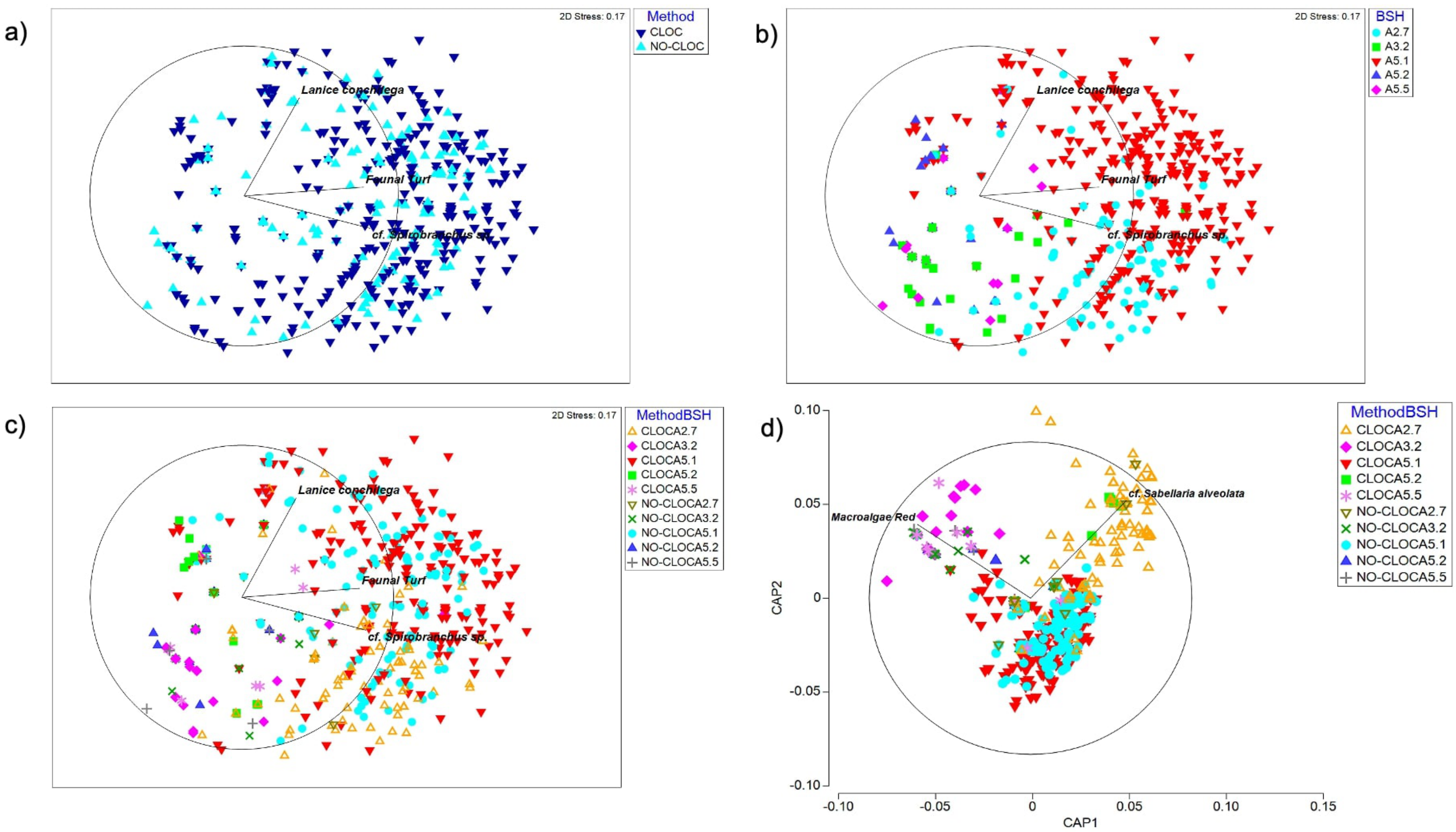

Figure 6: nMDS ordination plot showing similarities in presence/absence epibiota assemblage composition for a) deployment method b) BSH type c) interaction between deployment method and BSH type d) CAP plot showing the interaction between deployment method and BSH type. Vector lines refer to strongly correlated epibiota assemblages using the Pearson's correlation (>0.7) with the direction and length of line indicating the direction and strength of correlation.

Table 3: *PERMANOVA comparing epibiota assemblage composition for deployment method and BSH type.* ***Bold values indicate significant differences (P≤0.01).***

Source	*df*	MS	*Pseudo-F*	*P(perm)*	*Unique Perms*
Deployment Method	1	3897	27.134	**<0.001**	9958
BSH Type	4	86260	71.149	**<0.001**	9918
Method x BSH Type	4	22125	18.249	**<0.001**	9908
Residual	932	1212.4			
Total	941				

A two-way SIMPER analysis (Supplementary material, Section C; Table 1) identified higher abundances of *Lanice conchilega,* Cf. *Spirobranchus sp.* and faunal turf in CLOC deployment images as the top three contributing taxa responsible for the dissimilarity between deployment methods with a cumulative percentage of 32.15%.

Sabellaria alveolata Assessment

Stations 7, 8 15 and 16 in Swansea Bay targeted the EUNIS biotope A2.711 - *Sabellaria alveolata reefs on sand-abraded eulittoral rock*. Notable differences in reef detail were present between the two methods (Fig. 7) at these four stations with a larger number of unusable images collected during no-CLOC deployments in this biotope. Out of the 116 images taken across these four stations during CLOC deployments, 67 identified reef structures of either 'Low' or 'Medium' (58%) whilst 49 recorded 'No Reef' (42%) based on (Jenkins et al., 2018) criteria. Out of the 47 images taken across these four stations during non-CLOC deployments; one identified a reef structure of 'Low' (2%) and 46 recorded 'No Reef' (98%). All images captured during non-CLOC deployments across these four stations were assigned an image quality of poor to zero meaning *S. alveolata* features were difficult to distinguish.

Out of the 67 images with reef structures identified in imagery collected during CLOC deployments, tube apertures were visible in 62 images (93%). For the one image with reef structures identified using non-CLOC deployments, tube apertures were not visible. Confidence scores were also assigned to reef patchiness and elevation for CLOC and non-CLOC deployments. The mean confidence score assigned to reef patchiness and elevation was 0.59 ±0.05 SE and 0.26 ±0.03 SE respectively for CLOC deployments. Confidence scores assigned to patchiness and elevation in the one image recording *S alveolata* reef were both 0.00 ±0.00 SE.

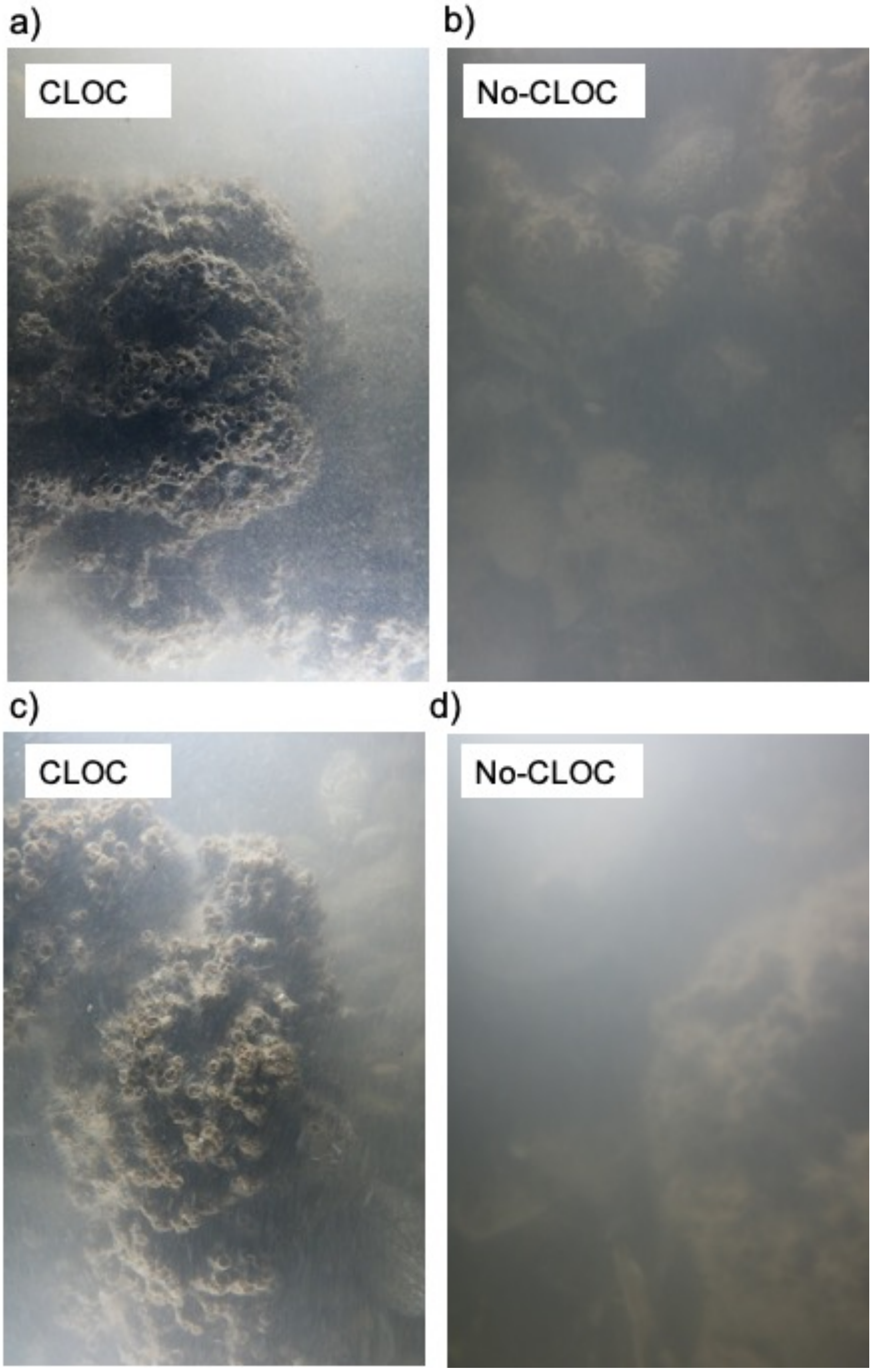

Figure 7: S. alveolata images taken at a) Station 16 with CLOC with TSS of 24.5mg/950ml and a secchi disk reading of 2.3m b) Station 16 without CLOC with TSS of 16.6 mg/950ml and secchi disk reading of 1.5m c) Station 8 with CLOC with TSS of 155.0mg/950ml and secchi disk reading of 0.75m d) Station 8 without CLOC with TSS of 51.5mg/950ml and secchi disk reading of 0.75m.

Multiple Investigator Assessment

A total of 28 taxa were detected by at least one investigator across both deployment methods. A total of 26 taxa were detected by investigators in the images collected during CLOC deployments with only 15 taxa detected by investigators in the images collected during non-CLOC deployments.

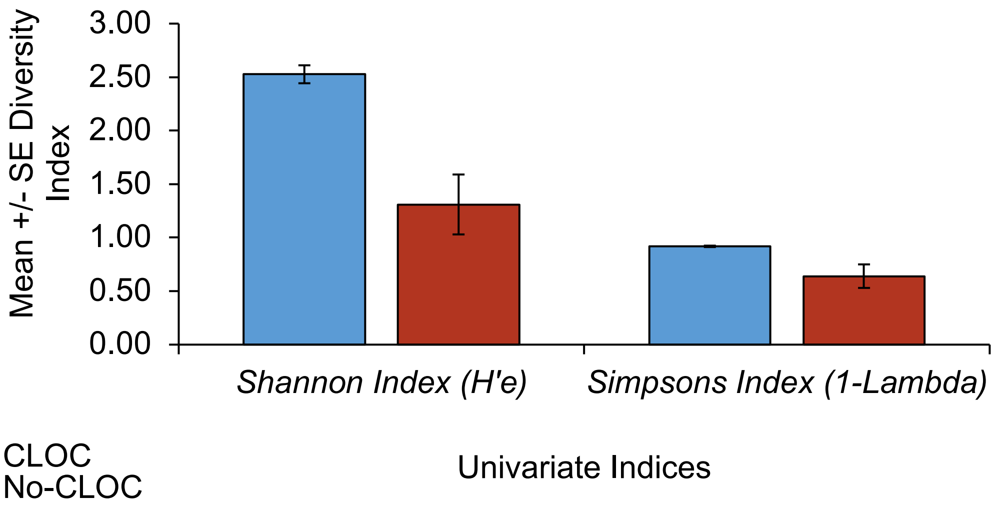

Figure 8: Mean ± Shannon and Simpson's indices calculated by deployment method for the eight investigators.

Overall, ecological diversity indices values derived from each investigator were higher for CLOC deployments compared to non-CLOC deployments. The mean Shannon and Simpsons Diversity Indices for all investigators for CLOC deployments was 2.53 ±0.08 SE and 0.92 ±0.01 SE respectively compared to 1.31 ±0.28 SE and 0.64 ±0.11 SE (Fig.8). A Kruskal – Wallis test showed the differences between methods for both diversity indices to be significant (Kruskal Wallis: $H_1 = 10.94$, $P = 0.001$).

The number of taxa which achieved a higher annotation success (>80%) increased in the images recorded using a CLOC compared to non-CLOC images (Table 4). Difficult to distinguish taxa groups including Porifera and Bryozoa as well as *Sabellaria* sp. had a lower annotation success (<40%) without the presence of a CLOC suggesting that these groups are much more difficult to detect and classify by investigators in more turbid conditions. No difference was observed in the annotation success of Cirripedia in the presence of a CLOC.

Table 4: *Annotation success for the taxa identified during the multiple investigator image analysis. Annotation success is defined as the number of investigators which similarly identified an organism fraction of the total number of investigators (n = 8).*

<40%	**40-80%**	***>80%***
CLOC		
Actinaria (Undetermined)	*Asterias rubens*	Bry_Hard_Enc_White
Alcyonidium gelatinosum	*Alcyonidium diaphanum*	Cirripedia
Bry_Hard_Enc_Brown	Macroalgae (Brown)	Hydrozoan sp.
Bry_Hard_Enc_Yellow	cf. *Sagartia* sp.	*Macropodia* sp.
Gastropod (Undetermined)		Macroalgae (Red)
Membranipora membranaxea		Sp_Cr_Enc_Orange
Macroalgae (Green)		*Sabellaria* sp.
Ochrophyta		Serpulidae
Patella vulgata		
Sp_Cr_Enc_White		
Sp_Cr_Enc_Brown		
Steromphala cineraria		
Turritellidae sp.		
Tritia sp.		
Trochidae sp.		
Urticina sp.		
No-CLOC		
Actinaria (Undetermined)	Hydrozoan sp.	Cirripedia
Alcyonidium diaphanum	Macroalgae (Red)	
Alcyonidium gelatinosum	Serpulidae	
Asterias rubens		
Bry_Hard_Enc_Brown		
Bry_Hard_Enc_Yellow		
Bry_Hard_Enc_White		
Gastropod (Undetermined)		
Macropodia sp.		
Membranipora membranaxea		
Macroalgae (Brown)		
Macroalgae (Green)		
Ochrophyta		
Patella vulgata		
Sp_Cr_Enc_Brown		
Sp_Cr_Enc_Orange		
Sp_Cr_Enc_White		
Sabellaria sp.		
cf. *Sagartia* sp.		
Steromphala cineraria		
Turritellidae sp.		
Tritia sp.		
Trochidae sp.		
Urticina sp.		

5.4. Discussion

Our findings show that the addition of a CLOC to a conventional benthic camera system significantly enhances the quality of marine biodiversity data collected under reduced visibility levels of up to 0.5m. Furthermore, the consistency in annotation success of benthic epibiotic assemblages were found to increase in the presence of a CLOC. We find that this enhancement is so great that it improves species accumulation curves, enabling less sampling to quantify species assemblages.

Camera System Comparisons

The addition of a CLOC to benthic camera deployments resulted in significant improvements in image quality, percentage seabed visible, EUNIS classification level identification and accuracy of recording taxonomic richness and epibiotic assemblage composition. Images recorded using the CLOC system also consistently recorded higher values for each univariate metric compared the non-CLOC derived images in reduced visibility environments. Improving such metrics leads to a clearer visualisation of benthic features and less ambiguity when identifying species and habitats. Differences in epibiota assemblage composition between methods were attributed to the visualisation of species such as *L. conchilega,* Cf. *Spirobranchus sp* and faunal turf commonly found in the infralittoral sedimentary habitats sampled. Required sampling effort was also reduced when using a CLOC system. Analysis showed that deployments using the CLOC were more efficient in observing higher numbers of taxa (27% increase) in smaller sample sizes compared to non-CLOC deployments. The addition of a CLOC is therefore a more cost-effective and accurate approach when sampling benthic habitats in reduced underwater visibility. If fewer numbers of deployments are required, less time in the field is needed to provide a clear and accurate representation of an area.

Sabellaria Alveolata Assessment

The visualisation of *S. alveolata* reefs in images recorded using the CLOC was also greatly enhanced when compared to those collected during non-CLOC deployments. These reefs are considered as being of high conservation importance and are protected under a range of national and international legislation (Gubbay, 2007; Jenkins et al., 2018). Biogenic reefs provide microhabitats for other organisms leading to higher levels of biodiversity when compared to their surrounding environment (Jonsson et al., 2004; Limpenny et al., 2010). They are also considered an important provider of ecosystem services including nursery areas for juvenile fish (Lefcheck et al., 2019).

Acquiring robust evidence and details of key attributes of biogenic reefs including extent, elevation and patchiness with a higher confidence is essential when undertaking benthic habitat assessments in these environments. It allows for a better determination of reef health and structure enabling application in the future monitoring and management of such biogenic habitats. CLOCs are commonly used on biogenic reef habitats associated with high sediment loading with a plentiful supply of sand and shell particles for tube building (Kirtley & Tanner, 1968). The CLOC method used in this study may be used for both *S. alveolata* and *S. spinulosa* reefs as well as other protected habitats found in turbid conditions including Annex I stony reef and sand bank habitats and both marine and freshwater mussel beds (Lindenbaum et al., 2008).

Multiple Investigator Assessment

Variation in epibiota assemblage identification between analysts is expected in large scale analyses of benthic imagery, however, minimising the magnitude of this variation and maintaining consistency is essential to limit ambiguity in data (Durden et al., 2016). If a large variability in epibiota classification is present, drastic differences in community diversity may become apparent (Gobalet, 2001). The addition of a CLOC in this survey allowed for a higher annotation success of taxa between investigators meaning a higher consistency and less variability in epibiota classification. Taxa groups including Bryozoa, Porifera and *Sabellaria* and larger individuals including Sagartiidae and *Asterias rubens* were more consistently classified in images generated from CLOC deployments. Furthermore, the ability to distinguish sensitive, protected biogenic reef habitats also increased between investigators when using a CLOC. Such improvements in data analysis allows for accurate conservation and management decisions to be made.

5.4.1. Conclusions

This evaluation of the CLOC concept shows that the application of this method greatly enhances the quality of information gathered during benthic camera surveys. By reducing the limitations of underwater visibility previously attributed to conventional underwater camera methods, this concept allows for the enhanced use of visual survey techniques for improving the conservation and management of sensitive and protected benthic habitats.

To determine whether a CLOC system should be used during a benthic survey, previous knowledge or data of the physical dynamics within an area is key. Areas subject to high amounts of suspended sediments or large tidal ranges and currents will have a decreased underwater visibility limiting the use of conventional benthic cameras (Davies-Colley & Smith, 2001; Jaffe, 2015). This study successfully acquired usable images using the CLOC in TSS values of up to 247.7 mg/950ml and underwater visibility measurements of 0.5m. However, the consistency of recording good to excellent quality images at these levels were reduced especially in the *S. alveolata* reef areas using both camera systems. We therefore recommend that CLOC systems should be implemented in aquatic environments with TSS levels of up to approximately 260 mg/L and ≥0.5m underwater visibility (secchi disk readings) depending on the survey aims and the levels of detail required.

5.5. Acknowledgements

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Supplementary Material

Section A – Image and *Sabellaria* assessment criteria

Table 1: *Image quality and seabed visible criteria adapted from Turner, Hitchin, Verling & van Rein (2016)*

Image Quality	**Number**	**Description**
Excellent	5	Image is clear and fully focussed. Colour and exposure are excellent. All levels of analysis are expected to be possible.
Good	4	Image is in focus but may be slightly over or under exposed. There may be small amounts of suspended matter. Small and cryptic taxa still be visible.
Poor	3	Some elements of the image may be in focus but other aspects such as illumination, turbidity, exposure or the angle of the camera are not ideal. Uncertain if all target objects can be accounted for. Conspicuous taxa may be enumerated but small and cryptic taxa are likely to be missed.
Very Poor	2	Image is predominantly blurred either due to suspended matter or unfocussed. Organisms are unlikely to be distinguished. Broad scale habitat may be determined in some cases.
Zero	1	No view of the seabed at all due to significant over exposure or the camera is too far from the seabed
Seabed Visible	**Number**	**Description**
76-100%	4	All levels of analysis are expected to be possible.
51-75%	3	Small and cryptic taxa still be visible.
26-50%	2	Conspicuous taxa may be enumerated but small and cryptic taxa are likely to be missed.
0-25%	1	Broad scale habitat may be determined in some cases.

Table 2: *Sabellaria reef structure matrix modified by Jenkins et al., (2018) from those initially proposed by Gubbay (2007).*

Reef Structure Matrix			**Elevation (cm)**			
			<2 Not a Reef	2 - 5 Low	5 - 10 Medium	>10 High
Patchiness (% Cover)	<10	Not a Reef	Not a Reef	Not a Reef	Not a Reef	Not a Reef
	10 - 20	Low	Not a Reef	Low	Low	Low
	20 - 30	Medium	Not a Reef	Low	Medium	Medium
	>30	High	Not a Reef	Low	Medium	High

Table 3: *Confidence score assignments for the two Sabellaria parameters patchiness and elevation adapted from Griffin et al., (2020)*

Confidence	**Score**	**Patchiness**	**Elevation**
Low	0	Limited ability to distinguish *Sabellaria* coverage.	Limited ability to determine height of *Sabellaria* above seabed.
Medium	0.5	Some ability to distinguish *Sabellaria* coverage.	Some ability to determine height of *Sabellaria* above seabed.
High	1	Full ability to distinguish *Sabellaria* coverage.	Full ability to determine height of *Sabellaria* above the seabed.

Section B – Taxa Lists for CLOC and No-CLOC Deployments

Table 1 Taxa lists and entry numbers for CLOC and No-CLOC deployments

CLOC	Entries	No-CLOC	Entries
Actinaria (Undetermined)	4	*Alcyonidium diaphanum*	23
Alcyonidium diaphanum	78	*Alcyonium digitatum*	1
Alcyonium digitatum	6	Ascidian sp.	9
Ascidian sp.	33	*Asterias rubens*	13
Asterias rubens	16	Bivalve (Undetermined)	2
Astropecten irregularis	1	Bry_Hard_Enc_Grey	46
Bacterial mats	3	Bry_Hard_Enc_Orange	2
Bivalve (Undetermined)	18	Bry_Hard_Enc_Pink	1
Bry_Hard_Enc_Grey	88	Bry_Hard_Enc_Purple	6
Bry_Hard_Enc_Orange	8	Bry_Hard_Enc_Red	6
Bry_Hard_Enc_Pink	1	Bry_Hard_Enc_White	10
Bry_Hard_Enc_Purple	10	*Cellaria fistulosa*	1
Bry_Hard_Enc_Red	6	Cirripedia	72
Bry_Hard_Enc_White	39	*Crisularia plumosa*	10
Bry_Hard_Enc_Yellow	4	*Electra pilosa*	1
Cirripedia	121	Faunal Turf (<1cm)	78
Crisularia plumosa	26	*Flustra foliacea*	2
Cnidaria (Undetermined)	4	Gastropoda (Undetermined)	14
Electra pilosa	6	*Hydrallmania falcata*	19
Faunal Turf (<1cm)	213	Hydroids (Undetermined)	7
Flustra foliacea	10	*Lanice conchilega*	73
Gastropoda (Undetermined)	19	Macroalage Brown	14
Hydrallmania falcata	73	Macroalgae Green	2
Hydroids (Undetermined)	19	Macroalgae Red	52
Lanice conchilega	149	*Ophiura ophiura*	9
Macroalage Brown	33	*Ostrea edulis*	2
Macroalgae Green	58	*Pagurus bernhardus*	6
Macroalgae Red	81	Sabellidae	3
Nemertesia antennina	2	Sp_Cr_Enc_Cream	1
Nemertesia ramosa	3	Sp_Cr_Enc_Green	13
Ophiothrix fragilis	2	Sp_Cr_Enc_Orange	13
Ophiura ophiura	34	Sp_Cr_Enc_Peach	4
Ostrea edulis	1	Sp_Cr_Enc_Red	1
Pagurus bernhardus	22	Sp_Cr_Enc_Yellow	3
Sabellidae	3	Tube worm (Undetermined)	2
Seagrasses (Undetermined)	2	*cf. Cerastoderma edule*	3
Sp_Cr_Enc_Cream	5	*cf. Crepidula fornicata*	3

Sp_Cr_Enc_Green	22	cf. *Echinocardium cordatum*	1
Sp_Cr_Enc_Orange	44	cf. Gobiidae sp.	7
Sp_Cr_Enc_Peach	16	cf. *Nucella lapillus*	1
Sp_Cr_Enc_Red	3	cf. *Sabella pavonina*	2
Sp_Cr_Enc_Yellow	5	cf. *Sabellaria alveolata*	2
Sp_M_S_Green	1	cf. *Sagartia* sp.	39
Sponge (Undetermined)	1	cf. *Spirobranchus* sp.	119
Tube worm (Undetermined)	1	cf. *Steromphala cineraria*	9
Urticina felina	2	cf. *Tritia reticulata*	2
cf. *Brachyura sp.*	2	cf. *Turritella communis*	3
cf. Buccinum undatum	5		
cf. *Cerastoderma edule*	2		
cf. Crepidula fornicata	5		
cf. Euspira nitida	8		
cf. Gobiidae sp.	5		
cf. *Macropodia* sp.	6		
cf. *Psammechinus* sp.	1		
cf. *Sabella pavonina*	4		
cf. *Sabellaria alveolata*	88		
cf. *Sagartia* sp.	64		
cf. Spirobranchus sp.	231		
cf. *Steromphala cineraria*	35		
cf. *Tritia reticulata*	19		
cf. *Tubularia indivisa*	3		
cf. *Turritella communis*	1		

Section C- SIMPER analysis results

Table 2 SIMPER analysis for species abundance and epibiota cover outlined by PERMANOVA showing the organisms which most contributed to the observed differences between CLOC and non-CLOC deployments and BSH type.

Species	Av. Abun.	Av. Abun.	Av. Diss.	Diss./SD	Contrib. %	Cum %
Deployment Method						
Av. Diss.: 82.88	**CLOC**	**Non-CLOC**				
Lanice conchilega	0.28	0.18	10.43	0.51	12.59	12.59
Cf. *Spirobranchus sp.*	0.44	0.29	8.72	0.72	10.52	23.10
Faunal Turf	0.40	0.19	7.49	0.77	9.04	32.15
Acorn Barnacles	0.23	0.18	6.22	0.60	7.51	39.65
Rhodophyta	0.15	0.13	4.46	0.27	5.38	45.04
Bry_Hard_Enc_Grey	0.17	0.11	3.2	0.62	4.61	49.64
Alcyonidium diaphanum	0.15	0.06	3.48	0.50	4.19	53.84
Cf. *Sagartia* sp.	0.12	0.09	3.44	0.51	4.16	57.99
Hydrallmania falcata	0.14	0.05	3.25	0.51	3.92	61.91
Cf. *Sabellaria alveolata*	0.17	0.00	2.30	0.22	2.78	67.67
Non-Identifiable Taxa	0.05	0.05	2.00	0.27	2.41	70.09

Species	Av. Abun	Av. Abun	Av. Diss.	Diss./SD	Contrib. %	Cum %
BSH Type						
Av. Diss.: 97.42	**A5.2**	**A5.1**				
Lanice conchilega	0.08	0.43	17.00	0.63	17.45	17.45
Cf. *Spirobranchus sp.*	0.01	0.51	11.57	0.81	11.88	29.33
Faunal Turf	0.01	0.48	9.76	0.87	10.01	39.34
Acorn Barnacles	0.00	0.30	7.76	0.50	7.97	47.31
Rhodophyta	0.11	0.03	4.35	0.31	4.47	51.78
Bry_Hard_Enc_Grey	0.00	0.26	4.26	0.58	4.37	56.15
Cf. *Sagartia* sp.	0.01	0.19	3.87	0.47	3.97	60.12
Alcyonidium diaphanum	0.00	0.19	3.85	0.43	3.95	64.07
Hydrallmania falcata	0.01	0.19	3.70	0.46	3.80	67.87
Ophiura ophiura	0.03	0.08	3.58	0.29	3.67	71.55
Av. Diss.: 93.59	**A5.2**	**A3.2**				
Rhodophyta	0.11	0.59	42.60	1.20	45.51	45.41
Phaeophyceae	0.04	0.20	11.31	0.61	12.08	57.60
Cf. *Spirobranchus sp.*	0.01	0.13	9.75	0.40	10.42	68.01
Lanice conchilega	0.08	0.00	5.60	0.29	5.99	74.00
Av. Diss.: 96.59	**A5.1**	**A3.2**				
Rhodophyta	0.03	0.59	16.04	0.77	16.61	16.61
Lanice conchilega	0.43	0.00	11.86	0.60	12.27	28.88
Cf. *Spirobranchus sp.*	0.51	0.13	10.69	0.72	11.07	39.95
Faunal Turf	0.48	0.03	8.14	0.84	8.43	48.38
Acorn Barnacles	0.30	0.03	6.44	0.51	6.67	55.05
Phaeophyceae	0.00	0.20	4.62	0.43	4.78	59.84

Bry_Hard_Enc_Grey	0.26	0.02	3.79	0.57	3.92	63.76
Alcyonidium diaphanum	0.19	0.01	3.25	0.43	3.36	67.12
Cf. *Sagartia sp.*	0.19	0.00	3.11	0.45	3.22	70.34
Av. Diss.: 93.34	**A5.2**	**A5.5**				
Rhodophyta	0.11	0.70	40.52	1.20	43.41	43.41
Phaeophyceae	0.04	0.25	10.92	0.62	11.70	55.11
Non-Identifiable Taxa	0.02	0.10	8.84	0.35	9.47	64.59
Faunal Turf	0.01	0.18	6.71	0.48	7.19	71.79
Av. Diss.:96.42	**A5.1**	**A5.5**				
Rhodophyta	0.03	0.70	17.83	0.86	18.49	18.49
Lanice conchilega	0.43	0.03	10.39	0.62	10.78	29.26
Faunal Turf	0.48	0.18	8.64	0.82	8.96	38.22
Cf. *Spirobranchus sp.*	0.51	0.00	8.38	0.83	8.69	46.91
Acorn Barnacles	0.30	0.00	5.47	0.51	5.67	52.59
Phaeophyceae	0.00	0.25	5.18	0.49	5.37	57.95
Non-Identifiable Taxa	0.07	0.10	4.32	0.32	4.48	62.63
Bry_Hard_Enc_Grey	0.26	0.00	2.90	0.43	3.01	68.93
Alcyonidium diaphanum	0.19	0.00	2.90	0.43	3.01	68.93
Cf. *Sagartia sp.*	0.19	0.00	2.86	0.45	2.96	71.89
Av. Diss.: 72.55	**A3.2**	**A5.5**				
Rhodophyta	0.59	0.70	26.36	0.80	36.33	36.33
Phaeophyceae	0.20	0.25	11.46	0.71	15.80	52.13
Non-Identifiable Taxa	0.03	0.10	6.21	0.33	8.56	60.69
Faunal Turf	0.03	0.18	5.40	0.47	7.44	68.13
Cf. *Spirobranchus sp.*	0.13	0.00	4.75	0.33	6.54	74.67
Av. Diss.: 98.17	**A5.2**	**A2.7**				
cf. *Sabellaria alveolata*	0.04	0.52	20.22	0.85	20.60	20.60
Cf. *Spirobranchus sp.*	0.01	0.58	19.01	1.01	19.37	39.97
Faunal Turf	0.01	0.36	9.20	0.78	9.37	49.34
Acorn Barnacles	0.00	0.33	8.34	0.72	8.50	57.84
Chlorophyta	0.00	0.27	7.02	0.61	7.15	64.98
Rhodophyta	0.11	0.02	5.39	0.32	5.49	70.47
Av. Diss.: 85.71	**A5.1**	**A2.7**				
Cf. *Spirobranchus sp.*	0.51	0.58	10.32	0.77	12.04	12.04
cf. *Sabellaria alveolata*	0.00	0.52	9.51	0.70	11.10	23.14
Lanice conchilega	0.43	0.04	9.40	0.55	10.96	34.10
Faunal Turf	0.48	0.36	8.22	0.85	9.59	43.69
Acorn Barnacles	0.30	0.33	7.41	0.65	8.64	52.33
Chlorophyta	0.00	0.27	3.91	0.53	4.56	56.89
Bry_Hard_Enc_Grey	0.26	0.07	3.51	0.45	4.09	65.14
Alcyonidium diaphanum	0.19	0.07	3.51	0.45	4.09	65.14
Cf. *Sagartia sp.*	0.19	0.09	3.42	0.50	3.99	69.12
Hydrallmania falcata	0.19	0.01	2.53	0.44	2.95	72.08
Av. Diss.: 94.94	**A3.2**	**A2.7**				
Rhodophyta	0.59	0.02	19.96	0.81	21.02	21.02
Cf. *Spirobranchus sp.*	0.13	0.58	15.55	0.84	16.38	37.40
cf. *Sabellaria alveolata*	0.00	0.52	15.55	0.84	16.38	52.72

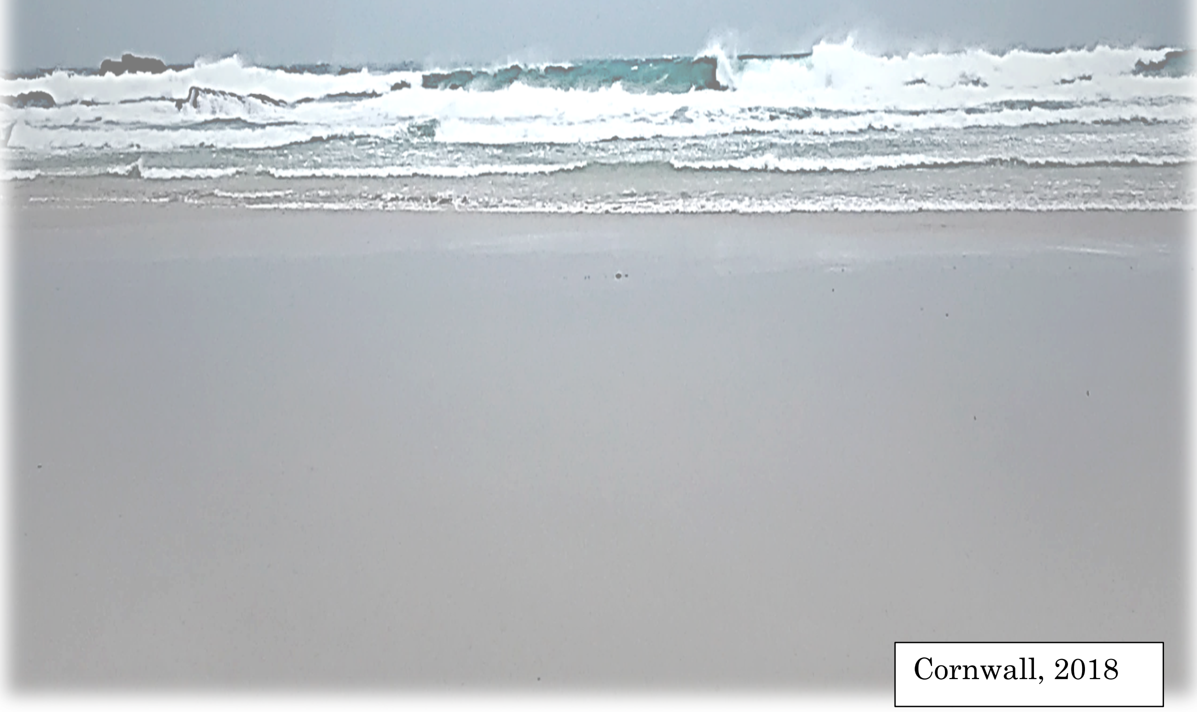
Faunal Turf	0.03	0.36	7.38	0.73	7.77	60.49
Chlorophyta	0.10	0.27	7.24	0.62	7.63	68.12
Acorn Barnacles	0.03	0.33	6.95	0.67	7.32	75.44
Av. Diss.: 97.06	**A5.5**	**A2.7**				
Rhodophyta	0.70	0.02	21.81	0.91	22.47	22.47
cf. *Sabellaria alveolata*	0.00	0.52	12.50	0.79	12.88	35.36
Cf. *Spirobranchus sp.*	0.00	0.58	12.42	0.95	12.79	48.15
Faunal Turf	0.18	0.36	8.58	0.75	8.84	56.99
Phaeophyceae	0.25	0.03	6.25	0.53	6.44	63.42
Acorn Barnacles	0.00	0.33	5.88	0.66	6.05	69.48
Chlorophyta	0.05	0.27	5.59	0.60	5.76	75.24

Chapter 6

Adaptive Resolution Imaging Sonar (ARIS) as a tool for UK marine fish identification

Status: *Submitted (Under Peer Review)*

Fisheries Research.



Cornwall, 2018

Abstract

Assessment and monitoring of marine biodiversity, including fish populations, is essential for evidence-based conservation management of coastal marine resources. The effectiveness of monitoring techniques for stock assessment varies with sea conditions. In dynamic marine environments with high turbidity, such as those found in estuaries, coastal straits, fjords, and bays, traditional assessment methods include the use of destructive techniques such as trawling. Hydroacoustic sampling techniques overcome such restrictions, methods such as echosounders have commonly been used for biodiversity assessments including fish community structure, biomass, behaviour, and dynamics studies. However, hydroacoustic methods have been shown to be less reliable for species identification.

The high frequency Adaptive Resolution Imaging Sonar (ARIS) is widely used for underwater object detection and imaging. Our study investigated the suitability of ARIS 3000 for the species identification of North-East Atlantic marine species using experimental aquarium studies, field surveys and multi-investigator assessments. Aquaria results showed that 82% of species were detected by observers, of which five were identified correctly identified consistently. The remaining four species were identified correctly <67% of the time. During field surveys, a 150% higher confidence in identification was given to more morphologically distinct groups such as elasmobranchs.

Whilst our results highlight the suitability of the ARIS for accurate and repeatable identification of some of the model species used in this study, we have also shown that factors such as size and morphological traits limit the accuracy of identification for all species. We suggest that monitoring techniques combine the paired use of ARIS sonars alongside other sampling tools for assessing motile faunal communities.

Keywords: Acoustic cameras, ARIS 3000, Environmental management, Imaging sonar, Fish identification, Biodiversity assessments, Video analysis

6.1. Introduction

Comprehensive baseline assessments and monitoring of coastal biodiversity is an essential component for the conservation management of marine resources (Carstensen, 2014). It is also important for increasing our general ecological understanding of the marine environment. Such assessments must accurately reflect the state of the environment potentially affected allowing for mitigating factors to be put in place to minimise impacts (Innes et al., 2015). Underwater cameras are a non-destructive, tool for use in marine biodiversity assessments of marine fauna (Bicknell et al., 2016; Jones et al., 2019). These methods allow, where possible, for the identification of flora and fauna. Visual information provides quantitative measures of biodiversity including species richness (R), Shannon Index of Diversity (H^1), Simpson's Index of Diversity (λ) and Species Evenness (J) indices (Gray, 2000; Jones et al., 2019). Although useful in optimal conditions, underwater camera methods are limited by underwater visibility and low light conditions (Cook et al.,2019; Mallet & Pelletier, 2014). Example areas subject to reduced visibility conditions include those targeted for coastal marine renewable developments such as tidal lagoons, and offshore windfarms (Mélin & Vantrepotte, 2015; Shields et al., 2011).

Hydroacoustic sampling techniques (e.g., sonar) allow for data collection in aquatic areas of poor visibility such those found in estuaries, coastal straits, fjords, bays and other dynamic coastal environments (Gordon Jr., 1983; Moursund et al., 2003). Acoustic methods are less affected by the properties of water in which they are deployed and have been widely used in studies of fish community structure, biomass, behaviour and group dynamic studies (Becker et al., 2011; Becker et al., 2017; Handegard et al., 2012; Jurvelius et al., 2011; Kimura & Lemberg, 1981; Martignac et al., 2015). However, research relating to the use of acoustic techniques for aquatic faunal identification often describe the difficulties faced (Charef et al., 2010; Lefeuvre et al., 2000; Scalabrin et al., 1996) with some research concluding that the use of sound alone is not an adequate tool for accurate identification (Horne, 2000).

Dual-frequency Identification Sonar (DIDSON) (Sound Metrics Corp., Lake Forest Park, WA, USA) (Belcher et al., 2002) was developed as a marine surveying tool in 2002; a big leap in the development of a new generation of hydroacoustic devices commonly known as acoustic cameras (henceforth referred to as imaging sonar) (Martignac et al., 2015). Compared to earlier models, this imaging sonar provided superior high-resolution (1.8 MHz) images of fish including identification of morphological features such as skin and fins (Belcher et al., 2002). DIDSON has been used for a variety of purposes including fish identification via acoustic shadows (Langkau et al., 2012), quantification of marine fauna (Han & Uye, 2009; Holmes et al., 2006; Maxwell & Gove, 2007), identification of migratory fish (Martignac et al., 2015), fish length measurements (Burwen et al., 2010; Han et al., 2009) and assessments of fish behaviour (Boswell et al., 2008; Doehring et al., 2011; Grote et al., 2014). Many studies using these methods in the field have focused on the assessment of known fish populations i.e. assessors have a target species they are looking to record such as migratory fish moving unidirectionally through the water column (Martignac et al., 2015). However, few studies have utilised DIDSON for generically characterising species within aquatic areas of interest (Able et al., 2014). This is likely because most research has concluded that accurate species identification using imaging sonar is still challenging.

The new generation of acoustic cameras include Adaptive Resolution Imaging Sonar (ARIS) (manufactured by Sound Metrics Corp, WA, USA), which operate at higher frequencies (greater resolution) compared to DIDSON. Recent underwater research utilising ARIS for biogenic reef assessments (Griffin et al., 2020, Appendix I), automated image processing (Shahrestani et al., 2017), validation of fish length estimates (Cook et al., 2019) and comparisons of fish assemblages to net-based sampling methods (Egg et al., 2018). Currently, there is no published literature assessing the capabilities of ARIS for accurately identifying species using both aquarium and field conditions.

This study evaluated the use of ARIS imaging sonar for the identification of motile marine fauna in aquarium and field conditions. The following objectives were investigated: (1) species-specific phenotypic traits (size) and behaviour (movement strategies; labriform, sub-carangiform and carangiform), in aquarium tank conditions. (2) Results from aquaria studies were applied to field sonar recordings taken from Swansea Bay, UK. (3) A blind multiple investigator assessment using experienced underwater image analysts was carried out on the aquaria footage to assess levels of incorrect species identification.

6.2. Materials and Methods

6.2.1. Imaging System

The ARIS 3000 Explorer imaging sonar (Fig. 1) operates at two frequencies: 3MHz detection mode for lower resolution images, but with an increased range of 15m (www.soundmetrics.com). Minimum range for both modes is 0.7m with a horizontal field of view of 30°. Images acquired when operating at 3MHz are produced from 128 acoustic beams: 0.2° horizontal (width) x 14° vertical. Objects detected at a 3m range from the sonar occupy an acoustic beam width of 10mm while objects detected at 12m occupy and acoustic beam width of 40mm (Cook et al., 2019). Beam spacing is 0.25° nominal for both operation modes with a down range resolution between 3mm and 19mm. In the current study, ARIS 3000 deployments in both aquarium and field assessments all used the 3MHz identification mode in order to acquire as detailed acoustic imagery as possible.

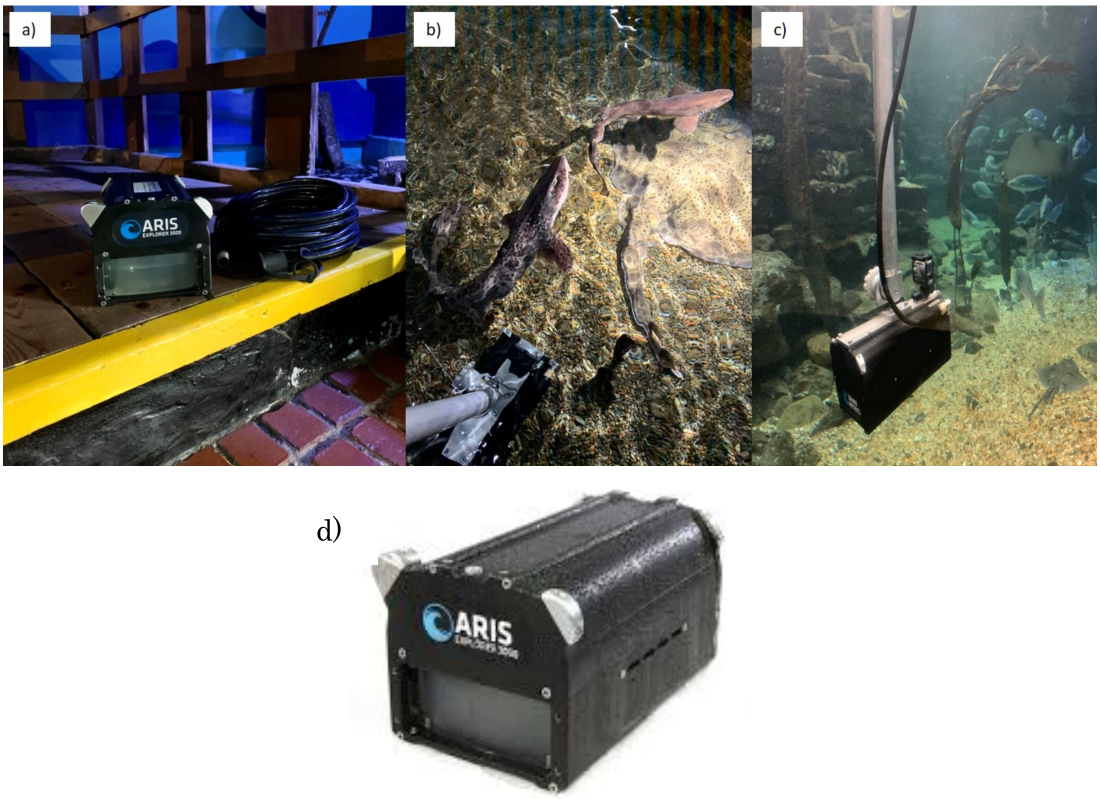


Figure 1: a) – d) ARIS 3000 Explorer imaging sonar b) in shallow Tank four c) in the deeper Tank Three during the aquaria experiments.

6.2.2. Aquarium Experimental Set Up

Aquarium experiments were undertaken at Anglesey Sea Zoo, North Wales, UK in November 2019, using live fish species. Four different tanks containing 11 different native marine species found around the coast of the UK were used during this experiment (Table 1). Tank depths ranged from approximately 1m to 3m and between 3.0m to 15.0m in length. All tanks were maintained at an approximate temperature of 12℃, salinity of 30 ppt, and pH of 7.8.

Table 1: List of species present and abundance in each of the four tanks the ARIS 3000 imaging sonar was deployed in.

Tank One (5.0m L x 2.0m W x 1.0m D)	**Tank Two (3.0m L x 1.5m W x 1.0m D)**
Turbot (*Psetta maxima*) x 1 European plaice (*Pleuronectes platessa*) x 2 Cuckoo wrasse (*Labrus mixtus*) x 2 Thick-lipped grey mullet (*Chelon labrosus*) x 15	European spiny lobster (*Palinurus elephas*) x 9
Tank Three (15.0m L x 5.0m W x 3.0m D)	**Tank Four (7.0m L x 4.0m W x 1.0m D)**
European seabass (*Dicentrachus labrax*) x 30 Gilthead seabream (*Sparus autata*) x 20 Spiny spider crab (*Maja brachydactyla*) x 2	Lesser spotted dogfish (*Scyliorhinus canicula*) x 12 Nursehound (*Scyliorhinus stellaris*) x 4 Thornback ray (*Raja clavata*) x 7

Due to the differences in tank depth, the sonar was mounted and positioned into each tank using a 1.5m custom stainless-steel pole (Fig. 2) at approximately a 0 to 20° downward facing angle. For species identification purposes, a GoPro Hero 7 White was fixed on top of the ARIS sonar with time stamps synced to ground truth species passing the ARIS. For the shallower tanks (Tank One, Two and Four), the ARIS camera was fixed in position approximately 5-10cm above the tank floor. For the deeper tank (Tank Three), the ARIS was positioned approximately 1m from the tank floor. This was due to accessibility and restrictions of using divers in the deeper tank.

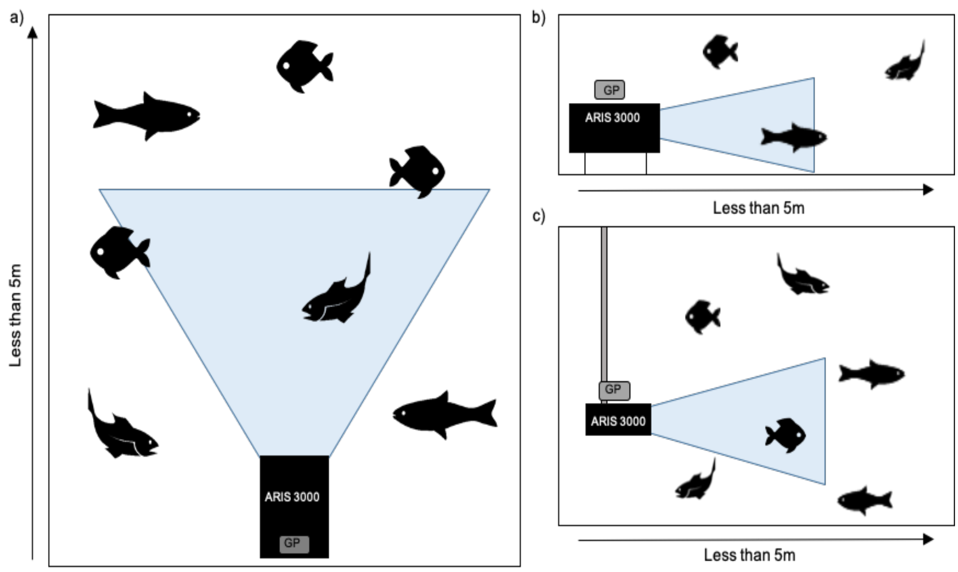


Figure 2: ARIS 3000 imaging sonar deployment set up in the various tanks used in this assessment. In shallower waters (<2m), a-b) ARIS was placed in a fixed position 5-10cm from the tank floor. In the c) deeper tank (>2m), the ARIS camera was lowered 1m from the tank floor. GP = GoPro.

6.2.3. Field Experimental Set Up

During October 2018, ten 1-hour deployments of the ARIS 3000 imaging sonar were undertaken in Swansea Bay, South Wales, UK using a 33ft vessel, 1-ton winch and stern A frame. Following the protocol described in Griffin et al. (2020), the ARIS 3000 imaging sonar was mounted on to a bespoke seabed frame attached to stainless steel mounting plate. Three mounting brackets were arranged horizontally with skids added to the base of the frame for stability. The frame was approximately 2m in height with a width and length of 1.5m and 1.75m respectively and weighed approximated 100kg. The camera was positioned at a height of 1.5m with a slight downward angle looking forwards at the seabed. A GoPro Hero 4 was also fixed on top of the ARIS sonar with time stamps synced to aid the ground truthing of species passing the field of view.

6.2.4. Data Analysis

All ARIS video files were viewed frame by frame using ARISFish (v2.6.3, Sound Metric Corp, WA, USA) and corrected using the platform motion filter tool where necessary. All analyses in both the aquarium and the field (except for the multiple investigator assessment) were undertaken by a single analyst (REJ) who was experienced and competent with marine fish species identification.

6.2.4.1. Aquarium Assessment

ARIS files were trimmed prior to analysis to ensure that the specimens present in each file were of a single known species based on the GoPro footage simultaneously recorded. Visual observations were taken for individuals entering the ARIS field of view. Observations were recorded for phenotypic features commonly used in species identification in underwater video analysis. A presence / absence checklist of features of each species was used during this process. These features were dorsal fins, pectoral fins, caudal fins (tail for ray species), anal fins, body shape and other appendages including antennae where applicable. Additionally, the ability to visualise swimming / locomotion movements were also recorded.

Measurements of individuals present in each trimmed ARIS acoustic video file analysed during the visual assessment were recorded to see whether individual size influenced the ability to distinguish identifying features. Ecograms for each trimmed ARIS file were created using the ARISFish software to gain a visual representation of the ARIS image and compressed to a vertical line of pixels for each image frame. Following the Cook et al. (2019) protocol, each fish was then marked with total length measurements recorded using the 'measure mode' feature in ARISFish. When analysing image sequences, organisms were selected from a single frame which clearly presented the full length of the organism. No additional data manipulation such as zooming into the image was undertaken. All fish were measured from the tip of the snout to the tip of the of the caudal fin (or tail for rays). Crustaceans were measured by carapace length.

6.2.4.2. Field Assessment

Observation data from the aquarium experiments was used to analyse sonar images and identify fish species from the ten field deployments. Following a similar criteria to Griffin et al. (2020) previously used to assess ARIS footage, identification confidence levels were valued high (1) medium (0.5) and low (0). A high confidence level was applied to individuals recorded on the ARIS footage where full ability to distinguish all features were present. A medium confidence level was applied to individuals recorded where some ability to distinguish features was present. Any species present on the ARIS footage where more than one feature, but less than all corresponding features for a species, were given this confidence level. A low confidence level was applied to individuals recorded where limited ability to distinguish features was present (0-1 features visible).

Individuals recorded by the sonar were identified to the highest possible taxonomic level during this process.

6.2.4.3. Multi Investigator Analysis

Video extracts of known species from the four tanks in the aquarium recorded using the ARIS imaging sonar were blindly given to three individual investigators familiar with current underwater video analysis procedures for monitoring motile fauna. Analysis focused on the consistency in faunal species identification between individuals.

6.2.5. Statistical Analysis

Regression analysis was conducted using RStudio (R version 4.0.0). Means are presented ± 1 standard error (SE) with only P values ≤ 0.01 considered significant to reduce the risk of Type II error due to the small sample sizes.

6.2.5.1. Aquarium Experiment

With assumptions of normal distribution for all data violated, a generalized linear model *glm()* using a binomial regression *(Link = "Logit")* was conducted during the size assessment to assess the effect of faunal assemblage body size (cm) on the proportion of features visible during the aquarium experiment. Sample sizes (number of images analysed) for the tanks are as follows: Tank One (n = 40), Tank Two (n = 12), Tank Three (n = 39) and Tank Four (n = 74).

6.2.5.2. Field Experiment

For the field application assessment, confidence scores were generated for each faunal individual observed in the acoustic imagery. Normality was not assumed prior to analysis comparing faunal type confidence scores. A non-parametric Kruskal-Wallis one-way analysis of variance on ranks *kruskal.test()* was therefore conducted to test whether differences were present for confidence scores generated between finfish and elasmobranch species. Sample size for the field deployments were n = 175.

6.2.5.3. Multi Investigator

For the multi investigator assessment, the classification success of species across identical video subsets taken from the aquarium deployments analysed by three investigators was calculated as the number of investigators which similarly identified species in the video as fraction of the total number of investigators (n = 3). For example, where all three investigators classified the same faunal assemblage when analysing video from the same tank, this was classed as 100% classification success.

6.3. Results

6.3.1. Aquarium Assessment

6.3.1.1. Species Identification

Observations of 11 different species under controlled conditions presented differences in the presence / absence of identifying features when visualised using the ARIS 3000 imaging sonar. Of these, six different finfish, three elasmobranch and two crustacean species were observed during the aquarium experiments.

The features observed for six finfish species were the presence and/or absence of fin types, body shape and swimming movements (Fig 3, Table 2). Out of the 75 images observed from these six species, the ability to see the dorsal fin occurred for 32 images (43%), pectoral fin 28 images (37%), caudal fin 67 images (89%) and anal fin 7 images (9%). Observations of body shape and swimming movements by finfish species were higher with 62 (83%) and 72 images (96%) respectively.

European plaice (Fig. 3b) had the smallest number of identifying features observed with the ability to only see the caudal fin and body shape across all images analysed for this species (Table 2). Swimming type for this species was also observed sporadically especially when the species remained stationary for a long period of time. Similarly, when visualising Turbot (Fig. 3f), swimming movements were undetectable as the individuals remained in a fixed position for the duration of the survey making it difficult to distinguish. European seabass (Fig. 3c) had the largest number of identifying features observed with the ability to see 100% of features assessed. With the exception of the anal fin, observations of Thick-lipped mullet (Fig. 3e) images using the ARIS imaging sonar also presented all identifying features.

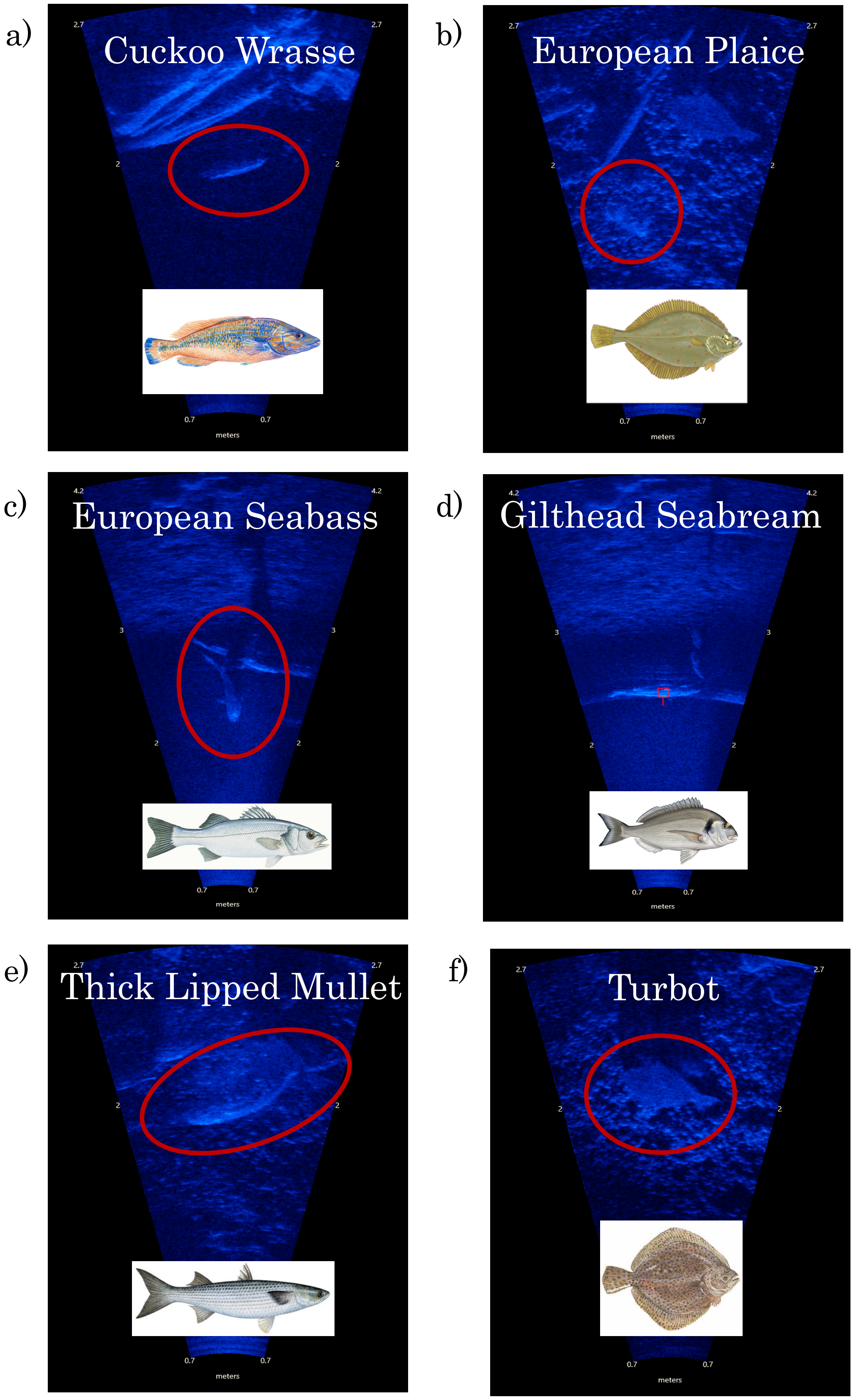


Figure 3: Images captured using ARIS 3000 imaging sonar in a shallow aquarium environment of a) Cuckoo wrasse (Tank One) b) European plaice (Tank One) c) European seabass (Tank Three) d) Gilthead bream (Tank Three) e) Thick-lipped mullet (Tank One) f) Turbot (Tank One).

Table 2: Visibility of finfish identifying features when using ARIS 3000 imaging sonar in a shallow aquarium environment. N = Number of images analysed for each species. Ticks represent presence/absence.

Species	Dorsal fin	Pectoral fin	Caudal fin	Anal fin	Defined body shape	Swimming movements
Finfish						
Cuckoo wrasse *(n = 9)*		✓	✓		✓	Vertical posture - Labriform
European plaice *(n =5)*			✓		✓	Horizontal posture - Carangiform
European seabass *(n = 25)*	✓	✓	✓	✓	✓	Horizontal posture - Carangiform
Gilthead seabream *(n =10)*	✓		✓		✓	Horizontal posture - Carangiform
Thick-lipped grey mullet *(n = 24)*	✓	✓	✓		✓	Vertical posture - Carangiform
Turbot *(n =2)*	✓		✓	✓	✓	-

Compared to finfish species, the fin type, body shape, and swimming movements were more clearly visible in elasmobranchii (Fig. 4, Table 3). Out of the 74 images observed for the three elasmobranch species, the ability to see the pectoral fin occurred for all 74 images (100%), pelvic fin 51 images (69%), tail / caudal fin 72 images (97%), body shape 65 images (87%) and swimming movements for 74 images (100%). For the Thornback ray (Fig. 4c), the ability to see its ridged back occurred for 18 of the 26 images analysed (69%).

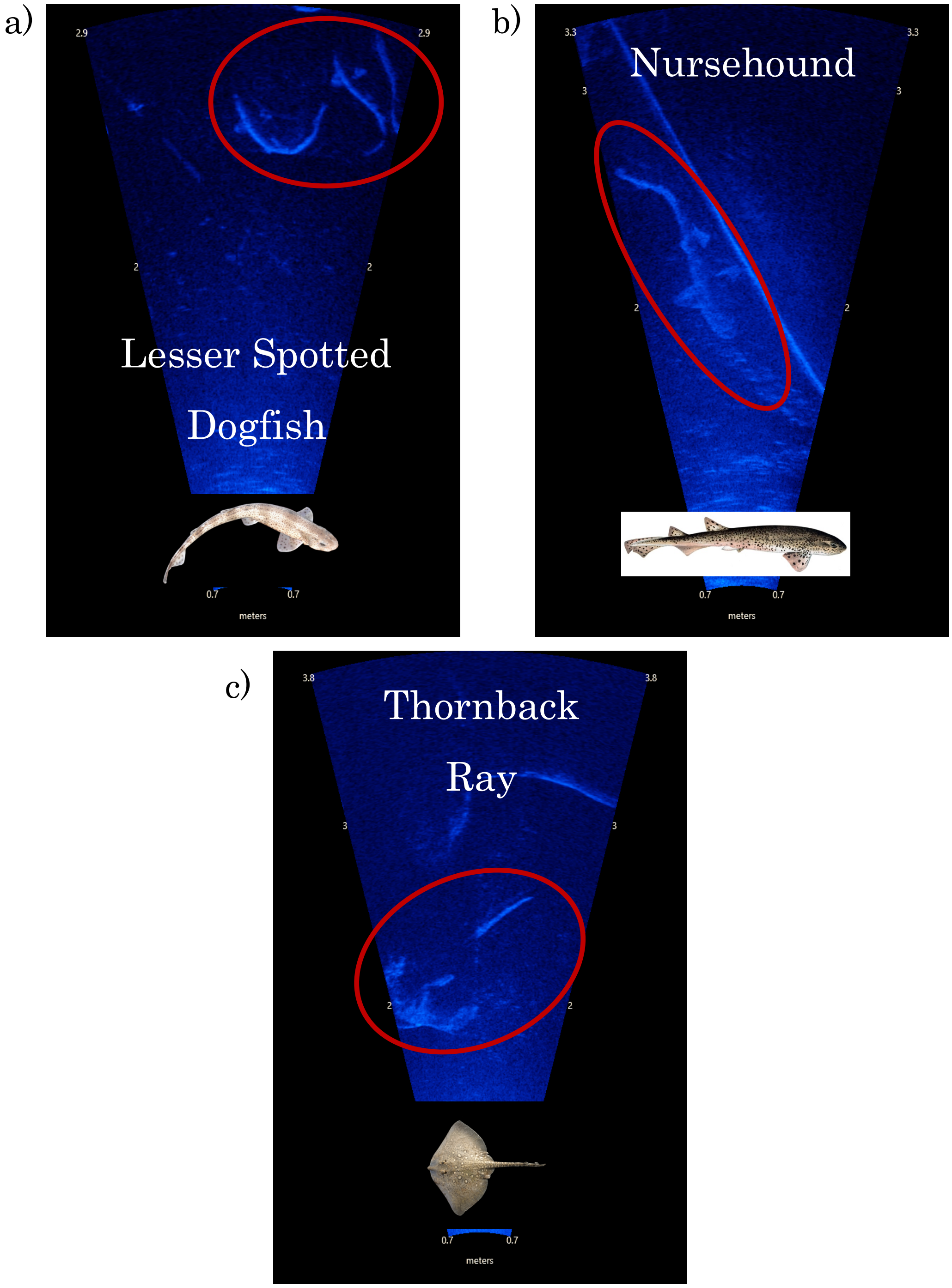


Figure 4: Images captured using ARIS 3000 imaging sonar in a shallow aquarium environment of a) Lesser spotted dogfish b) Nursehound c) Thornback ray (Tank Four)

Table 3: Visibility of elasmobranch identifying features using the ARIS 3000 imaging sonar in a shallow aquarium environment. N = Number of images analysed for each species. Ticks represent presence/absence.

Species	**Dorsal fin**	**Pectoral fin**	**Tail / Caudal fin**	**Pelvic fin**	**Anal fin**	**Ridged back**	**Defined body shape**	**Swimming movements**
Elasmobranch								
Lesser spotted dogfish *(n = 27)*	✓	✓	✓	✓	✓	N/A	✓	Vertical posture – Sub-carangiform
Nursehound *(n = 21)*	✓	✓	✓	✓	✓	N/A	✓	Vertical posture – Sub-carangiform
Thornback ray *(n = 26)*	N/A	✓	✓	✓	N/A	✓	✓	Dorsoventral Flattening-Undulatory

The three target identifying features for the spiny spider crab (Fig. 5a, Table 4) included jointed legs / limbs, defined body shape and locomotion. For the European spiny lobster (Fig. 5b, Table 4) these were antennae, body segmentation, jointed legs / limbs and defined body shape. Of the four individuals of spider crab observed on the ARIS imaging sonar, jointed legs / limbs and locomotion were visible in all images (100%) with a defined body shape visualised in 3 images (75%). For the 12 lobster images, antennae were visible in nine images (75%), segmented body in seven (58%), jointed legs / limbs for all (100%), defined body shape for 11 (92%) and locomotion visualised for all images (100%).

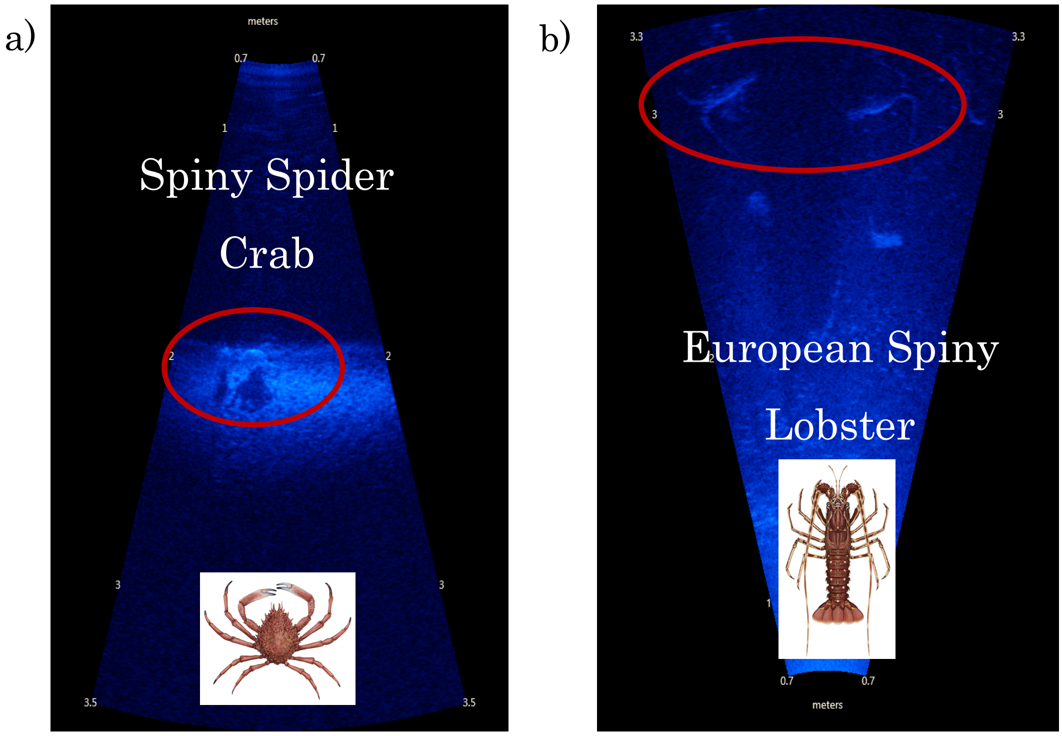


Figure 5: Images captured using ARIS 3000 imaging sonar in a shallow aquarium environment of a) Spiny spider crab (Tank Three) b) European spiny lobster (Tank Two).

Table 4: Visibility of crustacean identifying features when using ARIS 3000 imaging sonar in a shallow aquarium environment. N = Number of images analysed for each species. Ticks represent presence/absence.

Species	**Two antennae**	**Segmented body**	**Jointed legs**	**Defined body shape**	**Locomotion movements**
Crustaceans					
European spiny lobster *(n = 12)*	✓	✓	✓	✓	Walking
Spiny spider crab *(n = 4)*	N/A	N/A	✓	✓	Walking

6.3.1.2. Size

Comparisons between the proportion of features identified across all 165 images (75 finfish, 74 elasmobranch, 12 spiny lobster, and four spider crab) results from bionomial regression analyses of size and phenotypic traits, showed that as size increased, a statistically higher proportion of identifying features were visible (Estimate = 0.025, P = <0.001) (Fig. 6, Table 5). Larger species such as

Nursehound and Thornback rays had a much higher proportion of features visible compared to smaller species such as cuckoo wrasse (Fig. 6).

When adding species as a factor to this model, results show that only Spiny lobster presented a significant increase in the proportion on features visible at larger sizes (Estimate = 3.17925, P = 0.038). Species which overall tend to reach larger sizes are much more easily identifiable compared to those which on average reach a smaller body size.

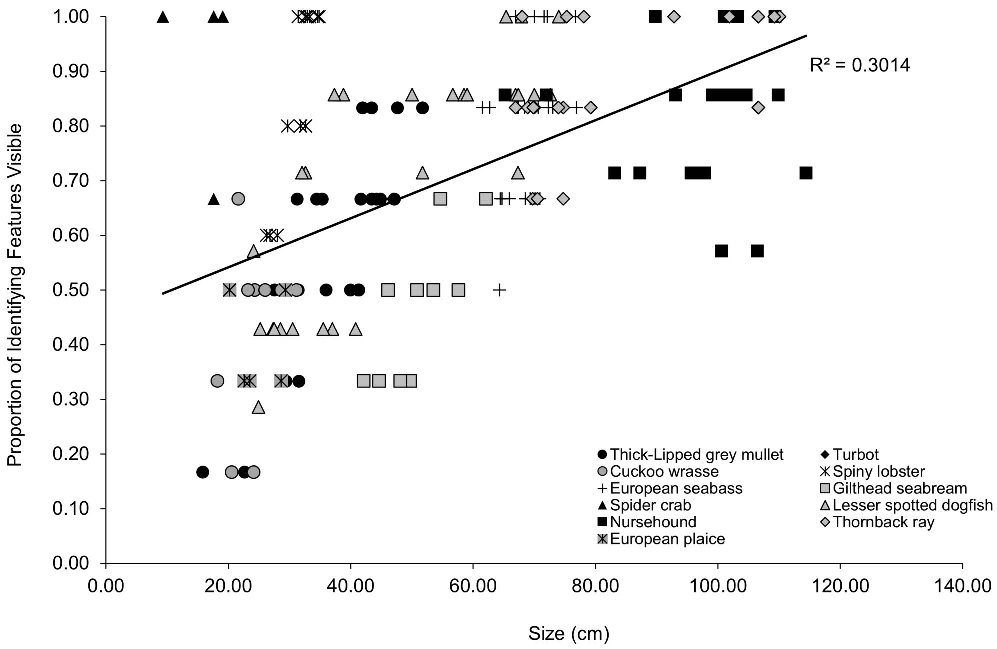


Figure 6: Proportion of identifying features visible from the 11 species recorded during aquarium ARIS 3000 imaging sonar deployments against the size of each specimen analysed. This figure presents the 165 images analysed during this assessment.

Table 5: Generalized linear model using a binomial regression showing the effect of body size (cm) on the proportion of features visible when using an ARIS 3000 imaging sonar.

	Estimate	**Std. Error**	**Z Value**	***P***
(Intercept)	-0.426991	0.408672	-1.045	0.296
Size (cm)	0.024608	0.007289	3.376	**<0.001**

6.3.2. Field Assessment

With the exception of Pleuronectiforms (flat fish), all species were taxonomically recorded to family level where possible. The identification of phenotypic feature for fin fish such as Sparidae, Gadidae, Labridae, Clupidae and Pleuronectiforms were again low (Fig. 7a) limiting further identification to species level. This in turn led to lower confidence scores in identification for taxa within this group (Fig. 7b) with a mean of 0.1 (± 0.02 SE). Furthermore, difficulties were present in identifying these individuals to species level. Smaller taxa such as Clupidae were difficult to identify with confidence as no features were visible other than size. Identification to family in this case relied upon behaviour observations including shoaling and swimming movements. In contrast to this, identifying features for individuals from the elasmobranch group were more visible in the field (Fig. 7a) allowing for these individuals to be confidently identified to species level. For example, taxa including *Mustelus mustelus*, *Scyliorhinus canicula* and *Raja clavata* were identified with a statistically higher confidence level with a mean of 0.7 (±0.04 SE) (Kruskal-Wallis: $H_1 = 79.35$, $P = <0.001$; Fig. 7b) in comparison to finfish taxa (150% difference).

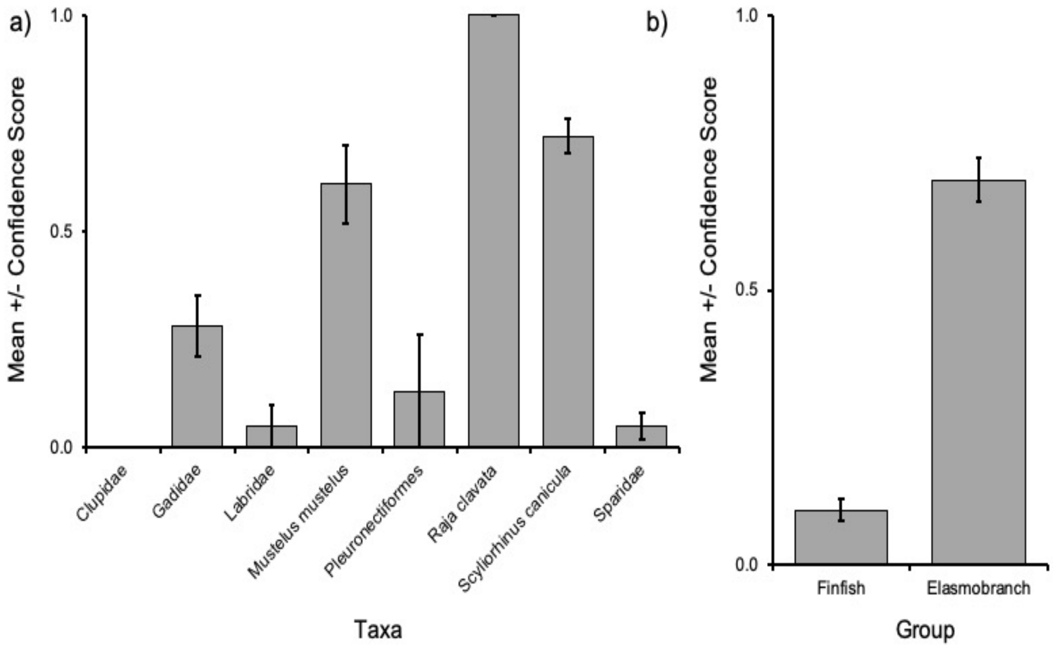


Figure 7: Mean ± 1 SE confidence scores for individuals recorded on the ARIS imaging sonar in Swansea Bay, UK by a) taxa and b) group.

6.3.3. Multi Investigator Assessment

Out of the 11 species recorded using the ARIS 3000 imaging sonar, nine were detected by at least one investigator across the four tanks. Five species were classified the same by all three investigators (100%) (Table 6), European seabass was identified by two investigators (66%) with Nursehound, Spiny spider crab and

Cuckoo wrasse identified by one investigator (33%). No investigators identified Turbot and Gilthead seabream from the footage (0%) (Table 6). It was noted by all three investigators that the ability to confidently identify individuals to species level using the ARIS 3000 footage was difficult.

Table 6: *Classification success for the taxa identified during the multiple investigator analysis. Classification success is defined as the number of investigators which similarly identified an organism fraction of the total number of investigators (n = 3).*

<40%	**40-80%**	**>80%**
Turbot	European seabass	Thick-lipped mullet
Cuckoo wrasse		European plaice
Gilthead seabream		European spiny lobster
Spider crab		Lesser spotted dogfish
Nursehound		Thornback ray

6.4. Discussion

Our study aimed to provide a unique quantitative assessment of the ability for the ARIS 3000 imaging sonar as a monitoring tool to provide species level identification of motile marine fauna specifically fish and crustaceans. Results showed that species identification is possible in some taxonomic groups and is more achievable with larger individuals. Factors such as size and morphological distinctness of the species were shown to have an influence over the confidence of classification, and this caused inconsistencies between multiple investigators when analysing sonar imagery.

6.4.1. Species Identification

Compared to identifying features such as fin type, appendages, and defined body shapes, using swimming movements and locomotion to identify some species was identified as a more reliable technique in this study. This is supported by previous studies using sonar techniques for species identification where swimming movements such as tail beat patterns (Mueller et al., 2010) and speed (Ridoux et al., 1997) have been used as a more consistent and reliable way of classifying individuals. However, swimming motions can be difficult and, in some instances, impossible to distinguish between species such as fin fish due to similarities in tail beat frequency and motion. Swimming behaviour using imaging sonar footage has not only been applied to fish populations (Able et al., 2014; Zhang et al., 2014), but also to marine megafauna such as dolphins (Francisco & Sundberg, 2019) and grey seals (Nichols et al., 2014).

In our study, the caudal fin was visible with a high level of regularity, however finfish in general were difficult to distinguish between species using fin types alone as identifying features as these were often masked from view in the sonar recordings. The size of the individuals in this case heavily influenced the ability to see identifying fin features with identification of these species relying on the simultaneous GoPro footage for ground truthing alongside swimming movements. Species which, on average, grow to larger sizes including European seabass and Thick-lipped grey mullet presented much clearer features on the sonar footage compared to smaller species such as Cuckoo wrasse or flatfish species including European plaice. This was mirrored by the field deployments where the identification confidence of finfish species remained low across the survey especially for small shoaling fish such as Clupidae and between Gadidae species. The orientation of fish in the water column may also influence the ability to visualise certain features with targets orientated at lower incident angles more likely to reflect sound to the sonar head compared to those at higher angles (Parsons et al., 2017, 2014). Langkau et al. (2012) also found differences in the visualisation of fin types when using DIDSON imaging sonar to identify fish shadows using the lower 1.8MHz frequency. In this instance, cyprinids presented a larger fin exposure within their shadows compared to salmonid species. Subsequent conclusions also suggested that species identification between species of similar shape and size are difficult to distinguish despite previous ecological knowledge (Langkau et al., 2012; Martignac et al., 2015).

In contrast to finfish species, identifying features for other taxa groups including elasmobranchs and crustaceans were much more easily visualised using the ARIS 3000. For example, the ability to visualise features including pectoral fins, caudal fins, swimming movements and defined body shape for species such as lesser-spotted dogfish and Nursehound were much higher in comparison. Furthermore, features for morphologically distinct species such as thornback rays were also clearly visualised. This was also the case for the field deployments where elasmobranch species were identified with a much higher confidence level as well as taxonomic level compared to finfish. However, we acknowledge that distinguishable features between ray species may not always be visible on ARIS footage; identification to family level here may be more appropriate. This further concludes that the successful identification of species using imaging sonar, even when applying a higher frequency of 3MHz, is still limited to morphologically distinct individuals.

6.4.2. Practical Application

Past research has concluded that imaging sonar is a powerful tool for collecting large amounts of abundance estimates and size classifications for fish populations (Cook et al., 2019; Martignac et al., 2015). In this instance, similar to its DIDSON predecessor, the ARIS 3000 imaging sonar may be applied to various habitats subject to low visibility conditions including dynamic estuarine environments, mangrove habitats, fjords, coastal lagoons, saltmarshes and other turbid riverine and coastal environments subject to agricultural run-off where traditional camera techniques are limited (Maxwell & Gove, 2007; Shahrestani et al., 2017). This tool may also be applied to sensitive habitats (Griffin et al., 2020) or seabed infrastructure (Wilber et al., 2018) where extractive techniques are restricted as well as to target species which are known to be difficult to catch using traditional net methods. Dangerous motile fauna such as crocodiles, jellyfish in some aquatic environments are also critical in some locations for restricting human underwater visual surveys; the use of imaging sonar removes this risk.

The practical application of ARIS 3000 imaging sonar for characterising and identifying unknown motile fauna in areas of low visibility is however still restricted. Rather than using traditional visual techniques used to identify individuals recorded on digital video where physical characteristics such as colour, appendage and fin features are often used, other parameters including size (Burwen et al., 2010), swimming movements (Mueller et al., 2010), speed, behaviour and schooling sizes are more appropriate for analysing imaging sonar footage (Cook et al., 2019). Our study also highlighted that ambiguity between investigators still exists when using higher frequency ARIS 3000 imaging sonars. Similar studies evaluating the use of DIDSON technology have also found inter-observer differences in enumeration and classification of fish (Keefer et al., 2017). Surveying projects spanning a wide area or monitoring time scale may consist of a substantial volume of data (tens of hours) requiring several individuals to simultaneously work on a project. These individuals may have mixed experience levels with acoustic video outputs. Slight variations in species identification between analysts is expected in these circumstances (Durden et al., 2016). However, minimising the magnitude of this variation and maintaining consistency during this time is essential to limit ambiguity in data. If a large variability in species classification is present, differences in community diversity may become apparent (Durden et al., 2016; Shafait et al., 2016). With this in mind, identification to family level may be a more accurate taxonomic rank to use, allowing for a larger identification error.

In order to reduce observer bias and gain quantitative biodiversity indices including species richness (R), Shannon Index of Diversity (H^1), Simpson's Index of Diversity (λ) and Species Evenness (J') (Gray, 2000), the addition of a second tool such as baited remote underwater video (BRUV) may be a valid option for addressing species identification issues. Recent advancements in BRUV methods using digital cameras have allowed for them to be applied to low visibility environments (Jones et al., 2019). A combination of both these visual tools for comprehensive environmental surveys may allow for both the accurate abundance assessments and size classifications provided by the imaging sonar and the identification of individuals enhanced through the use of a digital camera. Similar combination approaches have previously been implemented, with hydroacoustic methods used simultaneously with fishing nets (Guillard et al., 2004; Ransom, 1996; Romakkaniemi et al., 2000) and electrofishing techniques (Hughes & Hightower, 2015). An approach combining two methods may help inform conservation management strategies in low visibility areas.

6.4.3. Conclusion

We conclude that species identification is possible in some taxonomic groups when using the ARIS 3000 imaging sonar and is more achievable with larger individuals. However, factors such as size and morphological distinctness of the species have an influence over the confidence of classification which can cause inconsistencies between multiple investigators when analysing sonar imagery. Both reducing species identification to family level, allowing for a larger identification error, and using additional sampling tools in tandem with high resolution imaging sonar techniques may provide a more accurate portrayal of fish assemblages in an area.

6.5. Acknowledgements

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General Discussion

In the introductory literature review (Chapter 1), the limitations of traditional survey methods for assessing coastal biodiversity were identified. This provided a scope for future research through this thesis. With traditional extractive survey methods such as trawling and benthic sediment grabbing damaging to the target environment and limited by their proximity to seabed infrastructure, the use of static, non-extractive equipment such as underwater cameras, were identified as a viable alternative for monitoring coastal biodiversity. Such methods may be implemented in sensitive habitats as well as closer to marine infrastructure whilst minimising damage. However, one of the main challenges faced with regards to assessing biodiversity using underwater cameras in coastal temperate waters was reduced underwater visibility. For example, in the UK, a combination of high current velocities influenced by large tidal ranges and / or wave action as well as areas heavily influenced by large accumulations of phytoplankton may increase water turbidity; this in turn reduces underwater visibility. It is also these dynamic areas which are targets for the expansion of marine renewable developments, a key objective for many countries. This thesis has investigated the use of different underwater camera methods for improving marine biodiversity assessments in low visibility coastal environments around the UK. Such methods included baited remote underwater video techniques (BRUVs), the introduction of a clear liquid optical chambers (CLOCs) and the use of the ARIS 3000 imaging sonar.

Chapter 2 explored factors influencing the information gathered during coastal BRUV deployments in the UK. Results showed that methodological factors such as bait type (in particular oily fish meal and mackerel) and deployment duration had the most influence over response variables such as species richness/taxonomic diversity and relative abundance *(MaxN)*. Environmental factors such as image quality were also identified as having the most influence over relative abundance with poorer image qualities negatively impacting relative abundance. In terms of faunal assemblage composition, results presented significant interactions between the habitat observed during BRUV deployments and bait type, image quality, time of deployment and tide type (spring / mid / neap). Species identified as contributing most to these interactions included Gobiidae, *Scyliorhinus canicula*, Paguridae and *Merlangius merlangius.* This study emphasised the importance of methodological approaches to BRUV deployments in coastal UK waters and the environmental factors which may be affecting information gathered. Fluctuations and variations in data may be attributed to methodological inconsistencies and/or environment factors as well as over time and therefore must be considered when analysing and interpreting the data.

With Chapter 2 identifying bait type as a key factor in influencing BRUV performance, Chapter 3 investigated varying bait types commonly used in previous bait studies globally and applied them to two dynamic coastal areas in the North-Eastern Atlantic region. Bait has been found to attract individuals of different species towards the field of view of the recording camera by releasing chemical stimuli including water-soluble proteins into the surrounding water column. The inclusion of bait with underwater cameras has been shown to help with overcoming the problem of low fish counts associated with fish passing un-baited systems by chance. Results during this study found that overall, baited camera deployments attracted higher numbers of relative abundance and taxonomic diversity compared to unbaited deployments. Although no one bait type presented a statistically higher number of relative abundance and taxonomic diversity, smaller weights (50g) of mackerel and crab were found to produce similar values to larger amounts of squid (350g). Larger numbers of scavenging species were also present during mackerel deployments. It was therefore recommended from this research that for standardisation purposes, ≥50g of oily fish such as mackerel should be used for BRUV deployments in coastal areas found in the North-Eastern Atlantic region as it is a more cost-effective and readily available alternative to crab and squid.

Chapters 4 and 5 successfully evaluated the use of a CLOC in enhancing the information gathered for both motile fauna and benthic epifauna biodiversity assessments. The CLOC concept uses a body of clean freshwater in front of a camera sensor in a custom-built frame with an interface of non-concave glass or Perspex between the freshwater and seawater. This addition of clean liquid reduces the scattering of light that would otherwise occur when passing through the turbid water it displaces without limiting the field of view. During this research, a CLOC was applied to both conventional BRUV camera systems and benthic drop-down camera systems in several locations across South Wales and South-West England over varying gradients of underwater visibility. Results for the CLOC-BRUV deployments showed that the presence of a CLOC statistically enhanced the proportion of motile individuals identified to species level in areas of increased turbidity. Similarly, the addition of a CLOC to benthic drop-down deployments significantly enhanced the quality of information gathered. Images acquired using the CLOC system consistently recorded statistically higher values of image quality, percentage seabed visible, European Nature Information System habitat level identification, taxonomic richness and epibiota assemblage composition. Furthermore, it was also found that the 'annotation success' of taxa were found to increase between individual experts in the presence of a CLOC during benthic imagery analysis.

Chapter 6 and Appendix I evaluated the ARIS 3000 imaging sonar for use in motile species identification and *Sabellaria* reef assessments in low visibility environments. The ARIS 3000 uses high frequency imaging sonar to provide high definition, live *in situ* images of underwater environments subject to low or zero levels of underwater visibility. This camera can operate at two frequencies: 3MHz identification mode for higher resolution images within a 5m range and 1.8MHz detection mode for lower resolution images but with an increased range of 15m. Such systems are not restricted by underwater visibility and may therefore be considered a useful tool in the assessment in marine biodiversity in challenging environments. Evaluations of its capability in visualising distinguishing identifying features for motile fauna native to the UK showed mixed results with morphologically distinct species such as elasmobranchs much clearer in the footage compared to individuals belonging to finfish families. It was concluded that the ARIS 3000 imaging sonar is a powerful tool for collecting large amounts of abundance estimates and size classifications for fish populations in low visibility coastal environments such as saltmarshes and mangroves. However, this study identified that future improvements are required for confident and consistent fish identification using imaging sonar, especially for less morphologically distinct species. When applying the same imaging sonar system for benthic *Sabellaria* assessments, evidence suggested that the ARIS 3000 is also a useful tool for ground-truthing SSS interpretation and assessing the status of *Sabellaria* bioconstructions in low-visibility environments.

Improvements in the use of traditional digital cameras and the increased use of hydroacoustic technology allows for the increased implementation of these techniques in challenging turbid environments. A combined approach of camera methods may be most suitable for adequately assessing coastal biodiversity in low visibility environments. For example, combining imaging sonar and CLOC-BRUV systems / CLOC drop-down camera systems may provide a more accurate insight into abundance estimates and size classifications whilst also allowing for confident species identification.

Wider Research Fields

Recent improvements in computer software have further enhanced the use of underwater imagery for marine biodiversity assessments. The use of artificial intelligence for fish research has allowed for a more standardised and streamlined approach to video analysis. For example, numerous software have been developed and tested for identifying, counting and tracking fish movements (Marini, Corgnati, et al., 2018; Marini, Fanelli, et al., 2018). This standardised and streamlined approach has also been applied to benthic image analysis through software such as BIIGLE 2.0 (Langenkämper, Zurowietz, Schoening, & Nattkemper, 2017) to accommodate large data volume and rich content. In addition to this, colour enhancement (Akkaynak & Treibitz, 2019), machine learning (Mohamed, Nadaoka, & Nakamura, 2020) and computer vision (Piechaud, Hunt, Culverhouse, Foster, & Howell, 2019) techniques have also improved. Such research is a huge step in the right direction for improving the accuracy and efficiency of approaches to underwater biodiversity analysis. However some research has still identified potential limitations to these methods including biofouling, underwater turbidity and fish overcrowding (Marini, Fanelli, et al., 2018).

The method improvements identified during this thesis alongside research into various automated analysis techniques allow for the increased application of underwater camera methods by researchers, government and non-government organisations to coastal marine environments. These techniques may abe applied to the monitoring and management of sensitive conservation areas, informing important conservation decisions.

Future Research

The natural progression from the research outlined in this thesis in relation to CLOCs would be to downsize the system. Methodological testing to reduce the size and weight of the system whilst still maintaining the camera field of view would further widen its application for marine monitoring through use on smaller vessels. Furthermore, reducing its size and weight would also decrease its disturbance to the seabed when deployed. Additional tests into optimal lighting configurations would also be beneficial for this system.

This thesis, in addition to wider BRUV research, has identified that BRUVs have primarily been used to target demersal species. Further methods research into the application of midwater BRUVs to the North – Atlantic region would again be beneficial to widening the application of these tools to pelagic species.

BRUV Deployment Recommendations for the North-East Atlantic Region

This thesis has highlighted various issues associated with implementing underwater camera methods such as BRUVs in the North -East Atlantic region. Such issues include underwater visibility and method discrepancies. Below we outline key recommendations based on the results from this thesis which should be considered when using BRUV methods in coastal waters (0-30m) around the North – East Atlantic region.

Metadata

The following metadata should be recorded as a minimum when deploying BRUV methods:

- Location (including Latitude and Longitude and general survey location);
- Time of Deployment;
- Duration of Deployment (seconds or minutes);
- Depth (m);
- Tidal State (ebb or flood);
- Tide Type (spring, mid, neap);
- Bait Type; and
- Camera Type (e.g., GoPro, Canon).

The following metadata should be recorded when processing BRUV imagery

- Image quality (Excellent, Good, Poor Unusable); and
- Broad Habitats Observed.

Bait

Results from this thesis suggested that oily fish attract higher abundances and species richness (Jones et al. 2020) using minimum weights of 50g in the North-East Atlantic region. Species such as mackerel are widely available in local bait and tackle stores and are also considered a cheaper alternative to other baits such as squid and crab.

Following methods used in previous studies, best practice for bait is to defrost for at least 24 hours prior to deployments in order to generate a greater aroma and bait plume once in the water (Dorman et al., 2012). To maximise effectiveness, bait should be replenished after each deployment as an increased soak time has been found to reduce bait quality over time (Løkkeborg & Johannessen, 1992).

Bait position during this research was 60cm from the camera due to the low visibility environments targeted. This distance maximised the camera field of view whilst still keeping the bait visible within the camera footage. Depending on underwater visibility conditions, this distance may be increased.

Environmental Conditions

Results from Chapter 2 of this thesis suggested that a higher number of failed BRUV deployments occurred during spring tides where the camera system either toppled into the sediment or was subject to extreme reduced visibility. We therefore suggest that, where possible, BRUV deployments during spring tides should be avoided.

To gain a better understanding of the quality of camera footage to be expected in a deployment location, secchi disk, turbidity, or total suspended sediment measurements should be taken prior to BRUV deployments. It is advised that a CLOC-BRUV system should be considered in underwater visibility levels of 4m and below when measuring water visibility using a Secchi Disk.

Appendix 1

Supplementary Work

Effectiveness of acoustic cameras as tools for assessing biogenic structures formed by Sabellaria in highly turbid environments

Status: *Published*

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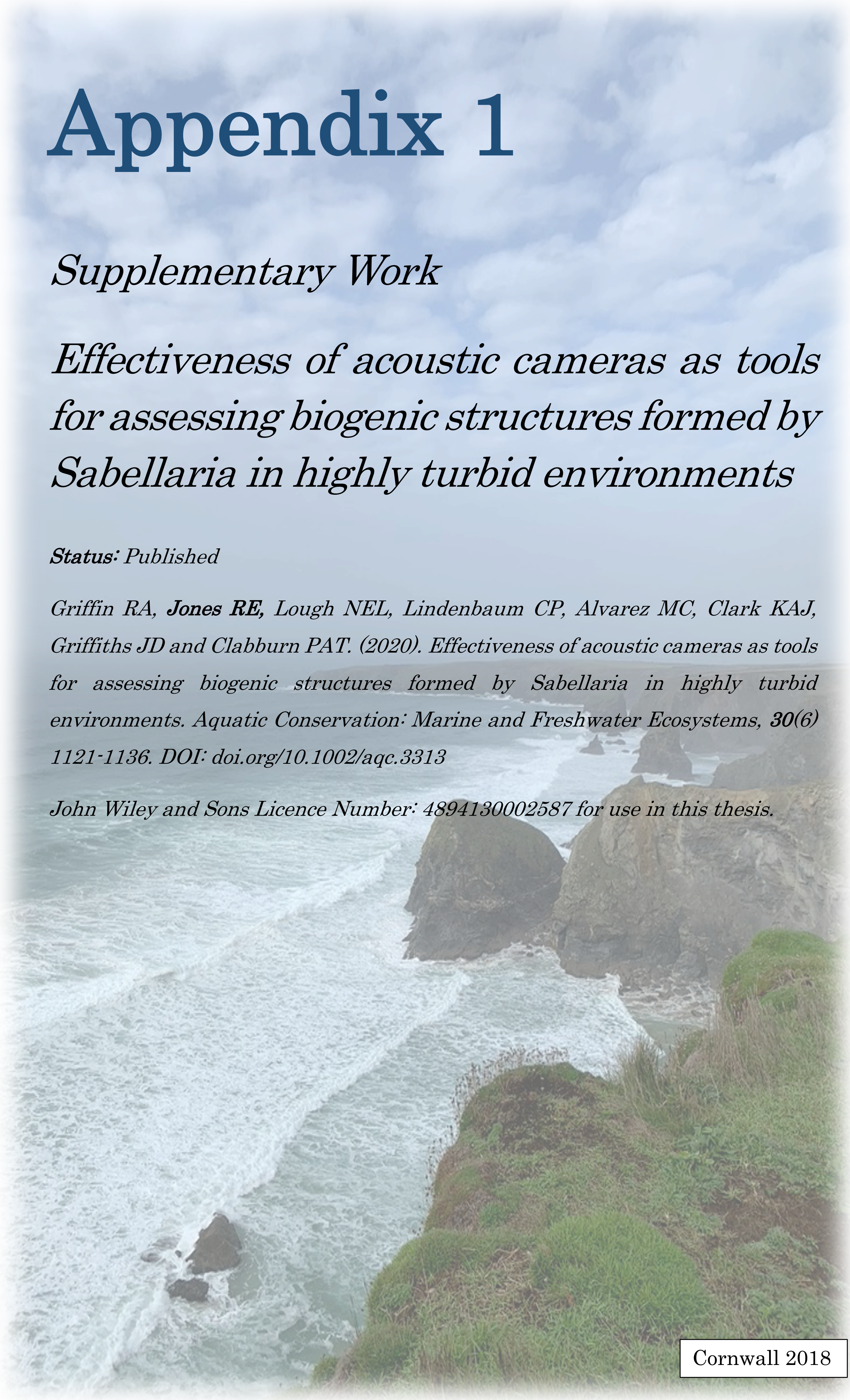


Cornwall 2018

Author Contribution

This study formed part of a collaborative venture between Natural Resources Wales (NRW), Natural England (NE), Ocean Ecology Limited and Swansea University.

The initial study concept was created by NRW and NE with the aim of establishing better methods for assessing the Severn Estuary. Data collection was led by Ross Griffin and Natasha Lough with the help of Robyn Jones and additional NRW, NE and Ocean Ecology personnel. Data analysis, writing and compilation of the manuscript was conducted by Ross Griffin and Robyn Jones. All authors critically reviewed this work and gave approval for this manuscript prior to submission.

Author	**Institution**	**% Contribution**
Ross Griffin	Ocean Ecology Limited	25
Robyn Jones	Swansea University	25
Natasha Lough	NRW	25
Charlie Lindenbaum	NRW	5
Maria Alvarez	NE	5
Katie Clarke	NE	5
Jonathon Griffiths	NRW	5
Peter Clabburn	NRW	5

All authors gave permission for me to use this publication as supplementary material to this thesis.

Ross Griffin (Lead and Corresponding Author)

Abstract

Accurately mapping the extent and status of biogenic reefs formed by polychaete worms of the genus *Sabellaria* is of conservation importance given their protected status across Europe. Traditionally, side-scan sonar (SSS) combined with ground-truthing in the form of seabed photography and videography has been widely accepted as the most suitable approach for mapping these reefs in the subtidal zone. In highly turbid environments visibility at the seabed can be near zero, however, rendering optical based ground-truthing redundant. Consequently, the true distribution and status of *Sabellaria* reefs in some shallow subtidal areas around the UK remains unclear despite their designation as Annex-I features of several Special Areas of Conservation (SACs) under the Habitats Directive. Acoustic camera imagery (ACI) collected using acoustic cameras in two deployment configurations matched well with the backscatter signatures of seabed features in corresponding SSS data. The ACI was of suitable resolution for visualizing *Sabellaria* colony structures, allowing for their Annex-I 'reef' defining attributes (extent, patchiness, and elevation) to be assessed. Colony formation 'type' was also distinguishable in the ACI, although confidence in differentiating between low-lying *Sabellaria* formations and surrounding substrates was low, particularly when using a pole-mounted configuration. This study provides a proof of concept for using acoustic cameras as tools for ground-truthing SSS interpretation and assessing the status of *Sabellaria* bioconstructions in low-visibility environments. Further development of this approach and incorporating it into statutory monitoring programmes could improve the management of the reef habitats in subtidal areas of the Severn Estuary and other highly turbid environments.

Keywords: Benthos; Estuary; Habitat mapping; New techniques: Reef: Sabellaria.

7.1. Introduction

The term 'biogenic reefs' refers to structures on the seabed created by ecosystem engineers such as corals, bivalves, polychaetes and seagrasses (Dubois et al., 2006). These bioconstructions are topographically complex, with features such as standing water, crevices and consolidated fine sediments providing microhabitats for other organisms and high levels of biodiversity relative to their surrounding habitats (Jonsson et al., 2004; Kent et al., 2017; Limpenny et al., 2010). They provide a variety of ecosystem services including nursery habitat provision for juvenile fish and invertebrates (Lefcheck et al., 2019; Rabaut et al., 2010), water filtration (Dubois et al., 2006), food provisioning (Pearce et al., 2011), coastal protection (Potts et al., 2014) and carbon sequestration (Fourqurean et al., 2012) and are therefore among the most functionally important habitats on Earth (Goldberg, 2013).

Common types of biogenic reef habitats in temperate waters include those created by *Sabellaria*, a genus of sedentary filter-feeding polychaete worms belonging to the family Sabellariidae (Johnston, 1865). Two key species are found in Europe, the Ross worm (*Sabellaria spinulosa* Leuckart, 1849) and the honeycomb worm (*Sabellaria alveolata* Linnaeus, 1767). *Sabellaria spinulosa* normally occurs in the subtidal zone with *S. alveolata* usually occurring in exposed low to mid-shore intertidal locations (Naylor & Viles, 2000) but also in the subtidal zone. Both are gregarious species and can form extensive sand and shell derived bioconstructions that cover hundreds of thousands of square metres of sea bed (Jenkins et al., 2018; Pearce et al., 2014) some of which are considered to be Europe's largest biogenic reefs (Yves Gruet, 1986). Both species are protected by a variety of European conservation legalisation and policies in their reef form, most notably as Annex I features under the Habitats Directive (Council Directive 92/43/EEC). There is continuing discussion around the qualifying characteristics required for an aggregation of *Sabellaria* worm tubes (i.e. a colony) to be classified as a 'reef' in the context of these policies (Gubbay, 2007; Hendrick & Foster-Smith, 2006; Limpenny et al., 2010) as well as continual development in approaches to assessing them (Jenkins et al., 2018). This is a particular issue for statutory nature conservation bodies in Europe who are required to report on the conditions of the Annex I reef habitat under Article 17 of the Habitats Directive. This is further confounded for *S. alveolata* as there is no single working reef definition as exists for *S. spinulosa* (Gubbay, 2007) despite the similar level of vulnerability to anthropogenic impacts. This is partially due to the high spatio-temporal variability exhibited by *S. alveolata* bioconstructions whereby they cycle through veneer, hummock and platform colony 'types' (Gruet, 1982) in response to numerous natural (e.g., cold winters and storms) and anthropogenic (e.g., trampling, shellfish farming, coastal development) disturbances (Dubois et al., 2002; Firth et al., 2015; Plicanti et al., 2016).

Another colony type found in the UK, formed by clusters of tubes growing vertically that, when coalesced, can form distinctly elevated 'clumps' aligned with seabed topographic features (e.g. the crests of mega ripples) and continuous sheets of vertically orientated clusters of tubes covering large expanses, is also described. The evolution of *S. alveolata* reef through the various structural types is thought to be linked to distinct 'growth' and 'destruction' phases (Y Gruet, 1982). It is the cycling and lack of understanding of these phases that stimulates much discussion relating to the conservation of *S. alveolata* reefs, which is further complicated when considering reefs that extend into turbid subtidal environments that are inherently difficult to study.

A variety of methods are traditionally employed for assessing the distribution, extent and condition of subtidal *Sabellaria* reef habitats. Side-scan sonar (SSS) with adequate ground-truthing methods is widely accepted as the most suitable tool (Jenkins et al., 2018). Ground-truthing is usually achieved through seabed imagery collected with a variety of methods including remotely operated vehicles (ROV) or drop-down video (DDV) systems (Limpenny et al., 2010) and is proven to be a reliable approach for assessing reefs from coastal to offshore waters (Jenkins et al., 2018; Pearce, 2014). High turbidity levels can markedly reduce the effectiveness of seabed imagery as ground-truthing information, particularly in dynamic estuarine areas that can support *Sabellaria* reef systems. There are other ground-truthing techniques such as benthic grab sampling or sediment profile imagery (SPI) camera systems (Germano et al., 2011; Solan et al., 2003) but their destructive nature makes them less than optimal alternatives (Davies et al., 2001) particularly within protected sites.

Digital Image Scanning Sonar (DISS) is an established technology based on SSS theory. These systems have primarily been used for freshwater fish assessments (Martignac et al., 2015) and in industries such as construction, oil and gas, military and law enforcement (Able et al., 2014; Moursund et al., 2003). The cameras generate acoustic images by transmitting sound pulses and converting returning echoes to produce video-like, acoustic visualizations in low visibility conditions (Langkau et al., 2012). Recent advances in this technology have resulted in the development of high-frequency acoustic imaging cameras (referred to as acoustic cameras herein) such as the range of Adaptive Resolution Imaging Sonar (ARIS) systems now widely available. These cameras operate at higher frequencies with more sub-beams than DISS systems, resulting in improved image resolution. These improvements have opened up opportunities to assess benthic habitats that could not previously be visualized with optical imagery, something which to date has not been fully trialled.

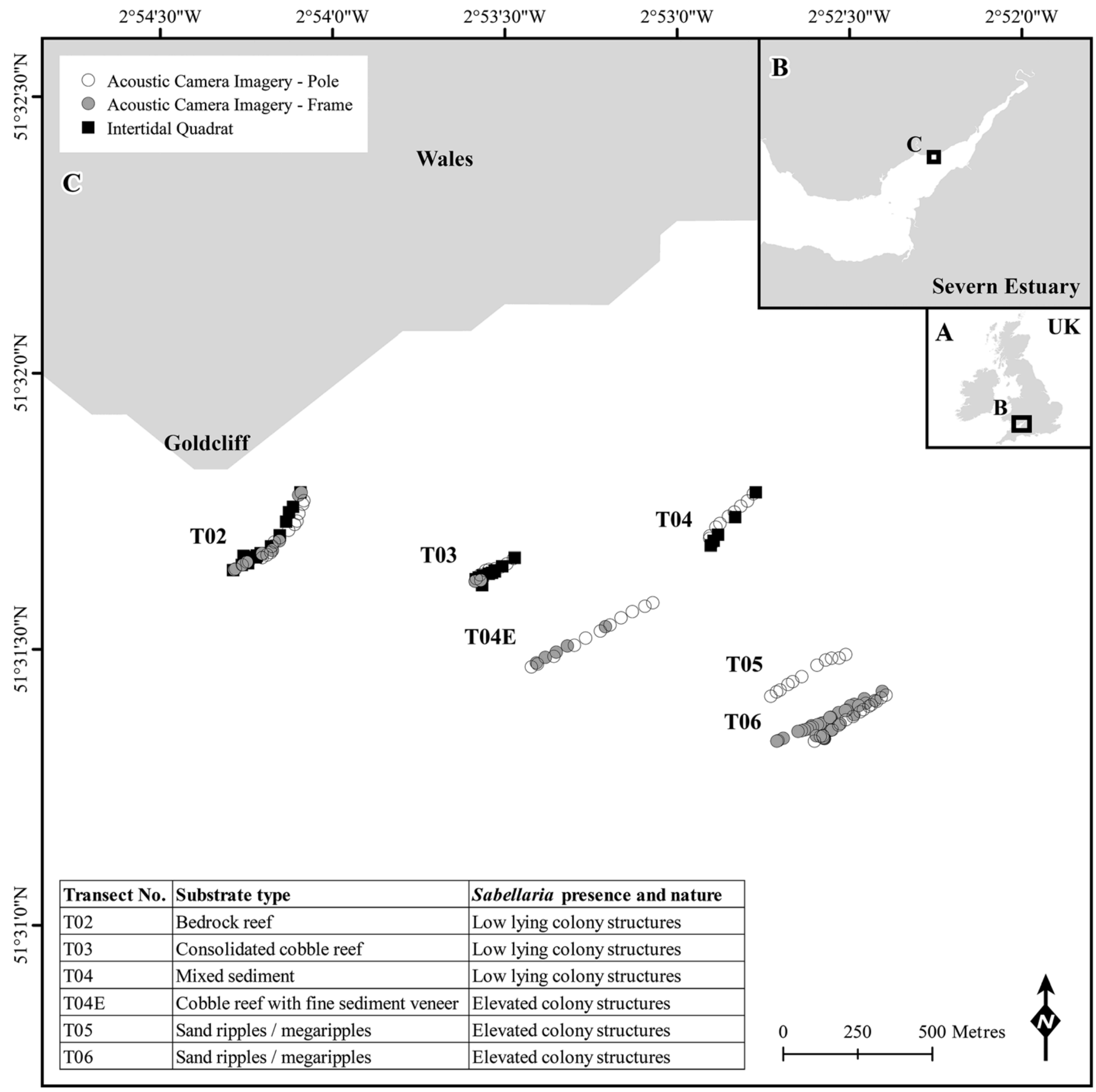
The aims of this study were to: (1) test the use of acoustic cameras for assessing *Sabellaria* colonies in a low visibility environment; (2) test different camera configurations; (3) describe the effectiveness of the technology in differentiating between areas of bare substrate and sea bed known to support reef structures; and (4) collate a catalogue of acoustic camera imagery (ACI) accompanied by corresponding SSS data, aerial imagery and intertidal photographs. This information will be used to better inform future assessments of seabed habitats, and in particular, *Sabellaria* reefs, in turbid subtidal environments in Marine Protected Areas (MPAs) throughout Europe.

7.2. Methods and Materials

7.2.1. Study Area

The Severn Estuary is a highly dynamic and unique environment that, with its funnel shape, has one of the largest tidal ranges (approaching 14 m) in the world (Xia et al., 2010) with mean ranges of 6.5 m at neaps and 12.3 m on springs (Langston et al., 2010). The estuary is influenced by river flows from its extensive catchments, as well as tides, surges and storms from the sea (Manning et al., 2010; Uncles, 2010). These conditions have led to the creation of approximately 25,000 ha of intertidal habitat such as mud and sand-flats, rocky platforms and islands harbouring plant and animal communities typical of macrotidal conditions (Natural England & Countryside Council for Wales, 2009; Uncles, 2010). The estuary is designated as a Special Area of Conservation (SAC) for the protection of a range of Annex I habitats and Annex II species, accounting for approximately 30% of the UK's Natura 2000 resource for estuaries, by area. The SAC features include 'estuaries', 'mud-flats and sand-flats not covered by seawater at low tide', 'Atlantic salt meadows', 'sandbanks which are slightly covered by seawater all the time' and 'reefs' (European Commision, 2016). The protected reef habitats refer to those formed by *S. alveolata* which are found in the intertidal and unusually in the subtidal area. The macrotidal conditions and riverine sediment load of the estuary ultimately lead to high levels of turbidity making the subtidal environment particularly difficult to monitor using underwater video and still image seabed cameras. Consequently, the knowledge of the distribution, extent and condition of subtidal *Sabellaria* reefs within the Severn Estuary is limited.

The study site at Goldcliff, South Wales (Figure 1) was selected as lower shore areas at the site are characterized by a variety of substrate types (see table insert in Figure 1). These support *S. alveolata* colonies of differing morphology ranging from single tubes to coalesced clumps of tubes formed by many individuals. The site, therefore, provided access to a range of substrates and *S. alveolata* colony types across a relatively small geographical area. Furthermore, the shore was accessible at both low and high-water periods, allowing for both visual and remote assessment of the intertidal *S. alveolata* reefs as surrogates for subtidal counterparts that can only be assessed using remote sensing techniques. Six survey transects ranging from 150 m to 500 m in length were surveyed during several field campaigns between September 2017 and October 2018 targeting key substrates and *S. alveolata* formation types known to occur across the study site (Figure 1). SSS and ACI were collected one to two hours either side of high tide periods. Aerial imagery and intertidal quadrat data were collected during low spring tide periods to maximize the area of shore that could be covered during short windows of opportunity.



Transect No.	Substrate type	*Sabellaria* presence and nature
T02	Bedrock reef	Low lying colony structures
T03	Consolidated cobble reef	Low lying colony structures
T04	Mixed sediment	Low lying colony structures
T04E	Cobble reef with fine sediment veneer	Elevated colony structures
T05	Sand ripples / megaripples	Elevated colony structures
T06	Sand ripples / megaripples	Elevated colony structures

Figure 1: Study site and transect array.

7.2.2. Intertidal Sampling

Intertidal sampling was conducted along three of the six intertidal transects (T02, T03 and T04) on the 6th October 2017. The transects were accessed via hovercraft due to the requirement for crossing expanses of shallow water to access them at low tide. It was not possible to access transects T05, T06 and T07 due to their location very low on the shore. At the start and end of each transect, digital photographs of both the transect and surrounding areas were taken, in addition to photographs of 0.25 m^2 quadrats positioned at the start of the transect. Photographs were also taken approximately every 10 m or where there were changes in sediment and/or reef type. The following features were recorded at each quadrat location; the percentage of standing water; underlying substrate; *S.*

alveolata colony formation (see Table 1); maximum colony elevation (cm) (i.e. the height of the colony from its base); and average colony elevation (cm) (n = 3).

7.2.3. Aerial Imagery

Aerial imagery was collected across all six transects during six unmanned aerial vehicle (UAV) flights conducted at low water on 8 October 2017 and 17 May 2018. The imagery was collected using a DJI Phantom 4 multi-rotor quadcopter flown at an altitude of 80 m and with 50–70% side lap and 50–70% front lap across the transects. Each pre-planned flight targeting the six transects was designed in ARCGIS 10.2. The resulting high-resolution nadir images were initially screened to remove erroneous data and were subsequently processed using DRONE DEPLOY (www.dronedeploy.com), resulting in orthomosaic outputs with resolutions ranging from 2 to 3 cm per pixel (cm/px) and route mean square error (RMSE) values of 1.2–8.5 (Table 2). The elevation profile tool available in DRONE DEPLOY was used to calculate average *Sabellaria* colony elevation measurements (n = 3) (i.e. the height of the colony from its base) within a 15 m buffer of each ACI snapshot location.

7.2.4. Side – Scan Sonar Imagery

The SSS data were collected from the RV Severn Guardian, 7–9 September 2017. A CMAX digital CM2 unit was used in conjunction with CMAX MAXVIEW acquisition software (www.cmaxsonar.com/index.html). The dual-frequency EDF CM2 towfish was operated at 325/780 kHz and a variety of ranges. The towfish was flown with an altitude of approximately 5–10% of the range setting (approximately 2–5 m above the seabed) with the tow cable scope measured using a counting pulley. This was integrated in the acquisition software MAXVIEW, where the information was displayed and embedded to the raw sonar record. Post-processing of the raw side-scan data was undertaken using SONARWIZ 7 (https://chesapeaketech.com/products/) and exported as a 4-cm resolution mosaicked GeoTiff for further analysis. The Contact Editor function available in SONARWIZ 7 was then used to measure the length of three uninterrupted Sabellaria colony shadows within a 15-m buffer of each ACI snapshot location, allowing for the trigonometric computation of elevation using the shadow length, altitude of the tow fish, and the perpendicular distance of the contract from the nadir.

7.2.5. Acoustic Camera Configuration

The ACI was collected during two separate excursions using different hardware and deployment configurations. The first was conducted upon the RV Salar Vie on 27 October 2017. An ARIS 1800 Explorer acoustic camera with a telephoto lens was mounted on a stainless-steel pole, which was deployed from the starboard side of the vessel. The camera was lowered to a point level with the keel, at a depth of approximately 0.8 m, and angled at the seabed using an ARIS Rotator AR2 at varying degrees depending on the water depth. Each transect was surveyed at a speed of 3–4 knots with data recorded and visualized using ARISCOPE 2.7 (www.soundmetrics.com).

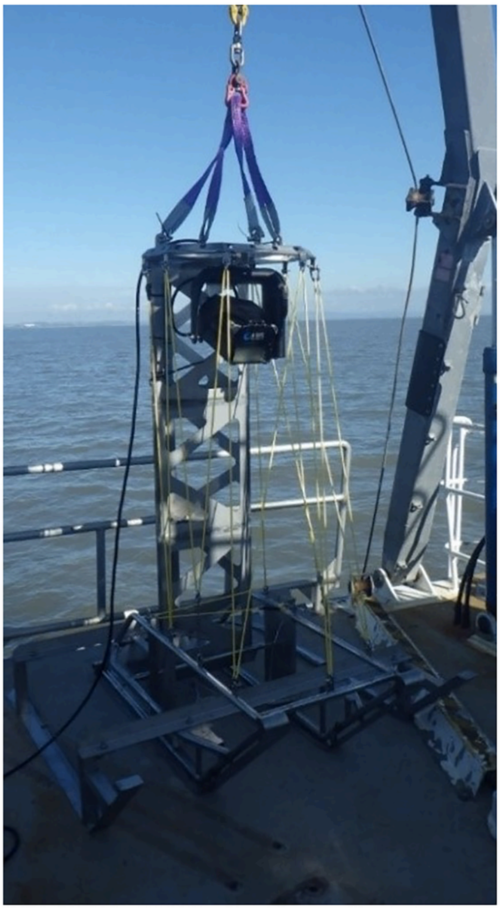


Figure 2: ARIS 300 Explorer camera mounted on bespoke seabed frame.

The second excursion was conducted from the RV Mersey Guardian, 1–3 October 2018. An ARIS 3000 Explorer acoustic camera was mounted in a bespoke seabed frame attached to a stainless-steel mounting plate. Three mounting brackets were arranged horizontally with skids added to the base for stability (Figure 2). The frame was approximately 2 m in height with a width and length of 1.5 and

1.75 m, respectively, and weighed approximately 100 kg. The camera was positioned at a height of 1.5 m with a slight downward angle, looking forwards at the seabed. Protective roping was placed around the camera to minimize damage if the frame did not land upright on the seabed.

With the dynamic nature of the study site, a tow-and-drop method was adopted along each transect, resulting in the successful collection of ACI at T02, T03, T04E, and T06. Data were recorded and visualized using ARISCOPE 2.7.

Table 1: Summary of methods and parameters used during the 2017 and 2018 campaigns.

Transect No.	ACI – Frame Mounted Configuration						ACI – Pole Mounted Configuration						SSS	UAV		Intertidal Quadrats	
	Down Range Resolution (mm)	Frame Rate (fps)	Range (m)	Beams	Frequency (MHz)	No. of Snap shots	Down Range Resolution (mm)	Frame Rate (fps)	Range (m)	Beams	Frequency (MHz)	No. of Snap shots	Resolution (cm / px)	Resolution (cm / px)	Accuracy (RMSE) (m)	Quadrat Size (m^2)	No. of Quadrats
T02	4.4 – 5.8	7.6 – 11	7.15 – 11.46	128	1.8	8	2.9	3	20.45	96	1.1	15	4	2.1	2.1	0.25	10
T03	7.3	7.7	11.48	128	1.8	3	2.9	3	20.45	96	1.1	8	4	2.1	1.6	0.25	8
T04	-	-	-	-	-	-	2.9	4.6	20.46	96	1.1	10	4	2.1	1.4	0.25	5
T04 E	7.3	7.7	11.50	128	1.8	9	2.9	3 – 4.6	20.46	96	1.1	11	4	2.1	1.2	-	-
T05	-	-	-	-	-	-	2.9	4.6 – 5.1	20.45	96	1.1	11	4	3	8.5	-	-
T06	5.8 - 7.3	7.7 – 11.6	6.19 - 11.43	128	1.8	57	2.9	4.6 – 5.1	20.45	96	1.1	13	4	3	8.5	-	-

7.2.6. Acoustic Imagery Analysis

All ARIS video files were viewed frame by frame using ARISCOPE and corrected using the platform motion filter tool, where necessary. Acoustic still images were taken at regular intervals along each transect using the snapshot function and named according to the frame from which the snapshot was taken. These snapshots were, where possible, taken in areas with at least a 15-mbuffer of homogenous substrate and/or *Sabellaria* colony type to account for the cumulative positional error of the GPS devices used for the intertidal, UAV, SSS, and ACI data collection.

The presence of *Sabellaria* reef in each acoustic still image was then determined, based on the criteria used to define subtidal *S. spinulosa* reef (Gubbay, 2007): a Sabellaria colony elevated by at least 2 cm from the underlying substrate, covering at least 10% of an area of 25 m^2. As a means of establishing whether colonies of *Sabellaria* visible in the images fitted this description, each acoustic still image was graded for extent, patchiness, and elevation. The extent was assessed through a combination of interpretation of corresponding SSS data (within a buffer of 15 m from each still image to account for positional inaccuracies) and consideration of the acoustic video footage before and after each snapshot was taken. Where present in the stills, *Sabellaria* colonies were graded as covering an area of either less than or greater than 25 m_2. Patchiness was assessed by overlaying a geometry-based grid on each acoustic image within the ARISCOPE software to assist in assigning a percentage cover category of <10%, 10–20%, 20–30%, or >30%. Basic trigonometry was used to determine the elevation using the height of the camera from the seabed and lengths of the acoustic shadows cast by the colony structures measured with ARISCOPE. Elevation was calculated through trigonometric computations for three random points in the first (closest to the camera), second, and third segments of the field of view to calculate an average colony height per still image.

For images where the Sabellaria colony extent was deemed to exceed 25 m^2, patchiness and elevation grades were combined to determine reef status (scored as low, medium, and high), in line with the reef structure matrix proposed by Jenkins et al. (2018) (Table 3). The confidence in the category assigned for each reef parameter was recorded as 1 (high), 0.5 (medium), or 0 (low), based on the descriptions in Table 4, and allowed for combined confidence scores ranging from

0 (low) to 3 (high) to be attributed to each image and for subsequent comparison between the two acoustic camera configurations tested.

Where the resolution allowed, colony formation 'type' (see Table 1) and 'phase' (see Curd *et al.*, 2019) were also assigned to each acoustic image for which the reef qualifying criteria were met. The visual quality of each acoustic image was also graded as either excellent, good, poor, very poor, or zero based on the resolution and focus of the seabed and Sabellaria colony structures, when present (Table 5).

Table 2: Sabellaria reef structure matrix modified by Jenkins et al. (2018) from the elevation and percentage cover categories proposed by Gubbay (2007).

Reef Structure Matrix			**Elevation (cm)**			
			<2	2 - 5	5 - 10	>10
			Not a Reef	Low	Medium	High
Patchiness (% Cover)	<10	Not a Reef	Not a Reef	Not a Reef	Not a Reef	Not a Reef
	10 - 20	Low	Not a Reef	Low	Low	Low
	20 - 30	Medium	Not a Reef	Low	Medium	Medium
	>30	High	Not a Reef	Low	Medium	High

Table 3: Confidence score assignments for the three colony parameters extent, patchiness and elevation.

Confidence	**Score**	**Extent**	**Patchiness**	**Elevation**
Low	0	Limited ability to visualise the extent of colonies.	Limited ability to distinguish colony coverage.	Limited ability to measure height of colonies above seabed.
Medium	0.5	Some ability to visualise the extent of colonies.	Some ability to distinguish colony coverage.	Some ability to measure height of colonies above seabed.
High	1	Full ability to visualise the extent of colonies.	Full ability to distinguish colony coverage.	Full ability to measure height of colonies above the seabed.

7.2.7. Data Archiving

The raw dataset resulting from his study was imported into the Marine Recorder database (https://jncc.gov.uk/our-work/marinerecorder/) under the '2017|2018 NRW/NE Severn Estuary SAC Sabellaria Reef Sonar Camera Survey' and is therefore available to download as a query-able Microsoft ACCESS 'snapshot'. This file included details of the multiple methods employed along with attributed habitat details, substrate composition, and the corresponding Marine Nature Conservation Review (MNCR) biotopes.

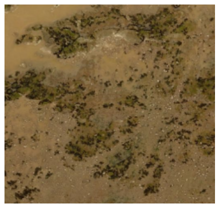
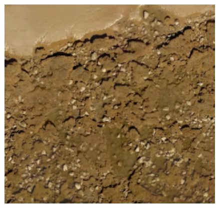
Table 4: ACI quality assignment categories adapted from Turner et al. (2016).

Image Category	Image Code	Description
Excellent	3	Image is clear with no motion blurring. Seabed fully focused across whole range with features clearly distinguishable and acoustic shadows clear and easily measured. All levels of analysis possible allowing for high confidence scores to be assigned.
Good	2	Image is clear with slight motion blurring. Seabed focused across the majority of the range with features still distinguishable and acoustic shadows measurable. Medium to high confidence scores may be assigned.
Poor	1	Image is unclear due to motion blurring. Seabed only focused across a limited portion of range. Features visible but difficult to distinguish. Acoustic shadows unclear and difficult to measure with accuracy. Low to medium confidence scores may be assigned.
Very Poor	0	Image is predominantly blurred from motion. Features undistinguishable and acoustic shadows unclear and unmeasurable. Low confidence scores may be assigned.

7.2.8. Cataloguing

All SSS data and UAV orthomosaics were visualized in ARCGIS 10.2, overlain with the position of each acoustic still image and intertidal quadrat. For each image location, thematic maps of the SSS data and UAV imagery were exported at a scale of 1:150 and catalogued with the corresponding acoustic image and intertidal photographs (where available) using the online tool BIIGLE 2.0 (Langenkämper et al., 2017). Each acoustic still and corresponding SSS, UAV, and quadrat image were annotated using a custom label tree created within BIIGLE. This allowed for the rapid viewing of all images tagged with any given category (e.g. *Sabellaria* colony 'type') and the rapid revaluation of the assigned labels to ensure consistency.

Table 5: S. alveolata colony formation 'Types' adapted from Gruet (1982).

Type	Description	Field Image	Aerial Image
Veneers	Encrusting low-lying colonies with overlapping tubes that lie at an acute angle often opposing the direction of prevailing wave action. These aggregations can cover large expanses of rocky shore and can completely outcompete other sessile fauna.		
Clumps	Colonies formed by clusters of tubes growing vertically that, when coalesced, can form semi-continuous sheets covering large expanses or mosaics of colonies aligned with topographic features (e.g. the crests of mega ripples).		
Hummocks	Ball shaped colonies constituted by tubes that radiate out from an initial settlement point. Generally found attached to cobbles/boulders frequently covering the entire upper surface and growing larger than the cobble/boulder itself. These colonies form the continuous platform formation when many coalesce.		
Platforms	Continuous, relatively flat colonies formed by fully coalesced hummocks. Generally found outcompeting all sessile epibiota on rugose rock or consolidated cobbles and/or boulders (including artificial structures).		

7.2.9. Statistical Analysis

Normality was not assumed for analysis comparing camera configuration confidence scores when assessing reef parameters. A nonparametric Kruskal–Wallis one-way analysis of variance (ANOVA) on ranks was therefore conducted to test whether differences were present for confidence scores given for extent, patchiness, and elevation measurements taken from the two camera configurations (frame and pole mounted). Comparisons were also made of the differences in confidence levels with regards to image quality between the two camera configurations. Bland–Altman analysis was used to compare the elevation measurements generated from both the frame and pole mounted ACI versus UAV and SSS elevations. This analysis calculates the mean difference between two methods of measurement and 95% limits of agreement as the mean difference

(Giavarina, 2015), with the 95% confidence interval being calculated as the mean of the two values ±1.96 SD. The inclusion of these limits of agreement is to aid visual assessments of how well the two methods of measurement agree: the smaller the range between these two limits the better the agreement is (Myles & Cui, 2007). All statistical analysis was conducted in R 3.4.2 (www.r-project.org).

7.3. Results

In general, the ACI collected using both deployment configurations matched well with the acoustic signatures of the seabed features in the corresponding SSS data as well as the same features visible in the aerial imagery (see Figure 3). The imagery was of suitable resolution for visualizing individual *Sabellaria* colony structures when present as well as other seabed features such as cobbles, mega-ripples and sand waves (see Figure 3). This meant that it was possible to establish the broad extent of the *Sabellaria* colony structures as well as their patchiness and elevation above the seabed. Ninety-two of the 145 acoustic images analysed were deemed to have met the reef qualifying criteria and were therefore assigned a reef 'status', in line with Table 3 (high, n = 46; medium, n = 32; low, n = 14). Of these 92 images, 88 were classed as clumps and four were recorded as veneers, in accordance with the descriptions in Table 1. No hummock or platform reef structures were recorded. The remaining 53 images were representative of areas of bare bedrock, sand, mixed sediment, and coarse sediment. These were assigned 'not reef' because of the absence of *Sabellaria* colony structures, although 25 locations were thought to be representative of Annex-I bedrock (n = 18) or stony reef (n = 7) habitat, as described by (Irving, 2009). It was not possible to identify the *Sabellaria* species based on the ACI alone, despite the high resolution (see Figure 3). Quadrat and site photographs taken during the intertidal sampling did, however, suggest that colony structures were formed by *S. alveolata*, by the presence of tube porches not normally formed by *S. spinulosa* (Pearce et al., 2014).

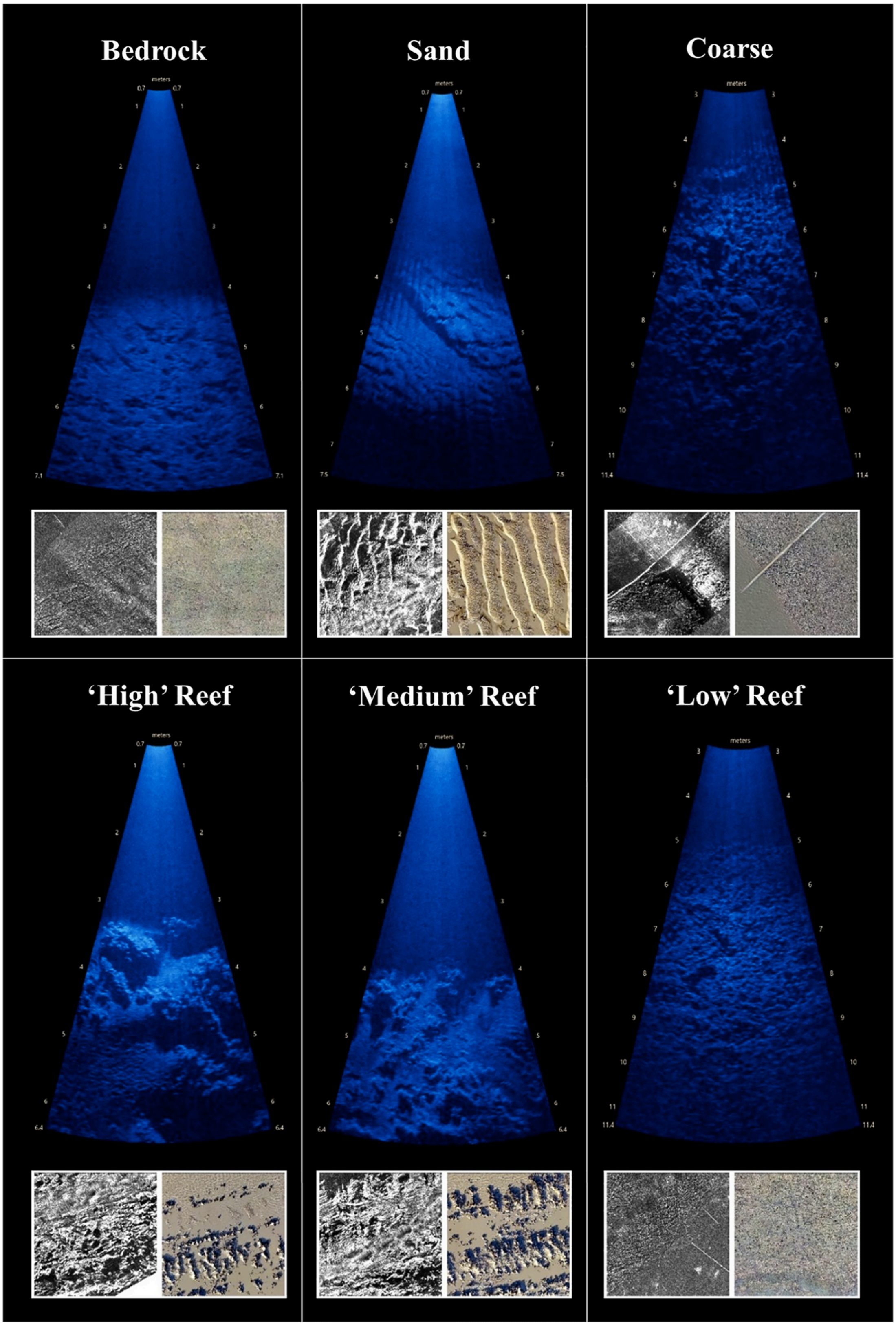


Figure 3: Acoustic camera imagery (ACI) representative of key non-reef substrates (top) and each Sabellaria reef 'status' (bottom). Corresponding side scan sonar (SSS, greyscale) and unmanned aerial vehicle (UAV) imagery is also presented for each location.

7.3.1. Reef Parameters

High confidence scores were generally assigned to imagery of seabed with no *Sabellaria* coverage (i.e. bedrock), bare sand waves, or areas of distinctly elevated and high-coverage *Sabellaria* reef (see Figure 4). Lower confidence scores were generally assigned to ACI of low-lying cobbles/pebbles and/or medium to low-lying reef (see Figure 4). In the most part, the extent could be confidently assessed through a review of the SSS and UAV data ground-truthed by the ACI. It was also found that the wide field of view and range of the ACI provided an independent means of assessing the extent, when the resolution allowed. Patchiness could be estimated using ACI combined with UAV and SSS data, whereas elevation was estimated based on measurements of the acoustic shadows cast by seabed features (i.e. *Sabellaria* colonies). Elevations were also calculated separately using the UAV, SSS, and quadrat data as a means of testing the relative accuracy of the measurements derived from ACI (Table 6). Figure 5 shows that the discrepancy between UAV- and ACI-derived measurement methods are, for the most part, within the limits of agreement. Two outliers are noted on and below the limits of agreement (dashed lines) when comparing frame mounted ACI and UAV elevation measurements, suggesting a large discrepancy between elevation readings at two locations. Similar results are evident when comparing SSS- and ACI-derived measurements, with all but one point lying within the limits of agreement.

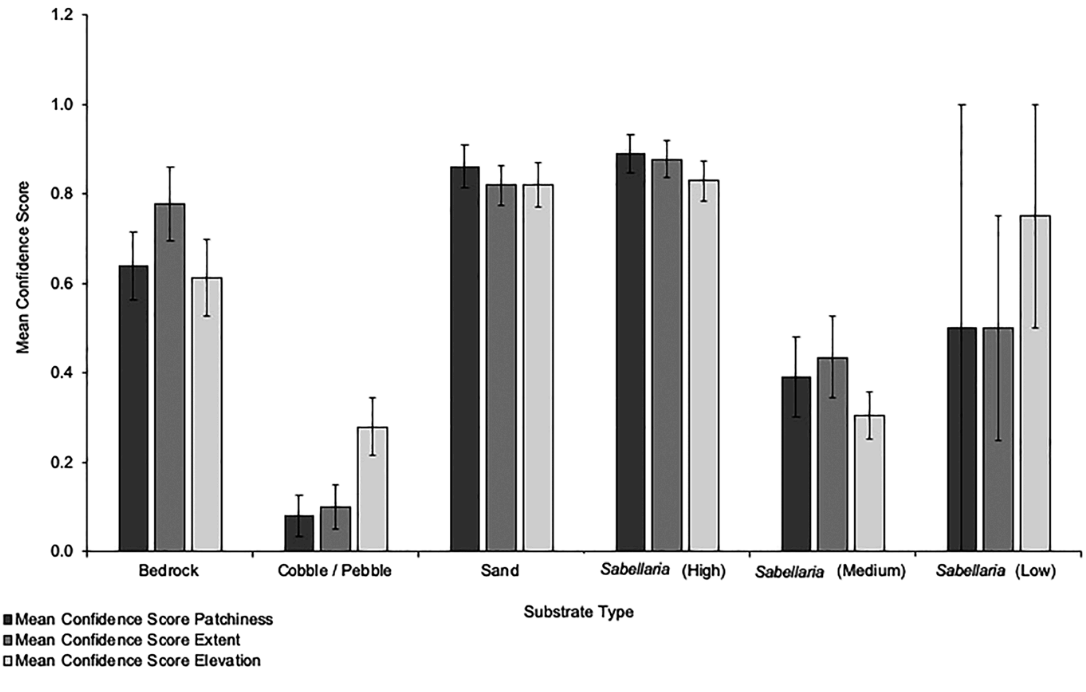


Figure 4: Mean (± SE) confidence scores for patchiness, extent and elevation for the different substrate types and Sabellaria reef status identified during the study from ACI.

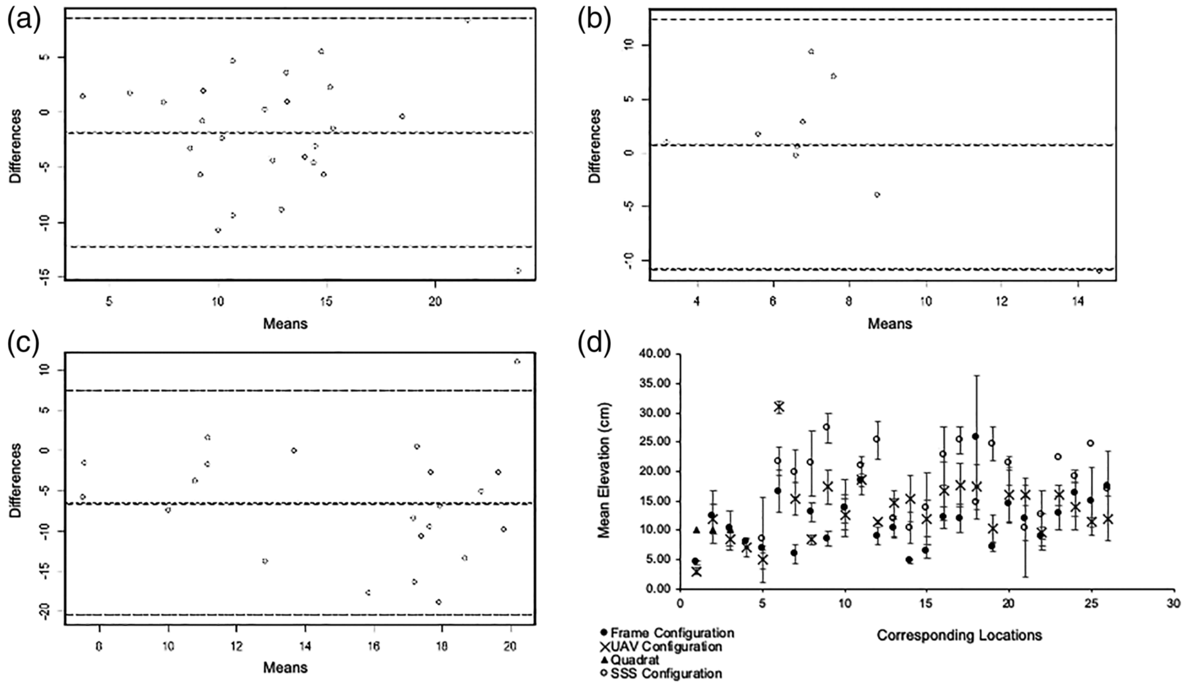


Figure 5: Bland–Altman plots for (a) frame-mounted acoustic camera imagery (ACI) and unmanned aerial vehicle (UAV) elevation, (b) pole mounted ACI and UAV elevation, (c) frame-mounted ACI and side-scan sonar (SSS) elevation, and (d) differences in elevation measurements (cm) taken from ACI, SSS, UAV, and quadrat analyses in corresponding locations of Sabellaria colonies. Difference = configuration – method. Means are the averages of the two measurements. Horizontal lines are drawn at the mean difference and the limits of agreement (defined as ±1.96 SD of differences).

7.3.2. Deployment Configuration Comparison

Comparisons were made between frame and pole-mounted deployment configurations (Figure 6; Table 6) to test whether a particular set-up was more appropriate for identifying *Sabellaria* colony structures and assessing whether they met the reef qualifying criteria when considered alongside corresponding SSS data. Analysis indicated that pole-mounted ACI was less effective for distinguishing between reef and cobbles/bedrock when compared with the frame-mounted ACI. Significantly higher confidence scores were assigned to ACI collected using the frame-mounted configuration in comparison with the pole-mounted configuration for each of the three *Sabellaria* colony parameters (extent, $H_1 = 8.21$, $P \leq 0.01$; patchiness, $H_1 = 14.02$, $P \leq 0.01$; elevation, $H_1 = 14.15$, $P \leq 0.01$; Figure 7). The average combined confidence score for the frame mounted configuration was 2.23 ± 0.11, whereas the average combined confidence score for the pole-mounted configuration was 1.49 ± 0.15 (combined, $H_1 = 12.40$, $P \leq 0.01$) although difficulties differentiating between low-lying *Sabellaria* reef structures and non-reef substrates were encountered for both configurations. Frame-mount-derived ACI was generally of higher visual quality than that derived using the pole-mounted configuration (Figure 7b). Of the 78 frame-mount-derived images, 25 were considered to be of excellent quality and 32 were considered to be of good quality. The remaining 20 images were considered to be of poor or very poor quality. Of the 68 pole-mount-derived images, only nine images were considered to be of excellent quality and 49 were considered to be of good quality. The remaining eight images were considered to be of poor quality. Combined confidence scores for the images assessed as excellent were similar for the two mounting configurations; however, the difference between them was still considered significant (excellent, $H_1 = 7.35$, $P \leq 0.01$). Differences in confidence scores for good and poor image qualities were more apparent (good, $H_1 = 9.45$, $P \leq 0.01$; poor, $H_1 = 9.14$, $P \leq 0.01$) (Figure 7). These findings suggest that, overall, the frame-mounted configuration allowed for the assessment of *Sabellaria* reef parameters with a higher level of confidence compared with the pole-mounted configuration.

7.3.3. Formation Type

Of the 95 acoustic images assessed as containing *Sabellaria* colony formations, 91 were considered to be representative of clump formations and four were classed as veneer formations. No hummock or platform structures were recorded. Quadrat assessments on transects T02, T03, and T04 and consideration of UAV imagery of T04E, T05, and T06 confirmed these findings. The high resolution of the ACI acquired meant that the erect form of the *Sabellaria* clump formations were easily recognizable and differentiated from veneer formations and other seabed features (e.g. cobbles). Veneers were difficult to differentiate from bare bedrock and coarse and mixed sediment because of their low-lying nature, however, and could only be confirmed when intertidal quadrat photographs were available.

7.3.4. Other Features

Despite the high resolution of the ACI collected, it was not possible to establish which *Sabellaria* species was responsible for constructing the colony formations, nor was it possible to determine whether the colonies were dead or alive. This requires the confirmation of the presence of tube porches and faecal pellets, respectively, neither of which were distinguishable in the ACI, even when confirmed as present during the ground-truth quadrat sampling. The ACI was, however, capable of identifying other reef types, including bedrock reefs and cobble/stony reefs. Cobble/stony reefs require areas of cobble substrate to meet the extent, elevation, and percentage cover criteria to quality as Annex-I 'reef' (Irving, 2009) each of which could be readily assessed from the ACI collected in combination with the SSS data in the same way as *Sabellaria* reef presence/absence was determined.

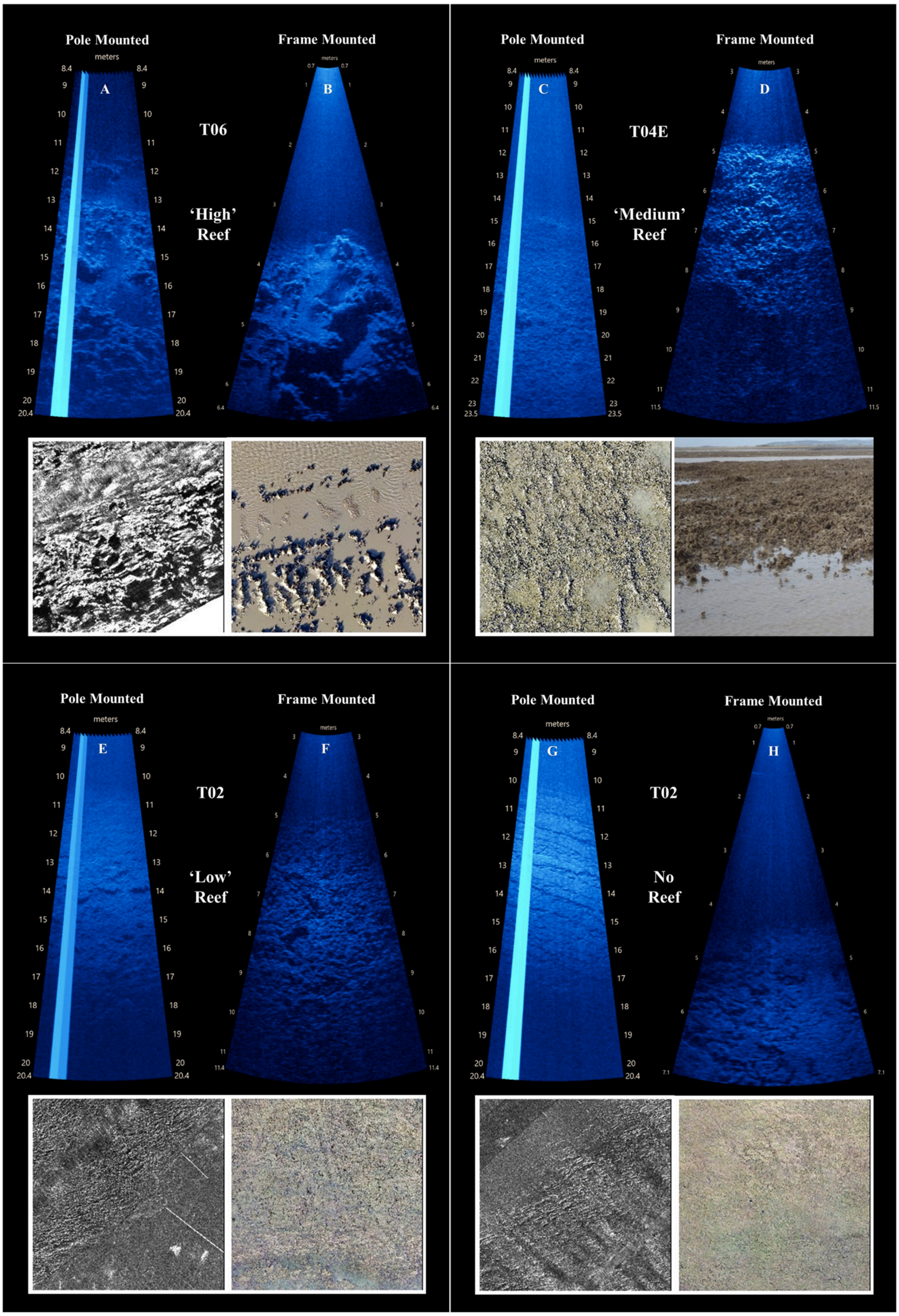


Figure 6: Comparison of ACI collected at the same locations along two transects (left: T06, right T04E) using the two camera configurations. A, C, E and G = pole mounted; B, D, F and H = frame mounted). Note there were a number of beams missing from all data collected using the pole-mounted configuration shown as light blue strips to the left of each sounding.

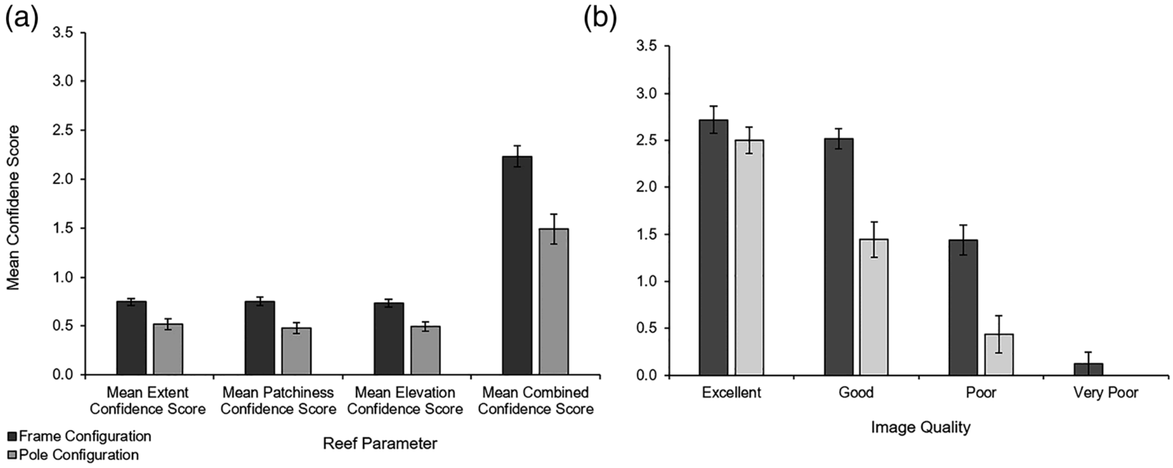


Figure 7: a) Mean (± SE) confidence scores for extent, patchiness and elevation for the frame and pole mounting configurations b) Mean combined confidence scores for the four image quality categories; excellent, good, poor and very poor for the two mounting configurations.

Table 6: Mean (±SE) confidence scores (0–1) for each Sabellaria reef parameter and acoustic camera imagery (ACI) image quality category assignment per transect sampled. Also shown are mean(±SE) elevation measurements derived from ACI, side-scan sonar (SSS), unmanned aerial vehicle (UAV), and quadrat (QDT) data

Mounting Configuration	Transect	Reef parameters*			Image Quality*				Elevation Measurements**			
		Extent	Patchiness	Elevation	Excellent	Good	Poor	Very Poor	ACI	SSS	UAV	QDT
Frame	T02	0.50 (+/-0.16)	0.38 (+/-0.18)	0.63 (+/-0.13)	0.58 (+/-0.14)	0.42 (+/-0.13)	0.17 (+/-0.17)	-	4.44 (+/-0.31)	-	3.00 (+/-0.58)	2.50 (+/-0.00)
	T03	0.50 (+/-0.29)	0.67 (+/-0.17)	0.67 (+/-0.33)	0.83 (+/-0.11)	-	0.17 (+/-0.17)	-	-	-	-	-
	T04	-	-	-	-	-	-	-	-	-	-	-
	T04E	1.00 (+/-0.00)	0.83 (+/-0.08)	0.56 (+/-0.06)	-	0.83 (+/-0.06)	0.78 (+/-0.09)	-	9.33 (+/-1.33)	8.33 (+/-1.45)	8.08 (+/-1.10)	10.00 (+/-0.00)
	T05	-	-	-	-	-	-	-	-	-	-	-
	T06	0.75 (+/-0.04)	0.80 (+/-0.04)	0.78 (+/-0.05)	0.98 (+/-0.01)	0.88 (+/-0.03)	0.43 (+/-0.43)	0.04 (+/-0.04)	12.40 (+/-0.91)	19.08 (+/-1.01)	14.94 (+/-0.79)	-
	Overall	0.75 (+/-0.04)	0.75 (+/-0.04)	0.73 (+/-0.04)	2.72 (+/-0.14)	2.52 (+/- 0.11)	1.44 (+/-0.16)	0.13 (+/-0.13)	-	-	-	-
Pole	T02	0.73 (+/-0.10)	0.57 (+/-0.08)	0.53 (+/-0.09)	-	0.73 (+/-0.05)	0.17 (+/-0.08)	-	8.24 (+/-1.04)	-	3.42 (+/-0.45)	3.00 (+/-0.71)
	T03	0.00 (+/-0.00)	0.00 (+/-0.00)	0.00 (+/-0.00)	-	0.00 (+/-0.00)	-	-	7.09 (+/-0.57)	-	7.25 (+/-0.83)	4.00 (+/-1.47)
	T04	0.00 (+/-0.00)	0.00 (+/-0.00)	0.30 (+/-0.08)	-	0.14 (+/-0.05)	0.04 (+/-0.04)	-	-	-	-	-
	T04E	0.14 (+/-0.07)	0.09 (+/-0.06)	0.23 (+/-0.08)	-	0.12 (+/-0.04)	0.50 (+/-0.00)	-	9.12 (+/-2.99)	-	20.00 (+/-2.31)	10.00 (+/-0.00)
	T05	1.00 (+/-0.00)	1.00 (+/-0.00)	1.00 (+/-0.00)	1.00 (+/-0.00)	1.00 (+/-0.00)	-	-	-	-	-	-
	T06	0.92 (+/- 0.05)	0.92 (+/-0.05)	0.69 (+/-0.07)	0.81 (+/-0.05)	0.90 (+/-0.05)	-	-	-	-	-	-
	Overall	0.52 (+/-0.06)	0.48 (+/-0.05)	0.49 (+/-0.05)	2.50 (+/-0.14)	1.48 (+/-0.18)	0.44 (+/-0.20)	-	-	-	-	-

*Mean ± SE Confidence Scores
**Mean ± SE Elevation Measurements

7.4. Discussion

This proof-of-concept study has demonstrated the effectiveness of collecting and interpreting ACI as a method of visualizing *Sabellaria* bioconstructions in turbid or near-zero-visibility underwater conditions. It also demonstrates that ACI can facilitate the classification of *Sabellaria* colonies into 'reef' or 'non-reef' categories, in line with optical imagery approaches. It can also provide information on reef morphology, enabling application in the monitoring and management of these and other protected biogenic habitats.

7.4.1. Assessing Reef Extent

Mapping the spatial distribution of key benthic habitats is an important process for conservation bodies across Europe to enable the effective management of protected sites and ensure they fulfil their statutory reporting obligations. These processes are well advanced in coastal and offshore areas, generally relying on the interpretation of backscatter (either SSS or multibeam echo-sounder derived) and bathymetry data to delineate boundaries of habitats based on their acoustic signatures (Boyd et al., 2006; Diesing et al., 2014). Adequate ground-truthing via visual or physical sampling approaches is vital to both prove presence and delineate the extent, especially for biogenic features formed by polychaete worms, bivalves, and seagrasses (Lindenbaum et al., 2008; Montefalcone et al., 2013). The most common approach is to collect video and stills photography of the seabed, which are the preferred approach over extractive sampling techniques for biogenic habitats sensitive to physical damage (Davies et al., 2001; Foster-Smith & Hendrick, 2003). This approach has been successfully employed to accurately map reef habitats in offshore areas (Jenkins et al., 2018; Pearce et al., 2014) but, with the growing interest in harvesting wave and tidal energy resources (Gill, 2005; Mélin & Vantrepotte, 2015), there is an increasing need to understand the spatial distribution of reef habitats in turbid nearshore areas where optical ground-truthing approaches are largely redundant. This highlights the need for alternative ground-truthing approaches capable of discriminating boundaries of reef habitats in low-visibility environments. The results of this study reveal that ACI can act as a reliable substitute for photographic approaches used to ground-truth acoustic signatures produced by *Sabellaria* reef habitat. The high-resolution nature of the ACI collected means that distinctly elevated *Sabellaria* colonies were clearly discernible from the surrounding seabed using both pole- and frame-mounted camera configurations. Importantly, this means that the extent of *Sabellaria* colonies can be determined from the ground-truthed SSS and, to some extent, the acoustic imagery itself over smaller spatial scales. This is important in a conservation context as one of the generally accepted criteria for Sabellaria colonies to qualify as Annex-I 'reef' is for colonies to cover an area of greater than 25 m^2. The results are also promising with respect to employing ACI approaches for ground-truthing SSS derived mapping of other important benthic habitats in low-visibility conditions. This could potentially include other designated SAC features, such as stony reefs and sand banks, both marine and freshwater mussel beds (Lindenbaum et al., 2008; Powers et al., 2015), and seagrass meadows, for which a number of trials are currently continuing in Wales (J. Griffiths, pers. comm., June 2019).

7.4.2. Assessing Reef Patchiness

The other reef qualifying criteria normally assessed using optical imagery include patchiness and elevation (Hendrick & Foster-Smith, 2006), both of which are unmeasurable using optical imagery approaches in highly turbid environments. Percentage coverage is usually assigned to stills or segments of seabed video as a proxy for patchiness (Gubbay, 2007); however, it is argued that this may result in a measure of reef density rather than a measure of true patchiness, resulting in the development of alternative approaches (Jenkins et al., 2018). To obtain imagery of suitable quality and resolution for assessing *Sabellaria* colony features, seabed camera systems are commonly configured to obtain nadir or off-nadir acute angle imagery, with the cameras deployed close to the seabed (Durden et al., 2016; Hitchin, 2015). This results in limitations to the field of view (FOV) of such systems, often ranging between 0.5 and 1 m_2, as well as variability of FOV when the camera frames are 'hovered' at variable altitudes above the seabed (Goudge et al., 2016; Sheehan et al., 2010).

This study demonstrates that acoustic cameras can not only produce high-resolution imagery across a much larger and consistent FOV than optical camera systems (between 6 and 20 m in this study), but also achieves this in near zero visibility underwater environments. The FOV achieved also allowed for the consistent assessment of patchiness over an area of near to the minimum reef-qualifying extent (25 m^2). As such, it was possible to assess whether patches of

Sabellaria colonies met the reef qualifying criteria based on snapshots of ACI, negating the need to analyse video segments, as is commonly undertaken for optically derived imagery (Jenkins et al., 2018; Turner et al., 2016).

7.4.3. Assessing Reef Elevation

The elevation of Sabellaria colonies is arguably the key characteristic of reefs formed by both *S. alveolata* and *S. spinulosa,* as it is this topographical distinctness that largely underpins their conservation value. This is evident when considering the Habitats Directive and OSPAR Convention (Convention for the Protection of the Marine Environment of the North-East Atlantic) reef definitions that describe them as 'concretions which arise from the sea floor' (European Commision, 2016), and note that *S. spinulosa reef* 'habitat should be thick and persistent' (OSPAR, 2008). Estimation of elevation from optical imagery is normally facilitated by including a measure of scale in the FOV. This is best achieved using point lasers separated by a known distance, providing a consistent scale regardless of the height of the camera from the seabed (Barker et al., 2001). Although effective in scaling features in the horizontal plane, this approach provides limited ability to accurately measure features in the vertical plane (i.e. elevation). As such, elevation measurements of *Sabellaria* colonies are often based on expert judgement, estimations using SSS, and/or multibeam echosounder (MBES) data (Pearce et al., 2014) or require validation through the collection of physical samples (Fariñas Franco et al., 2014; Hendrick & Foster-Smith, 2006). This study not only demonstrates that ACI can be used to accurately measure *Sabellaria* colony elevation, but also achieves this in near-zero-visibility underwater environments. We demonstrate that elevation estimates derived from ACI are largely within the limits of agreement when compared with elevation estimates derived from SSS and UAV data. These calculations do come with associated errors introduced through the processing of the imagery, but arguably provide a repeatable and less subjective method for quantifying elevation compared with estimations based on photographic footage.

The confidence in the elevation estimations calculated during this current study were shown to reduce substantially when assessing areas of known low-lying *Sabellaria* reef and cobble substrate (confirmed based on intertidal field observations and UAV imagery) and were reduced further using the pole-mounted configuration. This was because of the lack of acoustic shadows in the ACI that could be confidently attributed to *Sabellaria* colonies rather than cobbles, combined with the difficulty in measuring them accurately. Similar issues are well documented for assessing low-lying *Sabellaria* formations on coarse and mixed sediments using photographic imagery (Gubbay, 2007; Irving, 2009; Jenkins et al., 2018; Limpenny et al., 2010), and for coarse and mixed sediments in general (Diesing et al., 2014). These problematic low-lying colonies are commonly just above 2 cm in height, meaning that they meet the Annex-I reef qualifying criteria for elevation (Gubbay, 2007) (assuming other qualifying criteria are met). As such, their oversight may result in inaccurate distribution and extent mapping, potentially having implications in terms of management and conservation of designated Sabellaria reefs. Further development of ACI approaches and interpretation of the resulting outputs is therefore needed to ensure Sabellaria colonies can be confidently assessed, regardless of their topographic distinctness from the underlying and surrounding substrate.

7.4.4. Assessing Reef Type

Sabellaria colonies are known to take a myriad of forms, linked to prevailing environmental conditions and anthropogenic disturbances. This can result in a gradient of morphological types across relatively small spatial scales, related to factors such as position on the shore, supply of sand-sized particles, rate of smothering, and growth 'phase' (Curd et al., 2019; Gruet, 1986). In the intertidal, the range of formation types observed for *S. alveolata* have been grouped into distinct categories based on the morphology of the colonies (see Table 1). The cycling between these categories is thought to be linked to distinct 'growth' and 'destruction' phases (Gruet, 1986), whereby they retrograde or prograde, either partially or totally, through settlement (Curd et al., 2019). Unlike intertidal reefs formed by *S. alveolata,* subtidal reefs formed primarily by *S. spinulosa are* thought to have fewer defined sub-habitat types, as the developmental cycle and growth-and destruction phases similar to those described for intertidal reefs have not been documented. There are references to 'crusts' or 'veneers' where *S. spinulosa* occurs in high densities but do not form topographically distinct features (Holt, Rees, Hawkins, & Seed, 1998) as well as 'nodule'-like aggregations made up of clusters of vacant tubes unattached to the substrate (Limpenny et al., 2010). There are also descriptions of S. spinulosa colonies as 'clumps' (Fariñas Franco et al., 2014) that match the descriptions of intertidal *S. alveolata* colonies described in the Severn Estuary (R. Griffin, pers. comm., May 2019), as well as subtidal 'platform' type *S. spinulosa* colonies of up to 70 cm in height (Lisco et al., 2017) that resemble intertidal *S. alveolata* reef platform structures found in the UK (R. Griffin, pers. comm., May 2019; see Table 1) and France (Desroy et al., 2011). This study combined the colony types first described by Gruet (1986) with a 'clump'-type category to form a colony 'type' classification applicable to intertidal and subtidal reefs formed by both *S. alveolata* and *S. spinulosa* (Figure 8). This provided a framework to allow the intertidal *Sabellaria* colonies that are found across the study site to be used as surrogates for the range of colony 'types' that may also be encountered in subtidal areas.

Figure 8: Sabellaria 'clump' colonies found on the lower shore at Goldcliff (T04E), assigned 'high' reef status for the continuous coverage and distinctly elevated form.

Despite this, the majority of *Sabellaria* colonies observed during this study were classified as 'clumps', with no hummock or platform types recorded. This is not an indication that ACI techniques are incapable of detecting these formation types, as intertidal inspection and UAV mapping confirmed their absence across the study area. Furthermore, hummock and platform types are rare throughout Europe (Curd et al., 2019), and therefore it is suspected that these climax morphological forms are rarely reached in the UK. The lack of hummock and platform colony type observations during the current study does not necessarily confirm that these formation types are absent from the subtidal reaches of the study site. Rather, it highlights the need for further research to test whether the full spectrum of colony formation types can be visualized and differentiated using ACI.

7.4.5. Configuration Comparison

A greater FOV was achieved using the pole-mounted approach; however, the distance of the acoustic camera from the seabed using this set-up was dependent upon the water depth. Inconsistent image resolution was therefore linked with varying water depths at different stages of the tidal cycle. Conversely, it was shown that the frame mounted configuration provided consistently higher resolution imagery than the pole-mounted configuration, but across a reduced FOV (6–11 m), as the acoustic camera was positioned at a fixed height above the seabed, regardless of the water depth. A key issue faced with the pole-mounted approach was the blurring of the ACI as a result of the motion of the vessel. To build up a single image frame, the acoustic cameras (ARIS 1800 and 3000) required several ping cycles, causing distortion to be introduced if the cameras were moving too quickly. The hyper-tidal nature of the Severn Estuary means current speeds can reach several metres per second (Xia et al., 2010), which meant that maintaining a slow and steady speed along predefined transects was difficult. This, combined with the vertical movement of the vessel, meant that image distortion was unavoidable, even when applying the platform motion correction filter available in ariscope. By mounting the acoustic camera in a seabed frame similar to those used for deploying optical cameras (Hitchin et al., 2015) it was possible to land and hold the camera in a fixed location and altitude for several seconds, allowing for enough ping cycles to build undistorted images. As such, the frame-mount-derived ACI was of greater use for assessing *Sabellaria* colonies than that derived from the use of a pole mount, despite the reduced FOV. The pole-mount approach did not, however, require landing a 100-kg frame on the seabed along each transect, which inevitably resulted in damage to the colony. A number of studies have, however, demonstrated that *S. alveolata* reefs can rapidly recover from a single episode of trawling (Vorberg, 2000) and trampling (Cunningham, Hawkins, Jones, & Burrows, 1984), suggesting that any impacts caused by landing the frame were likely to have been temporary.

7.5. Conclusions

Very little is known of the spatial distribution and nature of *Sabellaria* reefs found throughout the Severn Estuary beyond the intertidal zone. It is, however, possible that the reefs studied at Goldcliff, and those present elsewhere in the Severn Estuary, may extend far into the subtidal, forming a continuous or a number of large biogenic structure(s) of notable conservation importance. Given the protected status of the estuary and the continued interest in the development of large-scale infrastructure to harness this tidal resource, it is clear that further research to map the currently unknown extent and status of the reef features that it supports is needed. The novel approaches tested in this study represent a promising means of achieving this, although additional studies will be needed to further improve the approach, particularly with regards to differentiating between low lying *Sabellaria* formations and other seabed features. This is especially important for the Severn Estuary SAC, as the conservation advice for the site places an emphasis on protecting all *S. alveolata* formations (Natural England & Countryside Council for Wales, 2009). This includes low-relief veneers that are thought to play an important role in supporting established reefs as they evolve through the progradation and retrogradation phases and should be considered as part of any management measures (Curd et al., 2019).

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Glossary

Assemblage Composition	Taxonomically related group of species populations that occur together in space.
Baseline survey	Determines the characterization of an area prior the development of a project and establish the initial environmental status.
Benthic	Anything associated with or occurring on the bottom of a body of water.
Biodiversity	The variety of plant and animal life in the world or in a particular habitat, a high level of which is usually considered to be important and desirable.
Coastal	Relating / near to the coast. For this thesis, the definition of the coastal zone follows the JRC-IES Coastal Zone Technical Note which identifies a 10km buffer seaward from the coastline and a 2km buffer inland to cover the following inland and intertidal areas; salines; intertidal flats; coastal lagoons and estuaries.
Control	An experiment or observation designed to minimise the effects of variables other than the independent variable.
Duel-frequency	Equipment which can operate at two frequencies.

Ecogram	A recording of depth or distance under water made by an echo sounder.
Ecosystem	A biological community of interacting organisms and their physical environment.
Error	A measure of the estimated difference between the observed or calculated value of a quantity and its true value.
Habitat	The natural home or environment of an animal, plant, or other organism.
Heterogeneous	It refers to the uneven distribution of various concentrations of each species within an area.
Homogeneous	Is a lack of biodiversity and even distribution of various concentrations of each species within an area.
Hydroacoustics	The study and application of sound in water.
Quantitative	Quantitative information or data is based on quantities obtained using a quantifiable measurement process.
MaxN	Maximum number of individuals of a family.
Metadata	A set of data that describes and gives information about other data.
Multivariate	Involving two or more variable quantities.

Organism	An individual animal, plant, or single-celled life form.
Remote	Deployed equipment with no correction to vessel or shore.
Replicate	A repeated test or experiment.
Spatial	Relating to or occupying space.
Species richness	The number of different species represented in an ecological community, landscape or region.
Sonar	A technique uses sound propagation to navigate, communicate with or detect objects on or under the surface of the water.
Standardisation	The process of making something conform to a standard.
Taxonomic diversity	The number of different taxa represented in an ecological community, landscape or region.
Turbidity	Turbidity is a measure of the degree to which the water loses its transparency due to the presence of suspended particulates.
Underwater visibility	Estimation of water clarity by the distance you can see either horizontally or vertically.
Univariate	Involving one variable quantity.

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