Evaluation of Baited Remote Underwater Video Systems (BRUVS) for monitoring fish communities in Lake Malawi/Niassa

This thesis is submitted in fulfilment of the requirements for the degree of

MASTER OF SCIENCE

at

RHODES UNIVERSITY

by

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December 2019

ABSTRACT

Baited Remote Underwater Video systems (BRUVS) have become a well-established *in-situ* monitoring technique in clearwater aquatic ecosystems. The relatively low cost, non-destructive and non-extractive nature of this technique permits BRUVS to be employed in a wide range of habitats. To date, the vast majority of studies using BRUVS technology have been conducted in marine habitats. Subsequently, BRUVS applications in freshwater habitats are scant, and techniques are not well developed. The primary objective of this thesis was to address this knowledge gap and develop standard operating procedures for BRUVS employment in Lake Malawi/Niassa and explore the potential of BRUVS as a monitoring tool for fish communities in the African Great Lakes.

Eight easily identifiable species groups, representative of Lake Malawi/Niassa inshore fish communities, were used to develop the technique. The optimal BRUVS deployment time to obtain 95 % species accumulation was achieved in a 15-minute recording period. Power analysis, using a pre-determined 80 % power, a confidence interval of 95 % and a significance level of $\alpha < 0.05$ was used to determine annual sampling effort requirements for each species group. The power analysis was performed to detect a 10 % change in abundance over a hypothetical 10-year monitoring scenario. In areas where fish abundance was lower, the sampling effort required to monitor key fisheries species was significantly higher. For example, Chambo, the local *Oreochromis* (*Nyasalapia*) species flock, required an annual sampling effort of 120 deployments in Malawi compared to 56 in Mozambique ($\alpha < 0.05$). Chambo had a higher detection probability in areas of lesser fishing pressure and were found in higher abundances in deeper, less accessible habitats. Deep-water (> 20 m) and rocky habitats were most important in explaining Chambo abundance and detection probability. The size-structure of Chambo in Lake Malawi/Niassa reflects size-specific depth and habitat migrations. Larger Chambo were observed aggregating in waters deeper than 20 m and a broader size range of individuals were observed utilising structured habitat. The effects of fishing are apparent in the size-structure of Chambo in the areas sampled. In study areas exposed to greater levels of fishing pressure – such as Malawi, the BRUVS detected significantly fewer individuals within sexually mature size classes, and the average size was smaller than in areas with less exploitation.

Keywords: Freshwater, Africa, Lake Malawi/Niassa, baited remote underwater video**,** BRUVS

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ACKNOWLEDGEMENTS

My MSc degree has been a great adventure. It would not have been possible, however, without my two supervisors. Professor Olaf Weyl and Doctor Anthony Bernard. Olaf, I will always be thankful for the opportunities that you have given me and for taking time out of your busy life to help me become a better researcher and scholar. And Ant, I cannot thank you enough for all your help and great advice that you have given me over the past few years – it has been a pleasure to be your student. I acknowledge use of infrastructure and equipment provided by the South African Institute for Aquatic Biodiversity (SAIAB) and funding channelled through the NRF-SAIAB Institutional Support system

I want to express my sincere gratitude to my master's degree funder, the National Research Foundation (NRF) of South Africa, DSI-NRF – South African Research Chairs Initiative of the Department of Science and Innovation (Grant No. 110507), to the funder of the Lake Malawi remote imagery research, the World Wide Fund for Nature (WWF) and the Department of Ichthyology and Fisheries Science (Rhodes University). The research that went into this thesis would not have been possible, of course, without a great team of international collaborators. This includes Antonio Pegado and Carlos Jonasse from the Instituto de Investigação de Pesquera, Eugido Gobo from WWF-Mozambique, Maxon Ngochera from the Malawi Fisheries Research Institute and the incredible research team at the SAIAB. Jeff, Sanet, Ferdi, Josie, Taku and the rest of the SAIAB family, thank you all so much for your help along the way. Two anonymous external examiners are also thanked and acknowledged for providing helpful and constructive comments which improved the thesis

My final massive thank you is to my partner, family and friends. Megan, thank you for always being in my corner and for your endless positivity. To my dad and Mavic, thank you again for tirelessly supporting me and my decisions. I could not have two better parents and role models.

DECLARATION

I have read and understood the University’s Policy on Plagiarism. All of the work in this essay is my own. I have not included ideas, phrases, passages or illustrations from another person’s work without acknowledging their authorship.

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Date: December 2019

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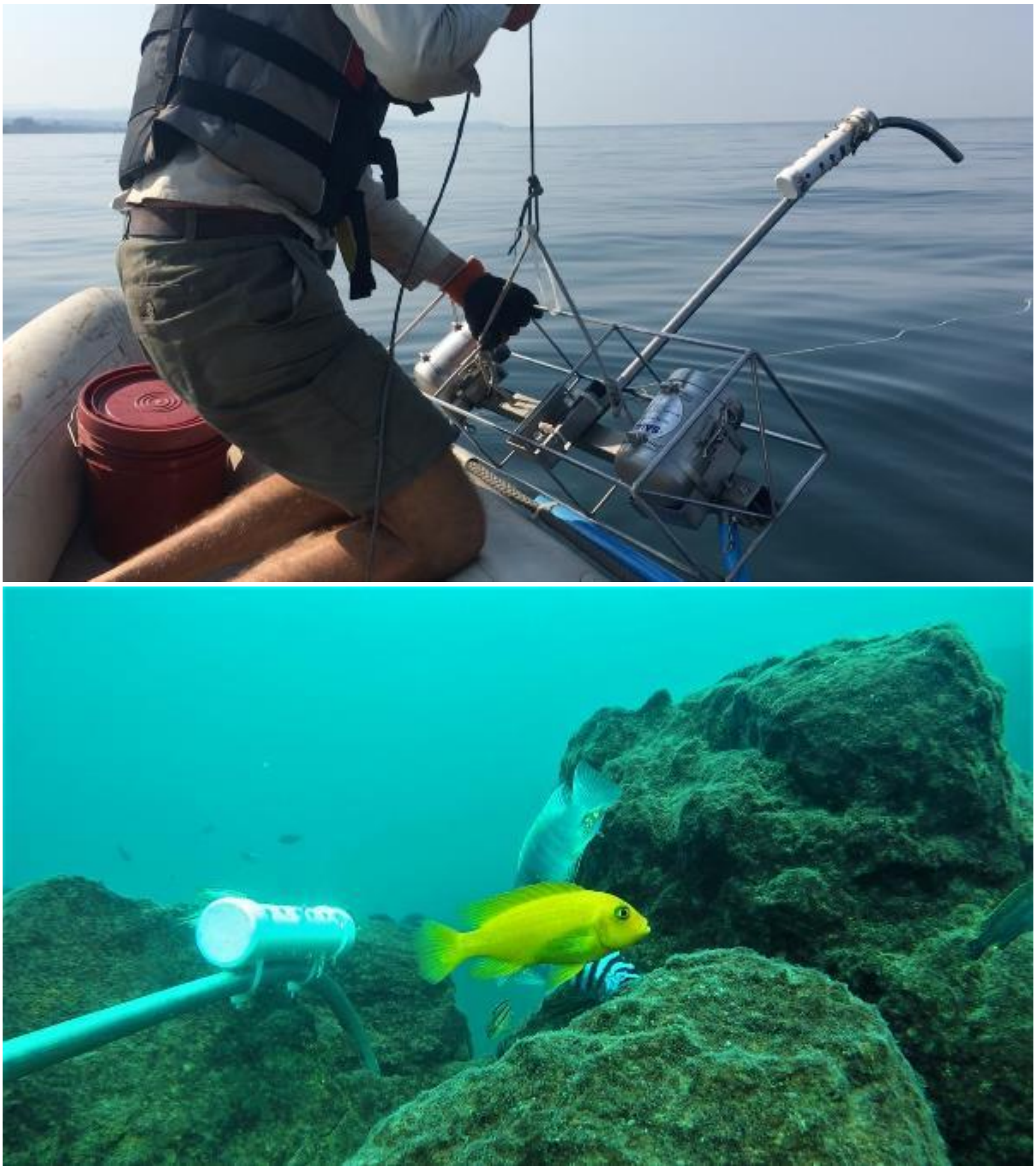
GENERAL INTRODUCTION



Deployment of stereo-BRUVS into Lake Malawi (Top), image of the bright yellow Metriaclima estherae upon a reef in the Lake Niassa Reserve (Bottom). P.C. Angus van Wyk (SAIAB)

On a global scale, destructive human practices and climate change are driving biodiversity loss at an unprecedented rate (Pimm et al. 2014). Determining trends and definite effects of these losses remains, however, a significant challenge – the consequences of which, are devastating for those human populations dependent upon the affected systems (Cardinale et al. 2012, Hoffmann et al. 2014, Schmeller et al. 2017). No more is this occurrence as apparent as it is in freshwater lakes, rivers, streams and wetlands. Freshwater habitats, although only comprising 1 % to the total surface area of Earth, contribute a disproportionate number of vertebrate species and approximately 40 % of all described fish species (Dudgeon et al. 2006, Darwall et al. 2018). Nevertheless, there is a significant paucity in understanding both the rate and effects of biodiversity loss in the freshwater realm which has, in the past lead to fisheries management being reliant upon adaptive measures – lacking baseline information or a thorough understanding of the system and its supported species (Strayer and Dudgeon 2010, Weyl et al. 2010).

Lake Malawi, also referred to as Lake Niassa (11.670°S, 34.686°E), is one such freshwater system that supports high levels of biodiversity whilst simultaneously being exposed to multiple stressors (Weyl et al. 2010). Lake Malawi is the 9th largest lake in the world and the 3rd largest lake in Africa. Although not being the Earth's largest or deepest lake, Lake Malawi supports over half a million people through its various fisheries and boasts the highest level of freshwater fish diversity, with 14 families being described and an estimated 700 species (Turner et al. 2001). Of these 14 families, the single Cichlidae family is responsible for most of the lake's species richness as well as biodiversity (Turner et al. 2001, Weyl et al. 2010). Overfishing and habitat destruction have, however, driven a collapse in local Cichlidae stocks (Palsson et al. 1999a, Banda et al. 2005b, Weyl et al. 2010) and the fish communities subsequently face an exploitation induced regime shift (Irvine et al. 2019). The industrial trawl and artisanal fisheries target approximately 160 species by way of demersal and pelagic fishing practices (Irvine et al. 2002, Irvine et al. 2019). Unsustainable rates of exploitation have, however, since led to the collapse of large, high-value Cichlid species. Perhaps most notable, was the collapse of the local *Oreochromis* (*Nyasalapia*) species flock in the southern waters of Lake Malawi (Weyl 2001, Banda et al. 2005b, Bulirani 2005, Bell et al. 2012, M"balaka et al. 2018). Chambo is the collective term for the species complex including: *Oreochromis lidole*, *Oreochromis karongae* and *Oreochromis squamipinnis* and is believed to have a lake-wide distribution (Turner and Mwanyama 1992).

The Chambo fisheries underwent a significant collapse in the 1980s, the effects of which, continue to have a detrimental effect upon local community livelihoods and the Malawian economy (Banda et al. 2001). At an ecosystem level, the depletion of larger-bodied, k-selected species in Lake Malawi as a result of overfishing may now be playing a role in the dominance of small, fast-growing r-selected species in reported catches (Irvine et al. 2019). Although the change in species catch compositions has not led to a decline in overall caught biomass, substantial dependence upon r-selected species, prone to unpredictable "boom and bust" life histories may prove to be detrimental in the future (Irvine et al. 2019).

In brief, Irvine et al. (2019) propose that ecosystem-based management approaches are to be adopted in Great Lakes, including Lake Malawi. Investment in management programs and techniques which can be both employed at a lake-wide scale and over long-term scenarios must be considered. A fundamental cornerstone to any such an endeavour is the ability to monitor the system under management effectively (Slocombe 1993). To date, the predominant research output of fisheries monitoring in Lake Malawi has been as a result of fisheries dependent surveys and focused heavily upon the artisanal and commercial fisheries operating in the productive of waters of Southern Lake Malawi (Ribbink et al. 1983, Tweddle and Magasa 1989, Alimoso et al. 1990, Reinthal 1993, Irvine et al. 2002, Halafo et al. 2004, Weyl et al. 2010). Although fisheries independent surveys have been conducted in Lake Malawi, the methodology behind their data collection has logistical limitations – particularly when operating across multiple depth ranges and upon structured habitats.

Research trawls first carried out by the Malawi Fisheries Department in 1965 were conducted to determine the potential for commercializing the trawler industry. These surveys have since evolved and now function as demersal stock assessments following the subsequent commercialisation of the trawl fishery (Tarbit 1972). Snorkelling and SCUBA have also since been conducted in Lake Malawi. The non-extractive monitoring technique has been used for the surveying of shallow water cichlid species (Ribbink et al. 1983), for observing fish spawning behaviour (Ribbink et al. 1980, Van Oppen et al. 1998, Smith 2000) and monitoring of translocated species (Genner et al. 2006). Although being incredibly useful and underpinning current policy and management decisions, the current fisheries dependent monitoring techniques may prove to be ineffective for long-term monitoring of proposed protected areas in the lake – a key to ecosystem-based management. Trawlers are limited by the area upon which they can operate and may cause fish mortality or potential habitat destruction. Snorkel and SCUBA surveys, although being relative non-destructive, are logistically limited to shallower depths and require considerable expertise (Harvey et al. 2002).

Demonstrated to be both cost-effective and efficient in the marine realm, baited remote underwater video systems (BRUVS) mitigate the effects of habitat destruction when using trawl nets, fishing mortality linked to extractive fishing techniques and the risks and bias attributed to using diver captured data (Roberts 2002, Merritt et al. 2011, Bernard 2012). Originally used and published by Ellis and DeMartini (1995), the systems are highly customisable, and their configurations (i.e. the number of cameras, construction material and design) may be adjusted according to the environment in which they are employed (Whitmarsh et al. 2017). Their underlying function and purpose, however, remains unchanged. The systems are unmanned and use a form of bait – presented in the cameras field of view, to attract fish into the frame. These fish may then be studied using post-video analysis software (Harvey and Mladenov 2001). The monitoring technology has the potential to be employed in a wide range of habitats, depths and taking into consideration the advancement and affordability of consumer camera technology, is more accessible to researchers than ever before (Letessier et al. 2015, Struthers et al. 2015). BRUVS, if effective in Lake Malawi, may be employed to conduct standardised lake-wide monitoring of fish communities. Including within non-fished protected areas and inaccessible deep-water habitats.

To date, there are few studies which have employed BRUVS in freshwater ecosystems (Whitmarsh et al. 2017). Advancing the technology for freshwater application will, however, require working case studies and should aim to develop and publish standard operating procedures while testing the feasibility of the technology in multiple ecosystems. Namely: lakes, rivers, streams and wetlands. Lake Malawi is one such system and was selected as a model application for BRUVS monitoring in 2016 based upon several factors. These included manageable operating conditions, such as weak currents, clear waters and the unique opportunity to monitor the potential effects of protected areas in an African freshwater lake (Weyl et al. 2016).

The two study areas selected were the Lake Niassa Reserve (LNR), Mozambique and the South Eastern Arm (SEA), Malawi. The two study areas are characterised by differing levels of fishing pressure. Nevertheless, their shorelines share several common fish species, depth profiles and habitat types (Weyl et al. 2010). Malawi's fishery has recorded reduced catch rates and shifts in species catch composition - exemplifying a fishery under significant rates of exploitation (Weyl 2001, Bulirani 2005, Weyl et al. 2010). Mozambique, however, although recording a significant increase in fishing effort between years 1983 to 1999, is yet to experience the equivalent fishing pressure experienced in Malawi (Halafo et al. 2004, Weyl et al. 2010).

The rationale of this thesis was to assess the possibility of using BRUVS as a fish monitoring tool in Lake Malawi. Achieving this rationale required the deploying of BRUVS within the LNR and SEA to develop a standard operating procedure and test the effectiveness of BRUVS to monitor selected species groups. This thesis contains five chapters summarised in Figure 1.1.

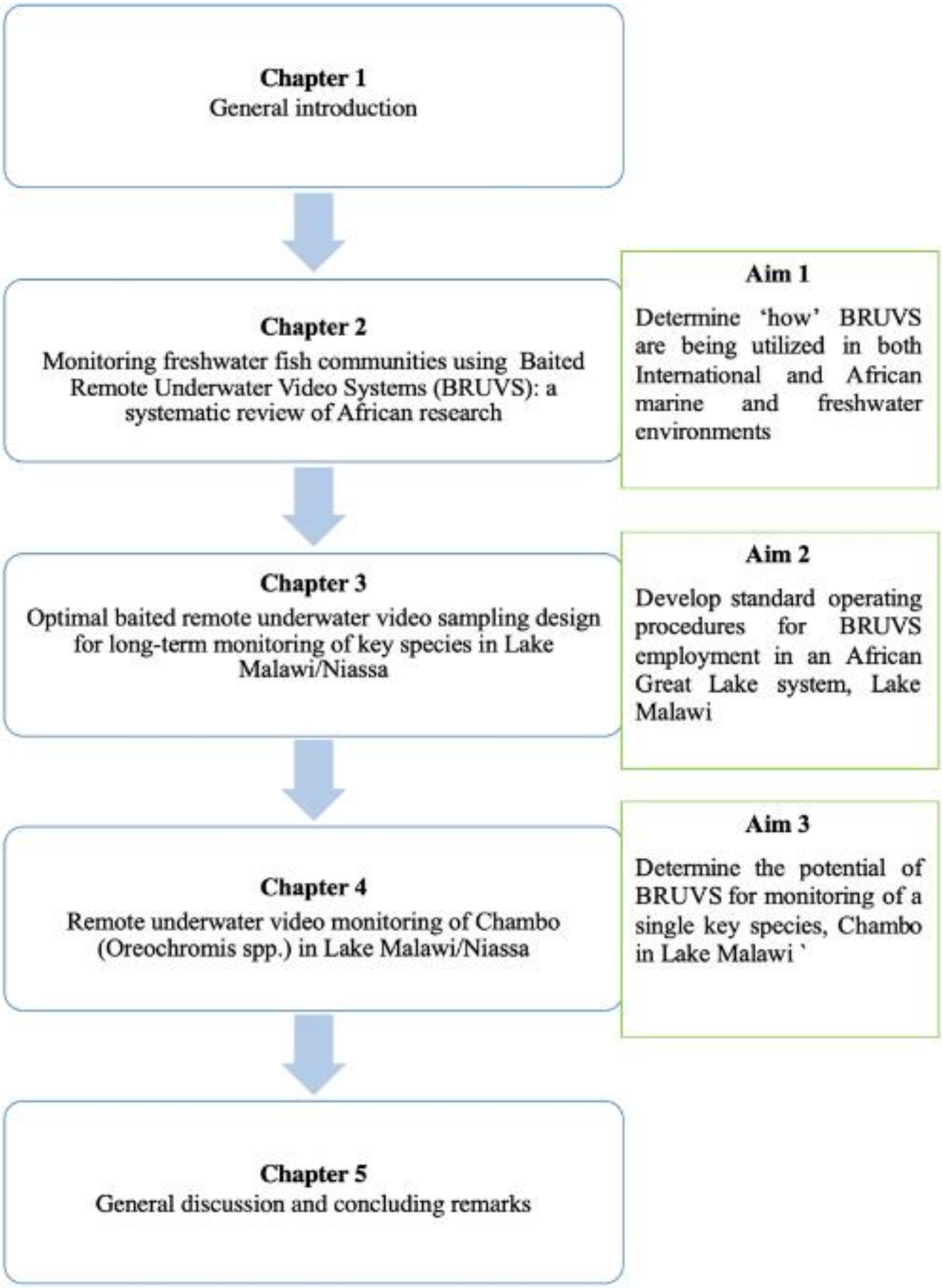
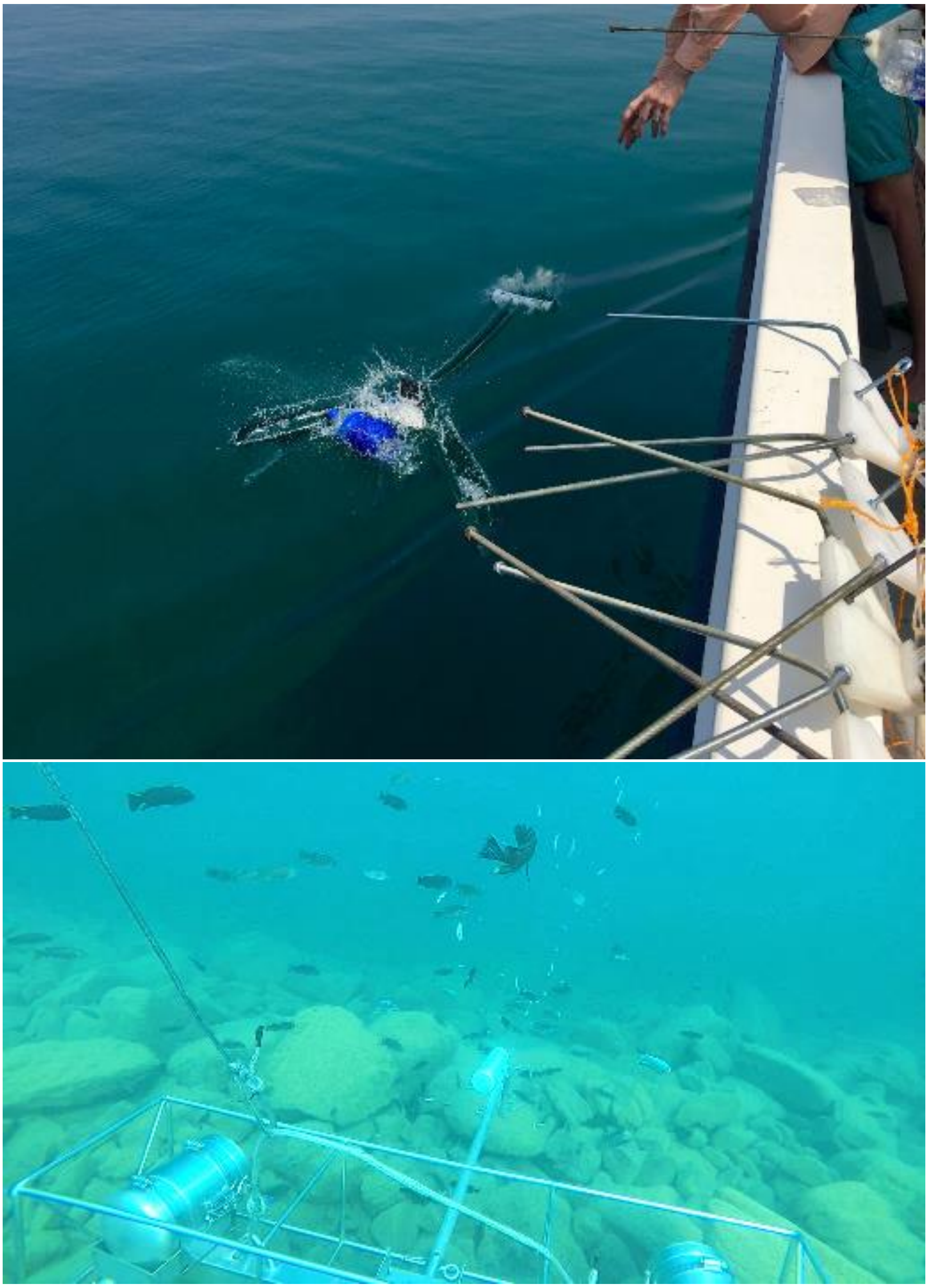


Figure 1.1: Infographic of thesis outline. The titles and ordering of Chapters 1-5 and the overarching aims of each included data chapter are displayed.

Following the General Introduction (Chapter 1), in Chapter 2, I systematically reviewed the current applications of BRUVS between the years 1995 to 2018. This chapter aimed to elaborate upon how BRUVS are currently being utilized globally by researchers and discusses potential considerations for BRUVS employment in African freshwaters. In Chapter 3, I used BRUVS data derived from 8 selected species groups to make recommendations for standard operating procedures in Lake Malawi. In Chapter 4, BRUVS were used to compare size-structuring and relative abundance of a single selected species group, Chambo in Lake Malawi. Chapter 5 is the final chapter of this thesis. In this chapter, I synthesize the study’s main findings and provide a final discussion - paying reference to the potential of BRUVS in Lake Malawi and points to consider for the further development of the technology.

MONITORING FRESHWATER FISH COMMUNITIES USING REMOTE UNDERWATER VIDEO TECHNOLOGY: A SYSTEMATIC REVIEW OF AFRICAN RESEARCH



Deployment of mono-baited remote underwater video system (BRUVS) into Lake Malawi (Top), and underwater scenes from a BRUVS deployment in Lake Malawi (Bottom) P.C. Angus van Wyk (SAIAB)

INTRODUCTION

Water is a fundamental natural resource - supporting most living organisms as well as a vast number of economic practices. Subsequently, it comes as no surprise that freshwater bodies of all forms are exposed to multiple stressors (Cardinale et al. 2012, Jackson et al. 2016). Water abstraction, the effects of urbanization, industrial growth and overfishing are but a few of the immediate stressors placed upon freshwater ecosystems on a global scale (He et al. 2017). When focussing on inland fisheries, these effects are often translated into reduced fish yields which, is of great concern, particularly in developing African communities" whose households are dependent upon the 2.8 million inland fishers who utilize these waters (Ormerod et al. 2010, Jackson et al. 2016, Funge-Smith 2018, Reid et al. 2019). An effort to collect data and better understand the current state of African freshwater bodies and the drivers of potential change is therefore of utmost importance if proactive ecosystem-based management strategies are to be developed (Slocombe 1993). Fish population monitoring or community analysis plays a crucial role in such data collection and forms part of a broader ecosystem component, freshwater biological monitoring.

The importance of biological monitoring, specifically long-term biological monitoring has been the focus of much scientific inquiry (Callahan 1984, Strayer et al. 1986, Franklin 1989). Quantifying environmental issues, gauging the success of policy and achieving evidence-based environmental decision making are all underpinned by a continued collection of quantifiable observations (Callahan 1984). Furthermore, observing both the presence and behaviour of species in their natural habitat continues to fuel biological research and provides invaluable insight into how an organism utilises its environment and how it may respond to future stressors (Block 2005). There is a wealth of information on the concepts of biological monitoring - mainly aimed at terrestrial ecosystems and a broad agreement that the method should be ecologically appropriate, statistically reliable and cost-effective (Karr 1987, Noss 1990, Lindenmayer and Likens 2009). Aiming to achieve these three prerequisites of biological monitoring is not, however, unique to the terrestrial realm but also a focus within aquatic environments (Hilborn et al. 2004, Richardson et al. 2006, Langlois et al. 2010).

Monitoring of fish communities *in-situ* has, however, proven challenging - particularly in inaccessible, remote environments or at great depths (Harvey et al. 2002, Cappo et al. 2006). Habitat destruction, fishing mortality linked to the removal of fish and the risks and bias attributed to using diver captured data are but a few of the present challenges (Roberts 2002, Merritt et al. 2011). Recent technological advancements, particularly in consumer-accessible technologies, continues to drive research towards developing remote non-destructive monitoring techniques. Many of which can more effectively collect *in-situ* data within challenging remote or sensitive habitats and may be able to mitigate several challenges presently faced by traditional monitoring methods (Struthers et al. 2015). Remote monitoring, in the form of unmanned underwater cameras, is one such technology which has become increasingly accessible to researchers and proven effective at collecting data on a range of species and within multiple different ecosystems (Harvey and Mladenov 2001, Whitmarsh et al. 2017).

Struthers et al. (2015) discuss in detail, the market-driven developments which have contributed to the increased effectiveness of the technology for *in-situ* monitoring. The three main factors being (1), reduced monetary cost per unit (2), the robust and compact design of modern cameras – particularly that of consumer action cameras and (3), the quality of footage attainable from the compact and affordable units (Struthers et al. 2015). Underpinned by these factors, researchers have since designed more sophisticated monitoring systems to monitor specific fish communities optimally. Remote Underwater Video Systems (RUVS) and Baited Remote Underwater Video Systems (BRUVS) are two examples of such systems. Initially published in 1995 and developed as a mono-camera system, BRUVS have been employed for a wide range of monitoring applications, and subsequently, their system configuration, design and number of cameras vary (Harvey and Mladenov 2001). All systems do, however, share a standard and fundamental function – to record underwater video footage remotely. Either with or without the use of bait as an attractant.

To gain a better understanding of BRUVS employment and methods reporting, Whitmarsh et al. (2017) completed a comprehensive review of 161 studies, investigating the methods and uses of BRUVS from the year 1950 to the 18th of July 2016. The review had a relatively broad scope and extracted a total of 24 variables from 161 included publications. In brief, the study presented a protocol for authors to include an adequate amount of information relating to recorded variables. Developing such a protocol was based upon the findings by Whitmarsh et al. (2017) that 34 % of all included publications failed to report three or more variables in their studies – hindering both the replicability and comparability of the publications.

The aim of this chapter was to determine how BRUVS are currently being utilized in the freshwater realm - particularly in African freshwater bodies and discuss the technology's potential employment as a monitoring tool in African freshwater systems. This aim was to be achieved by completing the following objectives:

(1) determine present knowledge gaps when employing BRUVS in either the marine or freshwater realm,

(2) provide an overview of the applications of BRUVS, and

(3) investigate the potential of the technology to be employed in African freshwater bodies.

METHODS & MATERIALS

This systematic review followed the PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-analyses) method and was formatted to align with a review previously completed by Whitmarsh et al. (2017). The authors used keywords such as “baited”, “video” and “BRUV” in search engines: ProQuest, Google Scholar, Scopus and Biological Abstracts (Moher et al. 2009). Sourced papers were then tabulated, analysed and graphically presented using Microsoft Excel 2016, version 16.14.1.

The scope for this study was set to (1), update the review completed by Whitmarsh et al. (2017) (2), determine the number of studies which were undertaken in both International and African freshwater systems and (3), to determine trends in system configurations and standard analysis methods. The keywords and search engines used in this chapter followed those used by Whitmarsh et al. (2017) and included all qualifying studies before the 25th of March 2018. Exclusion criteria, as discussed by the PRISMA method, were used to omit grey literature. These are sources defined as information compiled by the government, academic institutions, business and industry in all formats that have not been commercially published or peer-reviewed (Higgins and Green 2005).

RESULTS AND DISCUSSION

Overview of studies

The inclusion keywords “baited”, “video” and “BRUV” in search engines: ProQuest, Google Scholar, Scopus and Biological Abstracts resulted in a total of 251 sources of information being included in the initial review stage. From the 251 sourced papers, 29 publications were omitted following a closer review of the sources and removal of grey literature. The number of publications included in this review totalled 222, and a full table of metadata can be found at the supplementary data link provided in the Appendix.

In summary, there has been a steady increase in the output of studies employing all BRUVS configurations since the original paper by Ellis and DeMartini (1995) (Figure 2.1). This includes field-of-view (FOV) and stereo- or mono-baited remote underwater video systems. Field-of-view systems are those configurations which make use of more than two cameras. Necessary explanatory information relating to a geographical area of study and date of completion are standard in scientific writing and were subsequently included in all studies (Table 2.1). Other important variables relating to BRUVS methods such as bait type, volume and maximum operating depth were, however, omitted from several studies. The most commonly unreported variables were those surrounding bait amount, preparation and the software used to analyse the recorded footage with unreported values of 22.07 %, 22.52 % and 39.19 %, respectively (Table 2.1). Unreported bait values included those studies which reported whole fish numbers only as opposed to a standardisable metric weight or those studies which failed to use a standardised bait type or amount.

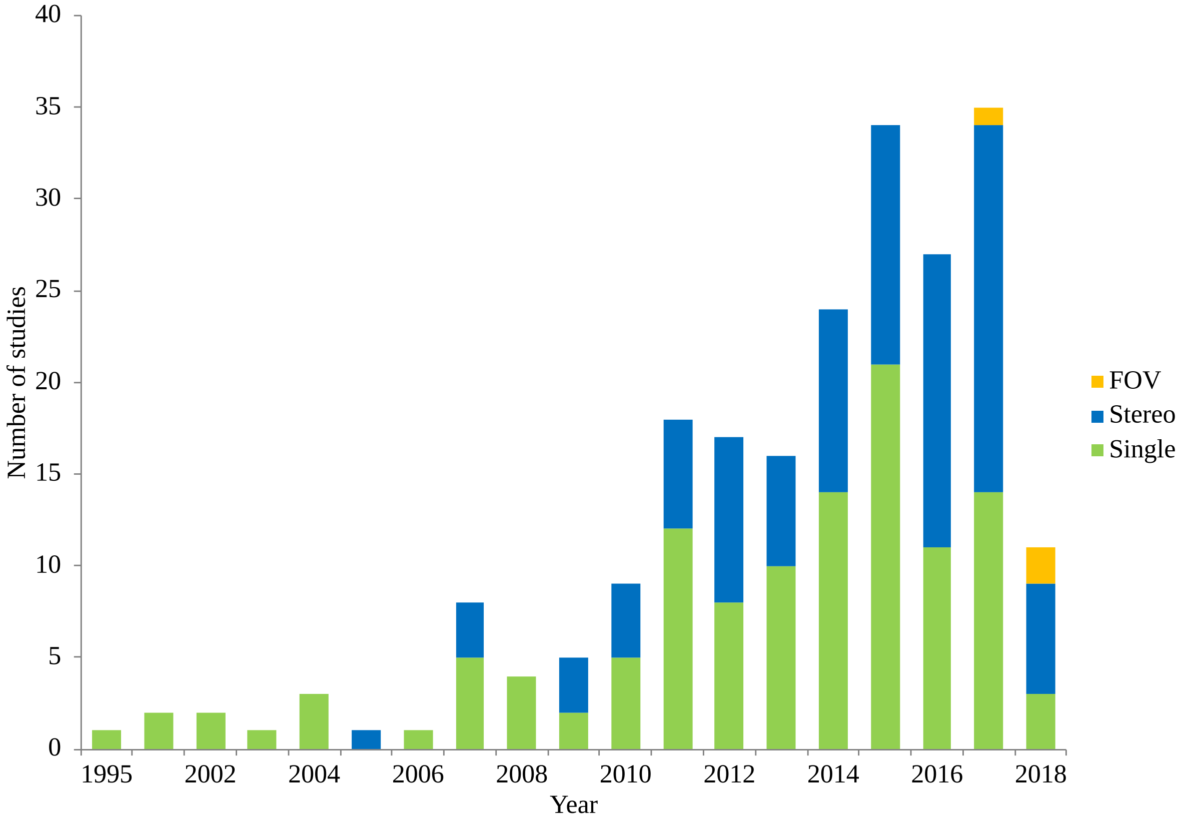


Figure 2.1: The number of studies employing field-of-view (FOV), stereo- or mono-baited remote underwater video systems (BRUVS) from the year 1995 to the 25th of March 2018 in both international marine and freshwater realms.

Table 2.1: List of 15 variables extracted from searched literature and the number of papers within which the variables could not be found from 1995–2018. The number of cameras represented by a single (mono), stereo and multiple (FOV). The metric examples reported are MaxN, T1st and MeanCount, which refer to common abundance metrics discussed by (Stobart et al. 2015).

Variables	Examples	# of studies variable not reported in 222
When and where?		
Study	Ellis and DeMartini 1995	0
Year published	1995, 2017, 2018…	0
Country of study	USA, UK, South Africa	0
Country development status	Developed, developing	0
Geographical Zone	Tropical, temperate, polar	0
Aquatic realm	Marine, estuarine, freshwater	0
System and methodology		
Number of cameras	single, stereo, FOV	5
Maximum soak time (minutes)	<30, 30-90, >90	10
Bait type	Fish, vegetable mix, non-fish	16
Maximum bait amount (grams)	<500, 500-1000, >1000	49
Preparation method	Whole, processed	50
Maximum working limit (meters)	0-49, 50-100, >100	30
Analysis		
Metric used	MaxN, T1st, MeanCount	5
Software used	EventMeasure, ImageJ	87
Studied taxa	Teleosts, chondrichthya	9

Bait type has a direct effect upon the outcome of the study as well as the volume and presentation (Harvey et al. 2007, Bernard 2012, Hardinge et al. 2013, Ghazilou et al. 2016). High feeding rates of densely populated reef systems may require greater amounts of bait compared to systems of lower population densities (Hardinge et al. 2013). Nevertheless, a standard of 1kg is recommended for the sake of comparability and to permit the merging of study data (Hardinge et al. 2013). Given the recent findings on the effect of bait preparation, volume and subsequent plumes upon the observable fish community, controlling, standardising and reporting all information on bait use in a study should be prioritised (Harvey et al. 2007, Bernard 2012, Hardinge et al. 2013, Ghazilou et al. 2016). The results of this summary echo the findings of Whitmarsh et al. (2017) and highlight the need for a standardised set of variables or metadata formatting to be included in-text or appended to all published work employing BRUVS.

BRUVS vs RUVS?

A principal aim in this review was to investigate those studies which employed remote underwater systems which utilised bait as an attractant. RUVS or remote underwater video systems differ by their lack of both bait and bait delivery assembly. The effects of using bait to draw species into the frame are significant. Bait type, method of preparation and the amount of bait will have a profound effect on the observed abundance of fishes (Dorman et al. 2012, Hardinge et al. 2013, Ghazilou et al. 2016). This increase in the likelihood of detection is ordinarily favoured in fish monitoring projects as it results in more stable abundance data and quantifying present species with relatively lower sampling effort - a costly and logistically challenging aspect of system deployments (Cappo et al. 2006, Bernard 2012).

Although useful when utilised appropriately, bait for remote video recording may not be suited to confined systems or when a projected aim requires a degree of inconspicuousness (Ebner et al. 2014). Such requirements are standard in laboratories, aquaculture systems and field applications investigating natural behaviours (Svensson et al. 2012, Zion 2012). These applications may not need to attract individuals into the frame, and the bait may understandably become a confounding variable (Svensson et al. 2012). Remote underwater video systems are often very similar in construction, camera configuration and video analysis methodology to BRUVS. Caution must, however, be taken when comparing the two systems, and determining which system configuration may be appropriate to a study (Harvey and Mladenov 2001).

Sampling effort, deployment time, logistical costing and ethical considerations will differ significantly between the two systems (Bernard 2012). Much like their baited counter-parts, however, employing RUVS has become increasingly popular, and their applications are broadening (Struthers et al. 2015). One such application maybe employing RUVS as an alternative to freshwater electrofishing or determining the presence-absence of fish species following river restoration projects (Ellender et al. 2012, Weyl et al. 2013).

Where are BRUVS being employed?

The manipulability and customising capabilities of BRUVS as a monitoring tool has allowed for the technology to aptly monitor a suite of differing fish communities (Willis et al. 2000, Watson et al. 2005, Cappo et al. 2006, Harvey et al. 2007, Langlois et al. 2010, Schmid et al. 2017). Subsequently, the technology has been employed to observe underwater communities across a wide range of geographical locations, depths and within all aquatic realms (Table 2.2). The technology is currently being utilised with similar effort across all geographical climates with the exception of the polar regions (comprising only 1 %) and represented internationally in countries such as the Philippines, Belize and Indonesia (Jaiteh et al. 2016, Clementi 2017, Abesamis et al. 2018). This ubiquitous effort across geographical climates is not, however, evident in geographical location or across aquatic realms.

Australia has in the past and to date, continued to dominate the publication output of BRUVS studies (Table 2.2). The single country has been responsible for 55 % of the output since 1995 and has subsequently contributed a significant amount of research and infrastructure toward the fast-growing technology. This apparent dominance may be as a result of the pioneering BRUVS researchers originating from Australian and operating within their nations" maritime boundaries (Harvey and Shortis 1995, Cappo et al. 2001, Cappo et al. 2003, Harvey et al. 2003). Including Australia, there is also an evident dominance of BRUVS applications in other developed nations such as New Zealand and the United Kingdom (Table 2.2). The combined dominance of developed nations has since resulted in developing countries contributing only 12 % of reported BRUVS studies.

Bernard (2012) and Halse (2016) discuss the importance of the BRUVS technology being both logistically and cost accessible as well as standardisable across studies in the context of a developing nation. At only 12 % of the study output and with the entire continent of Africa contributing only 4 %, developing nations are not equally accessing the monitoring technology. The same sentiment is shared for the technology's application in the different aquatic realms. 96 % of publications have employed BRUVS to study marine habitats or their occupants, and only 3 % have studied estuarine systems (Table 2.2). A total of 3 studies (contributing 1 %) have utilised BRUVS in the freshwater realm and were conducted in two countries, Brazil and Australia between the years 2013-2018 (Ebner and Morgan 2013, Ebner et al. 2015, Schmid et al. 2017). Although there are a small number of published RUVS applications in freshwater and Africa, there is still an evident dominance of research performed in both marine environments and developed nations.

Table 2.2: Percentage of all reviewed studies from 1995–2018 employing baited remote underwater video systems (BRUVS) by country, development status, geographical climate and aquatic realm.

Variables	Where?	% of studies
Top publishing countries		
	Australia	55
	Hawaii	5
	New Zealand	5
African Countries published		
	South Africa	4
Published country development status		
	Developed	83
	Developing	12
Geographical Climate		
	Tropical	30
	Sub-tropical	31
	Temperate	38
	Polar	1
Aquatic realm		
	Marine	96
	Estuarine	3
	Freshwater	1

How are BRUVS being designed and deployed?

In the nine years following the study by Ellis and DeMartini (1995), all applications of BRUVS used a single camera only (Table 2.3 & Figure 2.1). As a monitoring tool, BRUVS with a single camera present themselves as a more straightforward and cheaper opportunity to monitor fish communities (Langlois et al. 2010, Bernard 2012). From a single camera, observers can attain a wealth of information from the footage and can make inferences on species presence, behaviour, abundance and habitat coverage (Cappo et al. 2003, Malcolm et al. 2016, Roberson et al. 2017, Rolim et al. 2019). A trade-off, however, exists between the cost and simplicity of the system and the ability of the BRUVS to collect *in-situ* stereo data (Bernard 2012).

By having two or more cameras, observers can calculate lengths of fish, make more robust inferences on biomass, better quantify habitat composition and potentially track movements of individuals in space (Dell et al. 2014, Barley et al. 2017, Abesamis et al. 2018). The extra camera/s does, however, create higher costs and may require more specialised software to both calibrate and analyse the footage.

Although the contribution of mono-BRUVS has, to date produced a higher output (55 %) compared to stereo-BRUVS (44 %), there is a rising trend towards the latter, costlier configuration (Table 2.3 & Figure 2.1). Since 2016, 58 % of studies have employed stereo-BRUVS and for the first time, three studies investigated using a baited system with multiple (> 2) cameras to create an overlapping field-of-view system (Kilfoil et al. 2017, Campbell et al. 2018, Reynolds et al. 2018). There also exists a trend in both the maximum deployment time of BRUVS as well as their maximum operating depths. The majority of BRUVS are deployed to record for between 30 and 60 minutes (71 %), and most BRUVS are operated at depths shallower than 50 m (54 %). The former may be as a result of continued suggestions by authors to converge towards standard BRUVS operating procedures while overcoming the challenges which exist when working at depth continue to limit the expansion of BRUVS into deep waters (Bernard 2012, Halse 2016).

Table 2.3: General sampling methods (maximum soak time, maximum deployment depth, bait type and bait preparation) and baited remote underwater video systems (BRUVS) configuration (number of cameras) displayed as a percentage of all reviewed studies from 1995 – 2018.

Variable	How?	% of studies
Number of cameras		
	Single	55
	Stereo	44
	Multiple	1
Maximum soak time (minutes)		
	>90	19
	30-60	71
	<30	5
Bait type		
	Fish	83
	Non-fish (Squid, mussels, oils etc.)	6
	Vegetable mix	4
Maximum bait amount (grams)		
	>1000	4
	500-1000	59
	<500	12
Preparation method		
	Whole	9
	Processed (Crushed, chopped, mashed etc.)	68
Maximum deployment depth (meters)		
	>100	10
	50-100	18
	0-49	54

What techniques are being used to analyse BRUVS data?

The now broad utilisation of BRUVS and the various taxa they are designed to observe has created a demand for specialist software and continues to fuel research into reliable and efficient metrics (Schobernd et al. 2013, Stobart et al. 2015). The most commonly used metric reported since 1995 is a relative abundance estimate, MaxN (80 %) (Table 2.4). The metric requires counting the maximum number of identifiable individuals in a single video frame during a defined viewing interval (Ellis and DeMartini 1995). MaxN is considered to be a relatively easy to attain value from extended lengths of video footage and decreases the probability of the observer counting single individuals more than once (Schobernd et al. 2013, Stobart et al. 2015). MaxN is therefore considered to be a conservative index or a precautionary approach as it is unlikely to overestimate the true abundance of observed individuals. It must be noted, however, that using bait as an attractant will draw in fish from surrounding waters that may not ordinarily inhabit the studied area. Thus, increasing the number of observed individuals and potentially over-estimating the true abundance within each deployment (Ellis and DeMartini 1995).

Other standard metrics reported were T1st (10 %), those that studied behaviour (5 %) and MeanCount (1 %). T1st calculates the time at which an individual first enters the frame and is dependent upon the speed at which the individual moves from its start point and into the field of view (Stobart et al. 2015). The assumption is then, if a species is highly abundant in an area, it should appear in an earlier frame than if it was present in a low abundance. Understandably, T1st is highly dependent upon both an individual’s distance to the BRUVS as well as its behavioural response to the bait and system itself (Stobart et al. 2015). MeanCount is a relatively new metric introduced and considered to be a possible alternative to MaxN (Stobart et al. 2015). MeanCount requires multiple frames within a defied viewing interval from which individuals are counted, and an average of all frames is calculated. These frames can be selected at random by an observer, or the process can be automated by specialised software such as EventMeasure (Stobart et al. 2015). A key strength of MeanCount is its ability to track traditional metrics of catch per user effort (CPUE) better than the most widely used metric, MaxN. MeanCount has, however, been suggested to lack the precision able to be obtained by using MaxN and reduces the probability of detection of rare or fleeting species as the analysis does not include all video footage (Schobernd et al. 2013, Stobart et al. 2015).

Table 2.4: Techniques used for the analysis of baited remote underwater video system (BRUVS) footage, displayed as a percentage of all reviewed studies from 1995 – 2018.

Variable	What?	% of studies
Top metrics used		
	MaxN	80
	T1st	10
	Behaviour	5
	Habitat	4
	MeanCount	1
Top software used		
	EventMeasure	36
	PhotoMeasure	6
	BRUVS1.5.mdb	5
Taxa included in the study		
	Teleosts	84
	Chondrichthya	57
	Crustacea	11
	All mobile taxa	8

The majority of footage attained from BRUVS deployments since 1995 have been analysed using specialist software, EventMeasure (36 %) and its predecessor Photo Measure (6 %) (Table 2.4). EventMeasure is a payed-for software which has been explicitly designed to log and report events occurring within video footage. It can also be used to determine fish lengths as well as generate MaxN and MeanCount abundance estimates. The popularity of the software has grown with time, and since 2016, 57 % of reported BRUVS studies utilise EventMeasure. Combined freeware (such as VLC, ImageJ, Mac Media Player) made up only 11 % of the used analysis software. The shift away from generalist video player software may come from the increased usage of multi-camera systems, their stability and a growing need for more robust software. It must be noted that Macintosh-based freeware, VidSync has since been developed to view, record, and organise both 2D and 3D imagery (Neuswanger et al. 2016). The application is also capable of calibration and collecting length measurements from synchronised camera footage. Freeware such as VidSync, may prove useful in alleviating the costs of employing remote imagery systems and potentially aid in access to and the development of the technology moving forward (Neuswanger et al. 2016).

BRUVS in freshwater

To date, techniques used to monitor freshwater fish communities (such as lakes, reservoirs, streams and rivers) have been predominantly extractive or fisheries dependent in nature. Such techniques include gill nets, electrofishing, fyke nets, angling and pot gear. (Hubert et al. 2012). Although being cost-effective, successfully used to monitor multiple species and employed within different ecosystems, these types of gears have serious drawbacks linked to fish mortality and habitat destruction (Hubert et al. 2012). Remote underwater video systems have, in direct response to these shortcomings, gained popularity and their applications broadening in the freshwater realm (Struthers et al. 2015). One such application of camera systems challenging traditional sampling techniques has since been published by Ellender et al. (2012). The publication investigated the potential for remote camera systems to sample headwater stream fishes as an alternative to electrofishing. In summary, Ellender et al. (2012) state that the abundance estimates of the two sampling methods were significantly correlated and that the camera systems had a higher overall detection rate compared to electrofishing.

Demonstrated to be both practical and logistically feasible, RUVS have the potential to replace or bolster traditional monitoring techniques employed by fisheries management and researchers (Ellender et al. 2012, Weyl et al. 2013, Struthers et al. 2015). This statement may also be appropriate for the employment of their baited variants, BRUVS. In theory, BRUVS in freshwater habitats should be able to be employed in similar roles to the marine sector and function as efficiently. More studies are highlighting the potential for BRUVS to be used in the freshwater realm – such as within inaccessible rainforest streams. Ecosystems too dangerous for snorkel surveys may benefit from the employment of BRUVS. Which, have since been used to effectively monitor high numbers of fish taxa in Amazonian rivers (Ebner et al. 2015, Schmid et al. 2017).

Reviewing the current freshwater BRUVS publications, BRUVS studies were all aimed at calculating relative abundance measures or species richness and all deployments were shallower than 10 m (Ebner and Morgan 2013, Ebner et al. 2015, Schmid et al. 2017). Subsequently, although the overall trend in BRUVS usage is towards multi-camera BRUVS, the three studies were all able to employ relatively simple mono-BRUVS. From the analysed footage, the studies used the aforementioned metrics, MaxN and T1st as well as quantifying species richness. Interestingly, none of the studies made use of specialized, payed-for-software. The studies made use of either VLC, Windows Media Player or Mac Media Player as given the analysis performed, this software would be adequate and exists as a more affordable option. Like most publications in the marine realm, all three studies used an oily fish such as pilchard. Schmid et al. (2017) did, however, test the efficacy of different bait types such as cat food and sweet corn and concluded that crushed sardine yielded the most efficient recording of species richness, relative abundance and exclusive species.

BRUVS in developing countries

Baited remote underwater video systems are proposed to be a strong contender to more traditional monitoring techniques as they offer a suite of attributes which afford them the ability to monitor inaccessible and sensitive underwater habitats (Langlois et al. 2010, Bernard 2012). Furthermore, BRUVS are suggested to exist as a relatively robust and inexpensive option to monitor fish communities which makes it a feasible technology and available to institutions in developing countries (Ellis and DeMartini 1995, Langlois et al. 2010, Bernard 2012). The technology also presents itself as being readily customisable towards the habitat in which it needs to operate. The question is then, why have so few studies furthered the technology into the freshwater realm and, why are developing countries so heavily underrepresented in the field at only 12 % of 222 studies published since 1995?

Perhaps overcoming the logistical challenges to employing BRUVS remains too big a task in those countries or institutes which lack both technical capacity and monetary capital. Bernard (2012) calculated the costs of employing BRUVS while investigating the technology's ability to monitor subtidal reef fish in the Agulhas ecoregion of South Africa. The total cost of employing four mono-BRUVS equated to USD 9,460 while the same number of stereo-BRUVS would cost USD 32,231. These figures include the cost of an EventMeasure software key, which in the case of the stereo-BRUVS contributed to almost half the overall total cost (Bernard 2012). Other challenges facing remote underwater video analysis is the actual analysis of the video footage itself, storage of raw video and the initial requirements for employing BRUVS for monitoring. Analysing the video footage requires significant working hours by trained individuals – exacerbated by the need to develop both a working species identification library for each system and generating research upon standard operating procedures. These overcomings should not, however, deter research in freshwater systems. Baited remote underwater video systems are currently both cost-efficient and becoming increasingly feasible compared to other non-destructive monitoring techniques such as underwater visual consensus and underwater vehicles (Bernard 2012, Ebner et al. 2015, Schmid et al. 2017).

CONCLUSIONS AND RECOMMENDATIONS

Remote monitoring technologies such as BRUVS can bolster fish monitoring programs at a seemingly global scale. The camera systems, in their various configurations, have been successfully employed across multiple geographical climates, at depths over 100 m and may serve as a viable alternative to extractive monitoring techniques. There is an evident paucity, however, in the technology's advancement into freshwater systems – particularly in Africa.

The new development of BRUVS for freshwater-specific applications in Africa may benefit significantly from knowledge gained by the marine sector. Steps could be taken by future studies to explore the possibilities of merging the technologies currently employed by marine research and invest in optimisation for freshwater systems – as opposed to developing a completely novel freshwater-specific technology in isolation. Doing so may present opportunities to reduce costs and steer initial investments into tried and tested system designs and analytical software and not the costly venture of prototype system design or software research and development. Achieving this would require a clear understanding of the challenges faced by the marine sector, particularly regarding the establishment of standard operating procedures, the minimum set of metadata published by researchers and overcoming the logistical costs attributed to employing BRUVS.

The upcoming chapters in this thesis aim to address the African freshwater BRUVS knowledge gap – taking the first steps towards standardising operating procedures and investigating the potential for the technology to monitor fish species in Lake Malawi/ (11.670°S, 34.686°E). Both light-weight mono-BRUVS and marine-designed stereo-BRUVS were employed to monitor lakeshore fish species at a maximum depth of 64.90 m and over a three-year sampling period (2016-2018). Closing these gaps and making the BRUVS technology more appropriate for the African Great Lake will provide a means for fisheries management to overcome the present limitations of extractive techniques and further bolster the repertoire of monitoring tools available to local researchers – particularly in sensitive protected areas.

CHAPTER 3

OPTIMAL BAITED REMOTE UNDERWATER VIDEO SAMPLING DESIGN FOR LONG-TERM MONITORING OF KEY SPECIES IN LAKE MALAWI/NIASSA

Busy underwater scenes captured by mono-BRUVS in the Lake Niassa Reserve, Mozambique (Top), image of a curious Kampango in the Lake Niassa Reserve (Bottom). P.C. Angus van Wyk (SAIAB)

INTRODUCTION

Lake Malawi/ (11.670°S, 34.686°E), the most southerly of the African Rift Valley Lakes, is the third-largest lake in Africa (1500 km shoreline) and is bordered by three different countries (namely: Malawi, Mozambique and Tanzania) (Eccles 1974, Ribbink 2001). Although not the largest lake in Africa, what sets Lake Malawi apart from its continental counterparts is the lake's unsurpassed species biodiversity. The high biodiversity is largely due to the Cichlidae family which exploit almost all available habitat (Danley and Kocher 2001). The rapid speciation event that took place thousands of years ago is now responsible for the plethora of fishery-related industries making use of the lake today. Those species not targeted by either the artisanal or industrial fisheries play an essential role in either the global aquarium trade or in supporting local tourism industries (Weyl et al. 2010).

The current state of the lake's freshwater fisheries may be exemplified using the southern region of Lake Malawi. This area has the highest potential for productivity, is subjected to the most significant level of fishing pressure and has been relatively well monitored at a single-species level. A monitoring system which began in the 1960s and more recent efforts by researchers such as Weyl. (2003), Banda et al. (2005) and Bell et al. (2012) have since shed light on the challenges facing the region's fisheries. Many of these challenges are as a direct result of over-exploitation and include decreasing rates of catch, depletion of higher-value species and a general decline in species biodiversity (Weyl et al. 2010).

Further north on the Mozambique lakeshore, Halafo et al. (2004) observed an increase in both numbers of gears and gear owners over the years 1983-1999 in the country's primary fishing centre of Metangula. This increase in the fishing effort has since been shown to have continued at a broader scale along the Mozambique lakeshore, and a general decline of inshore and demersal fisheries catch per user effort (CPUE) has been reported (Weyl et al. 2017). The artisanal fisheries operating in Mozambique waters did not, however, develop at the same rate, nor reach a level equivalent to the lake's southern region. Relatively low population densities, weak market forces and over a decade of civil war meant that the fishing pressure along Mozambique's shoreline has remained relatively low (Halafo et al. 2004).

Malawi has taken steps to mitigate the adverse effects of overfishing and protect their own local fish fauna. Traditional input regulations including closed seasons, licence limits and gear restrictions have been implemented at a national policy level. Governed by the legal framework composed of the Fisheries Conservation and Management Act (1997), the Local Community Participation Rules (2000) and the Fisheries Conservation and Management Regulations (2000). The net success of the legal framework has, however, been questioned as all non-commercial fishers remain mostly unregulated and subsequently, the fishery still exists as a predominantly open-access system (Banda et al. 2001, Weyl 2001, 2005, Weyl et al. 2010, Hara and Njaya 2016). Investment into fisheries research by both states does, however, continue through various government institutes and non-profits including the Malawi Fisheries Research Institute, Malawi Department of Fisheries, Mozambique Fisheries Research Institute (Instituto Nacional de Investigação Pesqueira), Institute for Small Scale Fisheries Development (Instituto Nacional de Desenvolvimento da Pesca de Pequeno Escala), the World Wildlife Fund and the Food and the Agricultural Organisation of the United Nations.

Apart from fishery effort controls, another conservation measure undertaken by both states is the demarcation of protected areas. On the Malawian lakeshore, the Lake Malawi National Park was gazetted in the 1980s with the overarching aim to protect and better manage the freshwater biodiversity found within the park's large bay and surrounding rocky island habitats (Bootsma 1992). In theory, they are creating an area free from or with significantly lowered fishing pressure. Similarly, the Lake Niassa Reserve (LNR) was announced in 2011 and exists as the first and only protected freshwater lake system in Mozambique. Before this study, traditional monitoring techniques such as netting, fishing and trawl surveys were employed to monitor these delineated waters (Banda et al. 2005a). Although such monitoring methods are often sufficient and the physical examination of species for biological surveys is important, these techniques have obvious limitations for long-term monitoring of both sensitive or newly restocked habitats within protected areas. Habitat destruction, fishing mortality and the risk of stunting future recruitment are fundamental illustrations of these limitations (Merritt et al. 2011, Roberts et al. 2016). Addressing these monitoring challenges in the lake's protected waters shall require broadening the repertoire of non-extractive sampling techniques currently employed by fisheries researchers.

Underwater video recording systems are fast becoming readily available to the general consumer and function efficiently as relatively non-destructive monitoring tools. Chapter 2 discusses the key attributes of the technology and the non-extractive nature and cost-effective manner in which the systems can collect data. Encouraging a shift away from traditional monitoring techniques and driving advancement in remote observation and monitoring technologies (Alder et al. 2008, Langlois et al. 2010, Bernard 2012). Baited remote underwater video systems (BRUVS) are one such technology (Cappo et al. 2001).

There is currently a void in research surrounding the use and applicability of BRUVS in freshwater bodies, particularly in Africa. These systems have been used to significant effect in the marine realm as a long-term monitoring technique, and a major strength of the technology and benefit to the scientific community lies in the generation of long-term data sets and the ability for repeated post-analysis using the stored video recordings. The strength of the methodology is, however, underpinned by the comparability between each deployment. Essentially, deviations in standard operating procedures (SOPs) (such as bait type, amount of bait and deployment times) between studies or institutions reduces the likelihood that the data can be analysed as a single dataset. Thus, nullifying a core strength of the technology.

Considering the potential of BRUVS and the absence of any SOPs to direct standardised BRUVS research in freshwater systems, this chapter aimed to utilise a three-year BRUVS dataset (2016-2018) collected in the South Eastern Arm of Lake Malawi (Malawi) and Lake Niassa Reserve (Mozambique) and take the first steps towards developing an appropriate freshwater BRUVS sampling SOP. The objectives to be accomplished were:

(1) identify species which were able to be monitored effectively using remote video and compile a common-species checklist,

(2) to determine adequate BRUVS sampling effort required for long-term abundance monitoring upon selected species, and

(3) to determine an optimal BRUVS deployment time for monitoring the selected species.

MATERIALS & METHODS

Study areas

Lake Niassa Reserve, Mozambique

The Lake Niassa Reserve (LNR) is situated on the eastern-central lakeshore of Lake Niassa, bordering the single country of Mozambique (Figure 3.1). The Mozambique shoreline is approximately 245 km in length. The selected study sites were characterized by three dominant shoreline categories, namely: rocky, vegetated banks and sandy beaches (Table 3.1) (Halafo et al. 2004). The Niassa province of Mozambique, which borders the LNR and has an estimated population of 1,027 037 people, is the country's most sparsely populated province and experiences distinct rainfall seasons. The rainy season occurs annually between November and April and the dry season between May and October (Halafo et al. 2004). Although exposed to freshwater flooding and strong surface winds, Lake Malawi is a meromictic body of water and the oxic-anoxic boundary existing at ±250 m remains fixed due to a thermo-chemical gradient

(Eccles 1974).

The LNR has an area of 478 km^2 and extends 4.83 km offshore from the Mozambique shoreline. The northernmost study site selected was offshore of the town Cobue, and the furthest south was offshore from Meluluco (Table 3.1). The LNR"s delimited waters experience relatively warm surface temperatures ranging from 24-29°C and the lake's water is considered to be alkaline and ranges from pH 7.7–8.7 (Eccles 1974). It must be noted that relative to the productive waters of the southern Malawi lakeshore, the LNR is relatively data-poor. This paucity in data includes both long-term multispecies catches as well and community demographics data.

In the pioneering fisheries research by Bernacsek and Massinga (1983), a total of 42 fishing centres were identified on the Mozambique lakeshore of what is now known as the Lake Niassa Reserve. The researchers estimated the total annual catch of Metangula (the reserve's principal fishing centre) was roughly 1690 tons/year and landed by approximately 3382 fisherman and 1203 operating fishing boats (Bernacsek et al. 1983). This insightful study was followed by a 15-year dark period whereby political unrest played a significant role in the lack of fisheries monitoring in Lake Malawi. Monitoring would continue in 1998 by the Instituto de Investigação de Pesquera (IIP) and later form the part of the research published by Halafo et al. (2004). The 2004 study reported that there had been a significant increase in fishing effort throughout the studied fishing centres between the years 1983 and 1999 (Halafo et al. 2004).

Halafo et al. (2004) investigated the catch of six active fishing centres in the Lake Niassa Reserve, two of which are study sites in this study. Metangula (12.696°S, 34.816°E) and Meluluco (12.907°S, 34.767°E) are both active fishing centres (Table 3.1 & Figure 3.1). Although motorised vessels are in operation, the fishery is dominated by artisanal fishers who, employ a wide range of gears – including chilimera nets, long lines and gill nets (Halafo et al.

2004). These fishers target multiple rock-dwelling and pelagic cichlid species, local catfish (Mlamba) and a schooling pelagic cyprinid, Usipa (*Engraulicypris sardella*). Cobue (12.140°S, 34.760°E) is the northernmost study site and similar to both Metangula and Meluluco in terms of fisher gears and target species. The scale of the fisheries is, however, smaller by comparison. Access to the fishing centre via road remains a challenge and the town only recently (in 2016) received a connection to the national power grid. South of Cobue is Nkwichi Bay. Nkwichi (12.209°S, 34.707°E) is de facto closed to fishing due to negotiations between a local lodge in the bay and the local community (van Wyk et al. 2017).

Table 3.1: Summary of selected study sites in the Lake Niassa Reserve, Mozambique. Displaying location, dominant bottom types, mean depth sampled, lake-wide fishing pressure relative to all study locations and total baited remote underwater video systems (BRUVS) deployments (Halafo et al. 2004, Weyl et al. 2010).

Site	Coordinates	Dominant bottom type(s)	Depth sampled (m) (*Min, Mean, Max*)	Lake-wide relative fishing Pressure	Total BRUVS deployments (2016-2018)
Cobue	12.140°S, 34.760°E	Sand (>60 %)	1.1, 13.0, 48.6	Low	178
Nkwichi	12.209°S, 34.707°E	Sand, Sand reef	1.5, 10.2, 42.5	Closed	152
Metangula & Chuanga	12.696°S, 34.816°E	Reef, Sand	0.8, 14.4, 63.1	Moderate	377
Meluluco	12.907°S, 34.767°E	Sand, Reef	1.0, 11.7, 64.9	Low-moderate	331

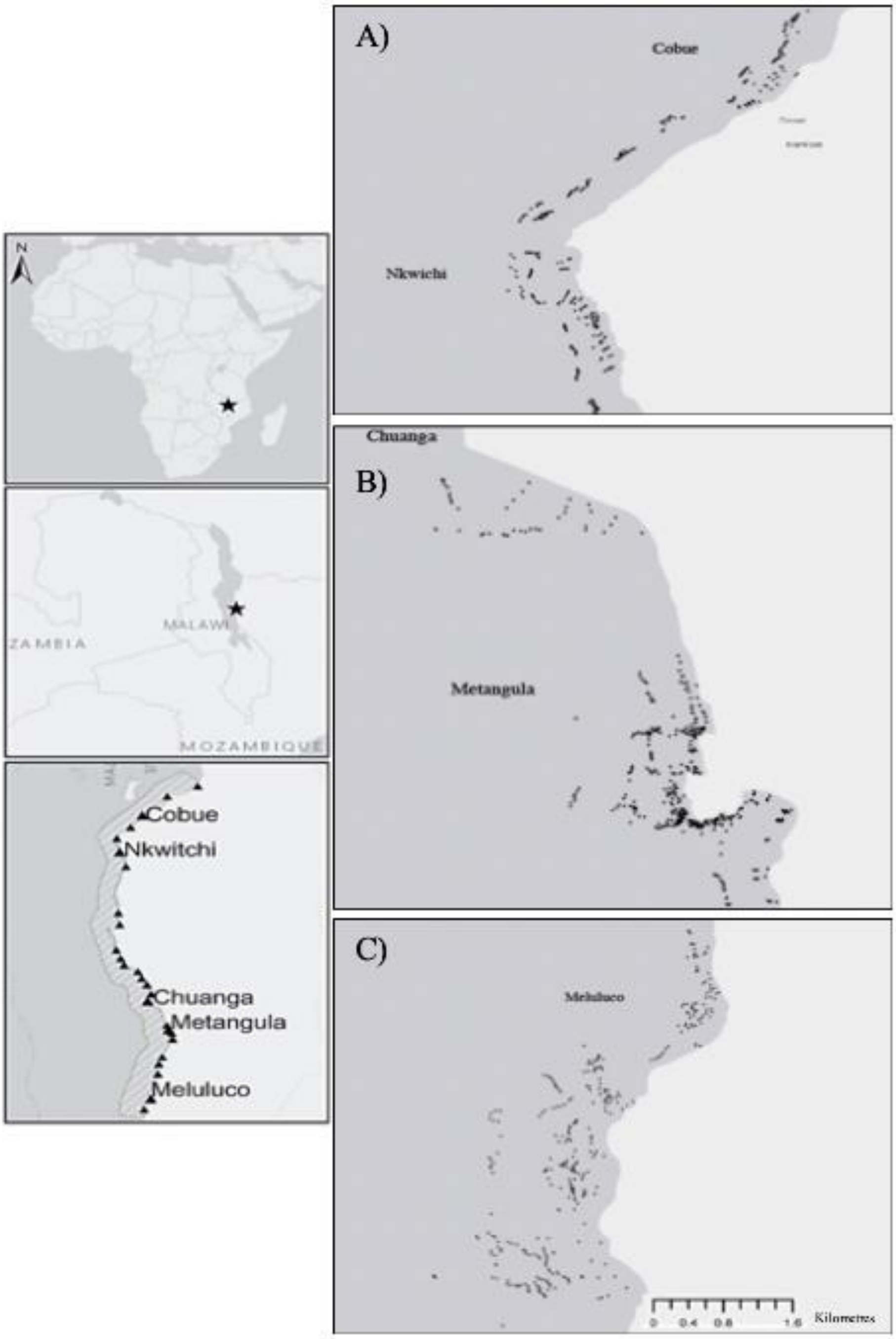


Figure 3.1: Map displaying baited remote underwater video systems (BRUVS) deployments in the Lake Niassa Reserve, Mozambique between years 2016-2018. A) Displaying the most northern study sites, Cobue and Nkwichi B) Chuanga and Metangula and C) the southernmost study site, Meluluco. Symbol (●) represents individual BRUVS deployments and the symbol (▲) represents recognised fishing centres.

Lake Malawi South East Arm, Malawi

The southern region of Lake Malawi, with a surface area of ±3000 km^2 can be further divided into two distinct areas. Namely, the South Western Arm (SWA) (1 210 km^2) and the South

Eastern Arm (SEA) (1 820 km^2). The Mangochi District, which borders the entirety of SEA is of great fisheries importance as it is the production powerhouse of Lake Malawi (Ngochera 2001, Weyl et al. 2004). This district alone has the potential to produce 60 % of the lake's total fish yield (Weyl et al. 2010). All Malawi study sites are within this district and subsequently, the SEA (Table 3.2).

Cape Maclear (14.014°S, 34.849°E) is situated within the Lake Malawi National Park and was declared a protected area in the 1980s. The study site has the lowest experienced fishing pressure of all Malawi study sites. The protected areas within the Lake Malawi National Park are, however, primarily positioned to protect the shoreline reef fish communities within the large bay. Subsequently, vast expanses of deep-water sand habitat near to the protected areas remain exposed to the fishery (Banda et al. 2005a). Monkey Bay (14.071°S, 34.917°E) is a state military harbour and commercial port south of Cape Maclear. Vessels entering the bay are strictly monitored and the setting of nets in the bay is challenging due to the high number of operating motorised vessels (Figure 3.2). Reef and sand habitats directly outside of the bay area are, however, targeted by the fishery. The most exposed study site selected in Malawi was Nkudzi Bay (14.181°S, 35.003°E). Relatively small protected areas of habitat are present in the bay. Large gill nets were, however, witnessed being set daily across the opening of the bay during the sampling period.

The shoreline population of the SEA is 404,850 and has a reported annual growth rate of 3.16 %. A contributing factor to the district's relatively high population density and growth rate is the migration of people into the district with intentions to participate in the fishing and agricultural sectors (Donda et al. 2014). The SEA is subjected to similar rainfall patterns to that of the Lake Niassa Reserve (LNR), and the study sites share the same dominant shoreline categories. A differing physical feature between the LNR and SEA is the level of productivity. The SEA has a relatively high productivity level and vast expanses of lake bottom dominated by diatom ooze (Weyl et al. 2010). The three sites (Table 3.2) were selected, similarly to the LNR as to encompass multiple bottom types (i.e. sand, reef and grass) at varying depths as well as including areas of different fishing pressures.

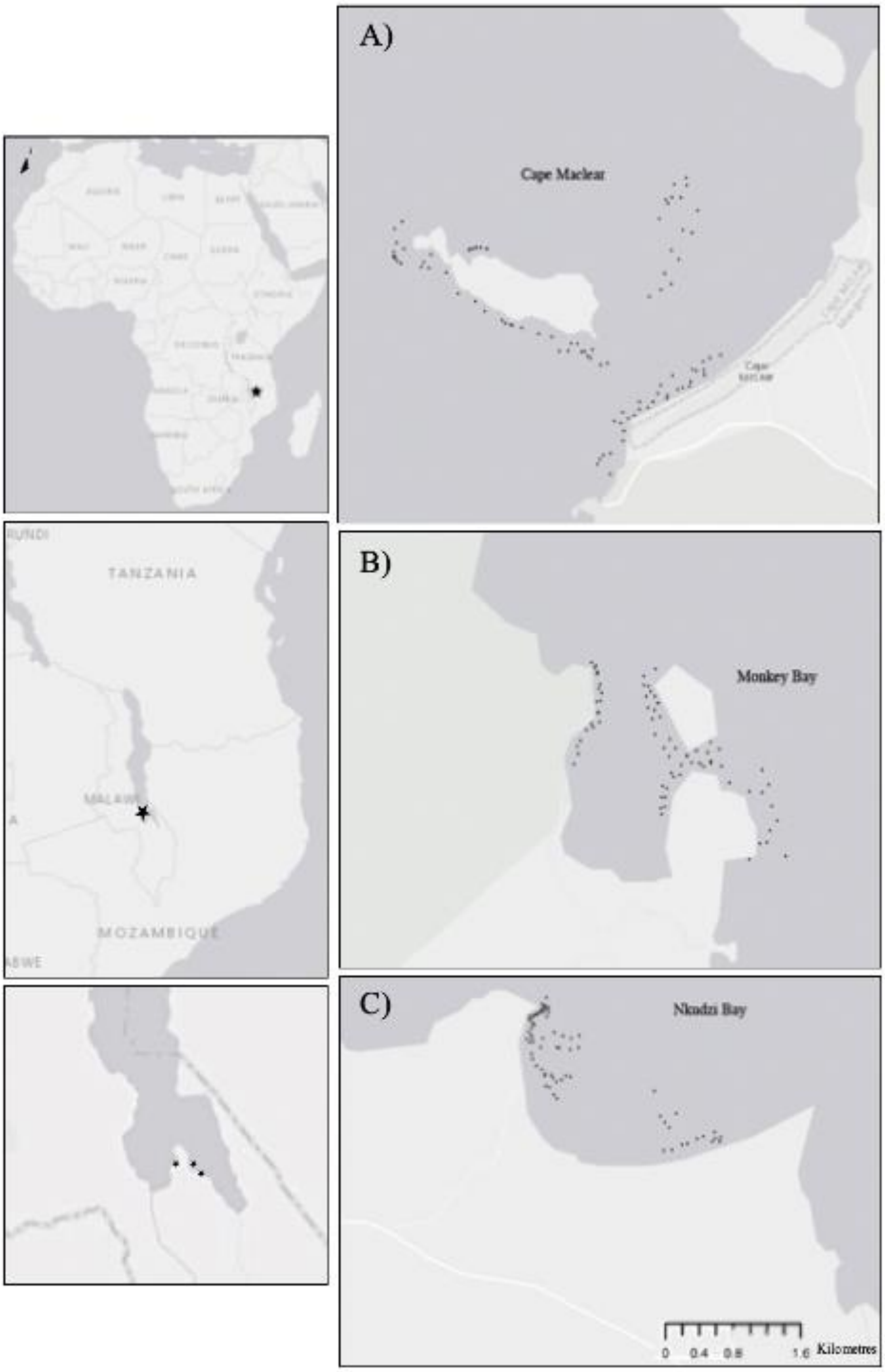


Figure 3.2: Map displaying baited remote underwater video systems (BRUVS) deployments in the South Eastern Arm, Malawi in the year 2018. A) the only study location within the Malawi National Park, Cape Maclear B) Monkey Bay and C) the southernmost study location, Nkudzi Bay. Symbol (●) represents individual BRUVS deployments and the symbol (★) represents recognised fishing centres.

Table 3.2: Summary of selected study sites in the South Eastern Arm, Malawi. Displaying location, dominant bottom types, mean depth sampled, lake-wide fishing pressure relative to all study locations and total baited remote underwater video systems (BRUVS) deployments (Alimoso et al. 1990, Mdaihli et al. 1992, Weyl et al. 2010, M''balaka et al. 2018).

Site	Coordinates	Dominant bottom type(s)	Depth sampled (m) (Min, Mean, Max)	Lake-wide relative fishing Pressure	Total BRUVS deployments (2018)
Cape Maclear	14.014°S, 34.849°E	Reef, Sand	2.5, 17.9, 42.5	Moderate	104
Monkey Bay	14.071°S, 34.917°E	Reef, Sand	1.7, 14.4, 39.3	Moderate-high	73
Nkudzi Bay	14.181°S, 35.003°E	Sand	1.1, 9.2, 35.7	High	55

Sampling strategy

Mono-BRUVS description

Ten mono-BRUVS were employed for sampling over the years 2016-2017. The systems had a standardised construction and were initially designed to be cost-effective, lightweight and able to operate at depths shallower than 70 m (Figure 3.3). A single GoPro Hero 3 action camera, inside a stock waterproof housing, was mounted upon a triangular plastic base plate. The GoPro Hero 3 was set to record with a medium field of view, at 1080p with autofocus disabled and recording at 60 frames per second. The base plate is the central component upon which a pair of steel legs, a mooring line and a single 1 m steel bait arm attaches. The mono-BRUVS were deployed and retrieved by hand from a single research vessel via a mooring line attached to both the system and surface buoy. A single perforated PVC container was positioned on the bait arm directly within the camera's field of view and acted as a bait container.

Before each deployment, a standardised 100 g of crushed Usipa (*Engraulicypris sardella*) was placed into the bait container. Usipa is an affordable and of-least-concern oily-pelagic species native to Lake Malawi (IUCN 2019). A pilot study in the Lake Niassa Reserve was conducted in 2015 and determined that 100g of Usipa was an adequate amount of bait to monitor fish species on both shallow sand and reef habitats and was found to be more effective at observing species than non-baited systems (Weyl et al. 2016).

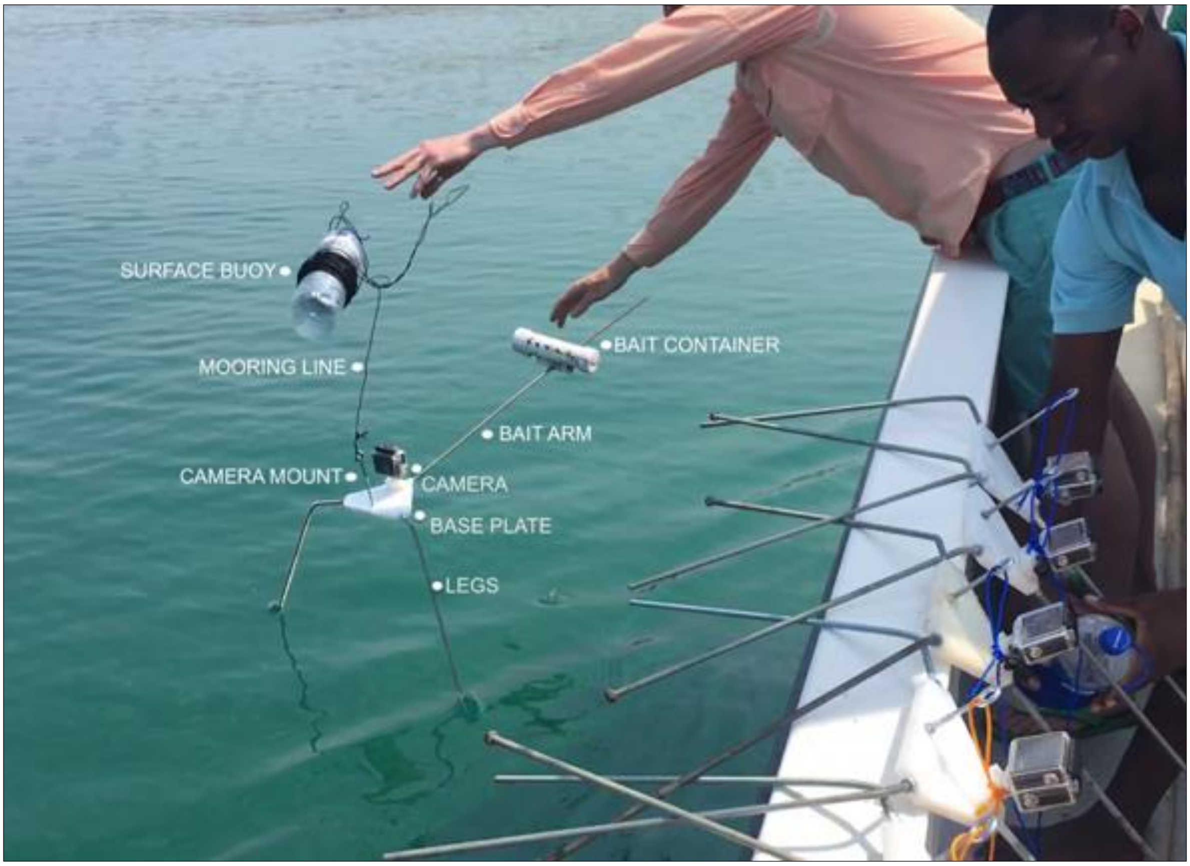


Figure 3.3: Annotated photograph of mono- baited remote underwater video systems (BRUVS).

Stereo-BRUVS description

Sampling in the year 2018 employed stereo-Baited Remote Underwater Video Systems (stereo-BRUVS) only. The stereo-BRUVS were constructed of a stainless-steel frame, two GoPro Hero 5 action cameras (enclosed within aluminium housings), a single PVC bait attached to a 1 m bait arm and an Onset HOBO Water Temp Pro v2 logger (Figure 3.4). Each GoPro Hero 5 was set to record with a medium field of view, at 1080p with autofocus disabled and recording at 60 frames per second. Two systems were deployed and retrieved by hand from a single research vessel. The stereo-BRUVS were heavier in construction and remained attached to a surface marker buoy for retrieval via a length of reinforced mooring line (4 mm Dyneema gp12 floating rope).

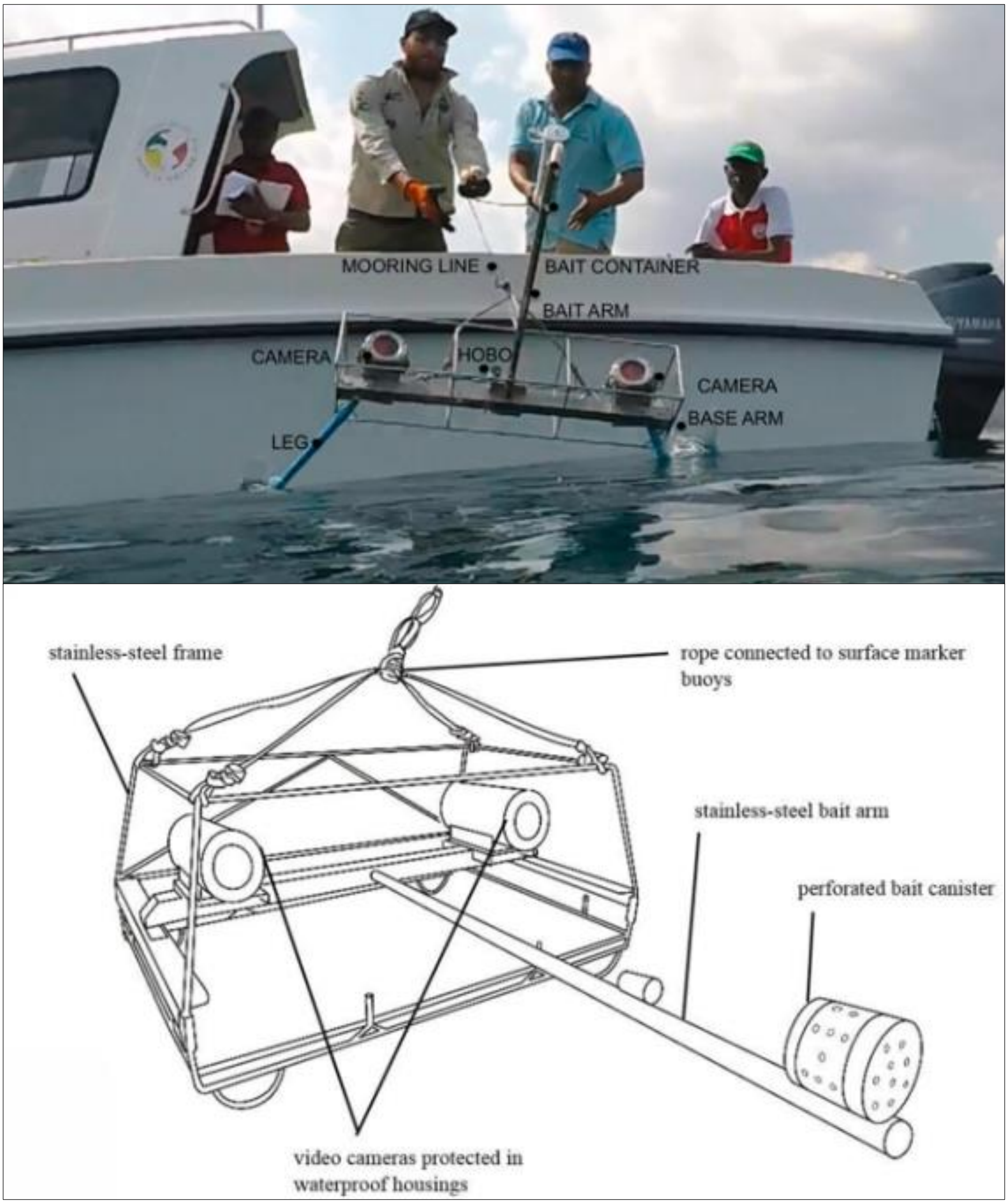


Figure 3.4: Annotated photograph of stereo-baited remote underwater video systems (BRUVS) deployment and a detailed diagram of general stereo-BRUVS design (Harvey and Shortis 1995, Heyns-Veale et al. 2016).

The two cameras (left and right), were mounted 0.7 m apart upon a steel base arm and the cameras on both systems face inward at a standardised angle of 8^{o}. The camera lenses and positioning of the housings allow for an overlapping stereo field of view and subsequently, the capability to collect length measurements during video analysis (Harvey and Shortis 1995). Calibration of the camera systems was conducted both before and following the completion of the field trips and conducted using CAL Software and proprietary equipment (SeaGIS Pty. Ltd.). The bait container was positioned directly within the cameras field of view, and all deployments were filled with 100g of crushed Usipa (*Engraulicypris sardella*).

Sampling approach

Selection of deployment sites was achieved using a stratified random sampling design. Selected study sites were initially explored using a motor vessel and an echo-sounder. Results from the exploration were then fit to a local map layer using ArcMap 9 and 10 (Environmental Systems Research Institute, Redlands, CA, U.S.A.). Both depth profiles and habitat type were noted on the map, and broad sampling zones within each study site were demarcated. Within each of these broader zones, random deployment points (a minimum of 50 m apart) were then selected

The stratified random sampling design was constructed to ensure an even allocation of sampling sites within the level of the habitat strata. The four habitat types classified were reef, sand-reef, sand and grass. The duration of each deployment was 15 minutes and was taken from the time that the system settled on the benthos of the lakebed. The deployment duration was achieved by noting the location of the system from the vessel’s echo-sounder. The sampling approach was standardised for all study sites (Malawi & Mozambique) and completed between months September to November.

Video analysis

All video data was analysed using EventMeasure Software (SeaGIS Pty. Ltd.). The software, as discussed in Chapter 2, is a paid-for software specifically designed to collect data from video footage – including stereo video. The software's period creator function was used to create a watching period of 15-minutes and named "*analysis*". The creation of the period was to ensure all the observer's collected data remained within a standardised time interval. In the instances of BRUVS being rotated by an underwater current or visually obstructed, the deployments were included in the analysis and observed for as long as feasible.

Using available literature, all selected monitoring species matching their respective descriptions and able to be positively observed within the standardised time frame were included in the analysis. Individual image frames of clearly presented individual fish were also collected during analysis. These images were stored and later used to develop a working species identification table.

Estimated species abundance was calculated using a MaxN approach. As discussed in Chapter 2, MaxN as an abundance estimate is relatively easy to attain from the footage and decreases the probability of the observer counting individuals more than once (Ellis and DeMartini 1995). The approach selects a single frame from a video sample which captured the highest number of individuals per species. MaxN is therefore considered to be a conservative index or a precautionary approach and is suited to conservation science as it is less likely to overestimate the true abundance (Ellis and DeMartini 1995). Explanatory environmental variables considered to influence species identification and MaxN abundance counts were recorded both before deployment as well during video analysis.

Monitoring species

Lake Malawi supports an extraordinary number of freshwater fish species, many of which are exploited by fisheries and are a nutrient source for millions of people. Other species, although not directly targeted by a fisheries sector, play an essential role in sustaining the local biodiversity, and many are highly sought after in the global aquarium trade (Weyl et al. 2010). The rapid speciation event that took place in the Cichlidae family over 700,000 years ago is mainly responsible for the captivating biodiversity synonymous with Lake Malawi. The scientifically intriguing event has since led to highly evolved trophic specializations, the dominance of almost all available habitat types and subsequently, both the potential for cryptic phenotypes as well as behaviours (Ribbink et al. 1983, Danley and Kocher 2001). Consequently, identifying total species richness from BRUVS video footage poses a significant challenge.

Ordinarily, BRUVS employed in easily-described systems have the potential to identify all fish observed down to species level (Harvey and Mladenov 2001). Lake Malawi however, is not such a system. Considerable effort has been put into identifying the lake’s fish species and over 1,000 species are believed to inhabit the system (Lyons et al. 2015). The actual number of species may, however, be much higher as researchers propose that the lake’s species diversity maybe double that (Turner et al. 2001). To address the uncertainty of calculating true abundance or species richness using BRUVS, monitoring species groups were selected based upon two broad requirements. One, the species” selected must be able to be positively identified using only recorded video and two, the complete collection of monitoring species groups must be representative of the Lake Malawi shoreline ecosystems (Table 3.3).

National catch data and fisheries monitoring in Lake Malawi classifies multiple species under broader biological classifications – treating species flocks and species of similar guilds as single management units (Halafo et al. 2004, Banda et al. 2005a).

Table 3.3: Summary of selected monitoring species groups in both the South East Arm, Malawi and Lake Niassa Reserve, Mozambique for years 2016-2018. The table displays the genus and species included in the species group, their respective feeding guilds, their International Union for Conservation of Nature (IUCN) status, years monitored and their key characteristics.

Study species grouping	Genus	Species	Feeding	IUCN status	Year monitored	Key characteristics
Chambo	Oreochromis	*O. Karongae, O. lidole, O. squamipinnis*	General algae and planktivore	Critically Endangered	2016, 2017, 2018	Endemic, high-value species, nest builder, large schools
Kampango	Bagrus	meridionalis	Predator	Critically Endangered	2016, 2017, 2018	Endemic, high-value spp., nest builder
Labeo	Labeo	cylindricus	General rock algae and diatoms	Least Concern	2016, 2017, 2018	Potamodromous, plicate lips, shoaling
Metriaclima estherae	Metriaclima	estherae	Omnivore	Least Concern	2018	Endemic, rock cichlid, female colouration, aquarium spp.
Mlamba	Bathyclarias and Clarias	Genus level	Predator	Least Concern	2016, 2017, 2018	Important fisheries spp., local aquaculture spp.,
Predatory haplochromines	Dimidiochromis, Tyrannochromis, Rhamphochromis	Genus level	Predator	Least Concern	2018	Endemic, piscivore, important fisheries spp., aquarium spp.
Sleepers	Nimbochromis	livingstonii, linni, polystigma	Predator	Least Concern	2018	Endemic, “sleeper” predation, aquarium spp., shoaling
Yellowfish	Labeobarbus	johnstonii	Omnivore	Least Concern	2016, 2017, 2018	Endemic, Potamodromous, schooling

Examples being the classification of the three *Oreochromis* (*Nyasalapia*) species as Chambo, the Bathyclarias and Clarias genera as Mlamba and multiple species of haplochromine cichlids as Predatory haplochromines. To draw a comparison, the fisheries-targeted species in this study were grouped to align with the broader management units. These groups included Chambo, Mlamba and Predatory haplochromines (Table 3.3). The single Mbuna species or rock cichlid, *Metriaclima estherae* was included as a key monitoring species. Although there are numerous species within the broader Mbuna classification, *Metriaclima estherae* was selected due to the species' identifiable colouration and limited distribution within the Lake Niassa Reserve (Sayer et al. 2019). The remaining species groups included species with differing ecological traits - including that of feeding behaviours, habitat utilization, migratory and schooling behaviours. Apart from the selection of monitoring species, a species identification table from the three-year BRUVS dataset was tabulated for the LNR, and a species identification database was developed with the intention to create on online identification portal.

Data analysis

Explanatory variables & descriptive statistics

Microsoft Excel 2016 was used for the compilation of all metadata and R commander and R Studio used for the running of all descriptive statistical summaries and tests (Fox and Bouchet-Valat 2019, RStudioTeam 2019). These statistical summaries involved calculating the means, counts and standard deviations of different factors and covariates as well as the running of statistical tests to determine the significance or lack thereof between selected means - including the use of t-tests during data exploration and determining possible differences between recorded variable means such as *Depth* and *Temperature* by study sites.

Optimal deployment time

Optimal deployment time, also commonly referred to as saturation time for all deployments, was calculated using a rate of accumulation for new species detected by the BRUVS using R Studio (RStudioTeam 2019). A two-parameter logistic-ogive or sigmoid curve function is typically used to model fish maturity, but in the case of this study allowed for the determination of 50 % and 95 % species accumulation using MaxN relative abundance counts (Weyl and

Booth 1999). The analysis included only those species groups which were included in all three years of monitoring and both Malawi and Mozambique. This included Chambo, Kampango, Mlamba, Labeo and Yellowfish. Ninety-second time intervals, over a period of 15 minutes were used to determine ten intervals (including 0) from which the cumulative proportion of the observed individuals at MaxN was calculated for each deployment sample.

Analysis followed the method discussed by Bernard et al. (2014) and Weyl & Booth (1999). Non-linear mixed-effects (NLME) models were used as they accommodate repeated measures data. In the case of this study, the individuals are assumed to be a random sample from the recorded population (Lindstrom and Bates 1990). The NLME model was fitted with a two-parameter logistic-ogive function proposed by Weyl and Booth (1999):

$P_t = 1 / (1 + e^{(t-t50) / \delta})$ **(3.1)**

and

$P_t = 1 / (1 + e^{-\ln 19(t-t50) / (t95-t50)})$ **(3.2)**

P_t being the expected accumulation of species recorded in the footage at time t and accounting for the total number of species recorded through the duration of the footage. t50 was the time taken for 50 % species saturation to be achieved and t95 the time taken for 95 % saturation. The gradient of the slope was determined by δ, the inverse rate of saturation. Parameters t50, t95 and δ were predicted by minimizing their respective sum-of-squares (Bernard et al. 2014).

Sample size requirements

Power analysis was used to calculate an adequate sampling effort required to detect an annual 10 % change in abundance of population size over ten years and followed the approach of De Vos et al. (2014). The authors originally used a 5-year monitoring period, however, to align with the monitoring time-frame proposed by the “Save the Chambo” campaign in Malawi, this study made use of a 10-year period. The analysis utilized a determined 80 % power, a confidence interval of 95 % and a significance level of $\alpha < 0.05$. Using an 80 % power is in reference to the acceptable probability that the model will detect statistically significant differences when the actual differences do exist. Thus, avoiding Type II errors and having an 80 % chance of correctly rejecting a null hypothesis (Bausell and Li 2002)

The abundance measure obtained from MaxN values were assumed to follow a Poisson distribution upon which a Poisson generalized linear model (GLM) was fitted with an average MaxN, μi. Depth was incorporated as a categorical factor following a step-wise Akaike Information Criterion model selection process. βd represents the estimated coefficient for depth stratum and α the intercept:

$$\mathrm{Ln}(\mu_i) = \alpha + \beta_d \times \mathrm{depth}_i \quad \textbf{(3.3)}$$

Model validation for over-dispersion was then performed according to the method proposed by Zuur et al. (2009). If the residuals from the GLM were found to be overdispersed, the models were then fitted to a negative binomial GLM. Negative binomial GLMs avoid the stringent assumptions of Poisson distributions, which require that the variance is equal to the mean. Hence, negative binomial GLMs are often referred to as overdispersion models (Bolker et al. 2009). All models in this study were overdispersed, and subsequently, all analyses were carried out with the negative binomial error distribution.

The power analysis followed a Monte-Carlo simulation approach adopted by De Vos et al. (2014). The Monte-Carlo method generates large numbers of simulated samples of data based on assumed population parameters. These parameters characterize the population from which the simulated samples are drawn (Muthén and Muthén 2002). For each run scenario, 1000 simulations were completed and were run to determine an associated power with an n sample size (n being the number samples per year). For each simulation, n random MaxN counts for all ten years were generated from the negative binomial distribution.

The deterministic trend in average MaxN was then based upon the following equation:

$MaxN_{z,y+1} = MaxN_{y,t} \times e^{r}$ **(3.4)**

where $MaxN_{y,t}$ represents the count value predicted for depth stratum, year and rate of population increase. z represents depth stratum, y the count value predicted for the year and r the rate of population increases. The power analysis relied upon a significance level of $\alpha < 0.05$ and subsequently, considered a Type I error rate of 5 %. Simulating a randomly stratified design required a relatively equal number of samples from predetermined depth ranges, localities and habitat types. An equal number of shallow, moderate and deep samples were randomly drawn for each year and a GLM was fitted to the data generated by each simulation (De Vos et al. 2014):

$Ln(\mu_i) = \alpha + \beta_d \times depth_i + \beta_r \times year_i$ **(3.5)**

β_d represented the estimated coefficient for depth stratum, α the intercept and β_r the estimated year trend. Where β_r was significant ($p < 0.05$), the number of simulations were recorded. The sample size of n began at four and was increased stepwise by either 4 or 8 steps depending upon the model requirement. The statistical power for the sample size of n was then calculated as a proportion of simulations from which a significant trend could be detected. An asymptotic growth function was then fit to the simulated power curve results:

$power = 1 - e^{-bn}$ **(3.6)**

where b was the rate of increase as power tends to the value one (De Vos et al. 2014). The point estimates for sample sizes required to achieve an 80 % power were calculated as:

$n_{p80} = b^{-1} \ln (1 - 0.8)$ **(3.7)**

RESULTS

Explanatory variables

The three-year sampling period of this study resulted in the collection of 1351 deployment of which, 1270 were successful and resulted in a sampling success rate of more than 94 %. Table 3.4 displays the complete recording of continuous and factorial covariates for both *Malawi* and *Mozambique.* Neither the mean *Depth* or *Temperature* differed significantly between the two countries and the complete operating depth range of the BRUVS was 0.8 m to 64.9 m.

Table 3.4: Continuous and factorial covariates recorded for Malawi (2018) and Mozambique (2016-2018). Including the number of baited remote underwater video systems (BRUVS) deployments per analysis level.

Continuous covariates									
Mozambique					**Malawi**				
Name	mean	SD	min	max	Name	mean	SD	min	max
Depth (m)	12.58	0.36	0.80	64.90	*Depth* (m)	13.13	0.74	1.10	42.50
Temperature (°C)	25.32	0.11	21.10	28.10	*Temperature* (°C)	25.86	0.07	21.70	28.84

Factorial covariates

Name	levels	n deployments
Country		
	Malawi	232
	Mozambique	1038
Year		
	2016	342
	2017	533
	2018	395
Location		
	Cape Maclear	104
	Chuanga	30
	Cobue	178
	Meluluco	331
	Metangula	347
	Monkey Bay	73
	Nkudzi Bay	55
	Nkwichi	152
Habitat		
	Grass	115
	Reef	398
	Sand	601
	Sand-reef	156

Optimal deployment time

The rate of species group accumulation only comprised those species groups which were included in all three years of monitoring and both *Malawi* and *Mozambique*. These species groups included *Chambo, Kampango, Mlamba, Labeo and Yellowfish.* The optimal deployment time whereby 95 % of selected monitoring species groups at MaxN had been accumulated in *Malawi* was 5.7 minutes (SE = 0.7) and 8 minutes (SE = 0.9) in *Mozambique* (Table 3.5). The comparison of the rate of species accumulation with increasing deployment time showed that there was no significant difference between *Mozambique* and *Malawi* at the 50 % (F = 3.52, p = 0.061) and the 95 % saturation levels (F = 5.87, p = 0.161) (Table 3.5). Similarly, the recorded number of species groups at the 95 % saturation level did not differ significantly between the countries (Figure 3.5).

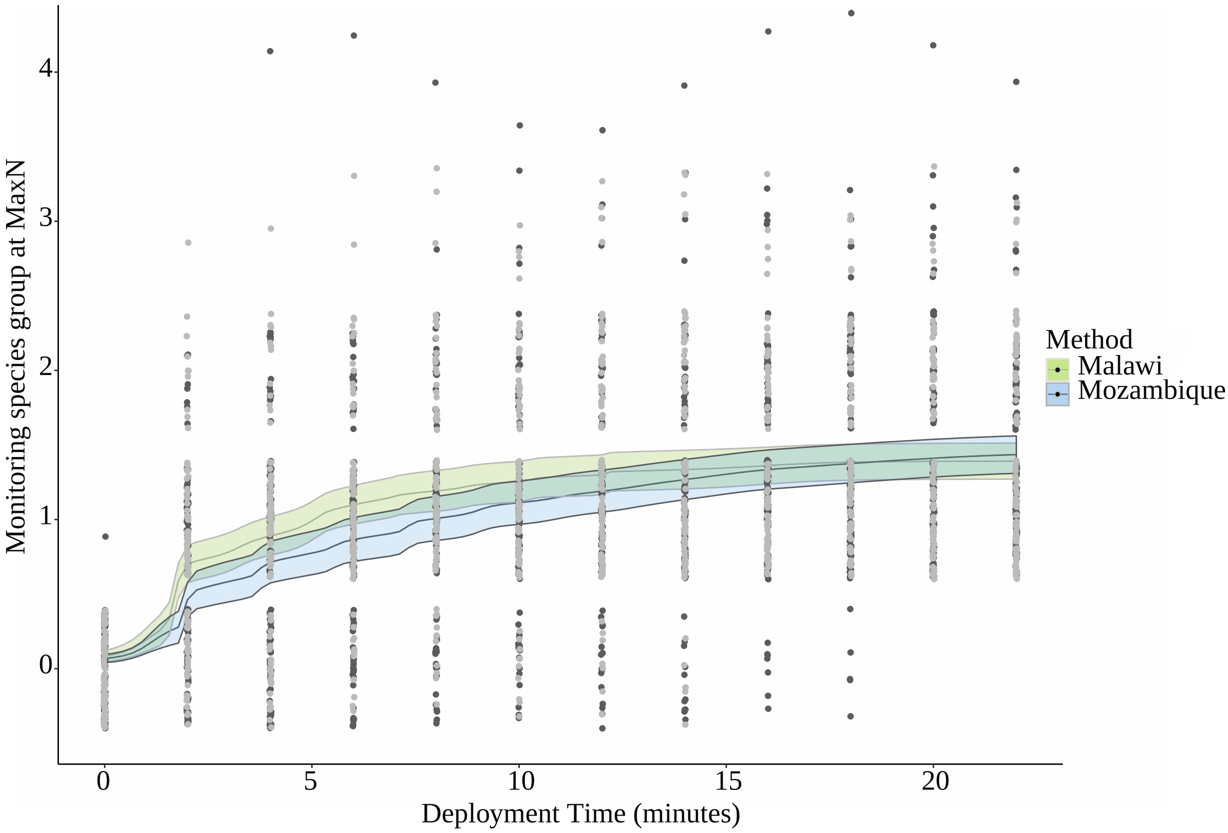


Figure 3.5: Predicted results from the non-linear mixed-effects models, showing the accumulation of species groups for Malawi and Mozambique plotted against the deployment time of BRUVS. The shaded areas indicate the 95 % confidence interval

The average time at MaxN was less than 10 minutes for all monitoring species groups (Table 3.6). The study species averaging the shortest recorded MaxN times were *Labeo* and *Metriaclima estherae* with mean 4.36 minutes (SD = 0.56) and 4.71 minutes (SD = 0.69), respectively. The two species groups with the longest required MaxN recording times were *Mlamba* and *Sleepers* with average times of 9.73 (SD = 0.60) and 9.05 (SD = 0.55) minutes.

Table 3.5: Results from the non-linear mixed-effects analysis on the baited-remote underwater video systems (BRUVS) data comparing the average time (± SE) at which species group accumulation were at 50 % and 95 % saturation levels for Mozambique and Malawi. The significance levels for the comparison of the optimal deployment time between Mozambique and Malawi have been provided.

	Mozambique		**Malawi**			
		Time			**Comparison of time**	
	Mean	SE	Mean	SE	F-test	P value
50 %	6.15	0.71	4.10	0.53	3.52	0.061
95 %	8.00	0.95	5.70	0.69	5.87	0.161

Table 3.6: Average MaxN recording times (minutes) for selected monitoring species for full video lengths. Displaying Standard deviation (SD) and the total number of MaxN counts for each species group.

Species group	MaxN/time (mins)	SD	Total species MaxN counts
Chambo	5.92	0.53	3450
Kampango	6.49	0.86	71
Labeo	4.36	0.56	231
Metriaclima estherae	4.71	0.69	232
Mlamba	9.73	0.60	241
Predatory haplochromines	8.85	0.72	47
Sleepers	9.05	0.55	57
Yellowfish	7.90	1.96	417

Sample size requirements

The amount of sampling effort required was negatively associated with the detection probability and abundance of all species groups. Similarly, estimates for required sampling effort increased with decreasing dispersion values, (σ) (Table 3.7). A more considerable sampling effort was required in *Malawi* to reach a power of 80 % in all but one species group, *Labeo*. For *Labeo, Malawi* required 42 samples and *Mozambique,* 56 (Figure 3.8). *Sleepers* differed by less than 50 % between *Malawi* and *Mozambique* and required 32 and 24 samples, respectively (Figure 3.10).

The sample size requirement for *Kampango, Chambo, Mlamba* and *Predatory haplochromines* present major (> 50 %) differences between *Malawi* and *Mozambique*. Three times more sampling effort and a total of 168 samples are required to reach 80 % power for *Kampango* in *Malawi*. Compared to the 48 samples required in *Mozambique* (Figure 3.6). *Chambo* requires double the sampling effort in *Malawi* compared to *Mozambique* with individual sample size requirements of 120 and 56 (Figure 3.7). The sample size requirement curve for *Predatory haplochromines* required 24 deployments in *Mozambique* and 50 deployments in *Malawi* to reach a level of 80 %. The final fisheries targeted species, *Mlamba* required 56 samples in *Malawi* and 36 in *Mozambique* (Figure 3.9).

The 2018 sampling effort in *Malawi* only detected *Yellowfish* twice in the 233 deployments, resulting in the general linear model (GLM) failing to converge and thus, the analysis was dropped from the study. The *Yellowfish* species group from *Mozambique* required the most considerable sampling effort of 320 deployments (Figure 3.12). The final graph (Figure 3.13) investigated the required sample size for *Metriaclima estherae* in its described distribution, Metangula and Meluluco. Although having a limited distribution within the Lake Niassa Reserve (LNR), the monitoring species required a relatively modest sampling effort of 24 annual samples.

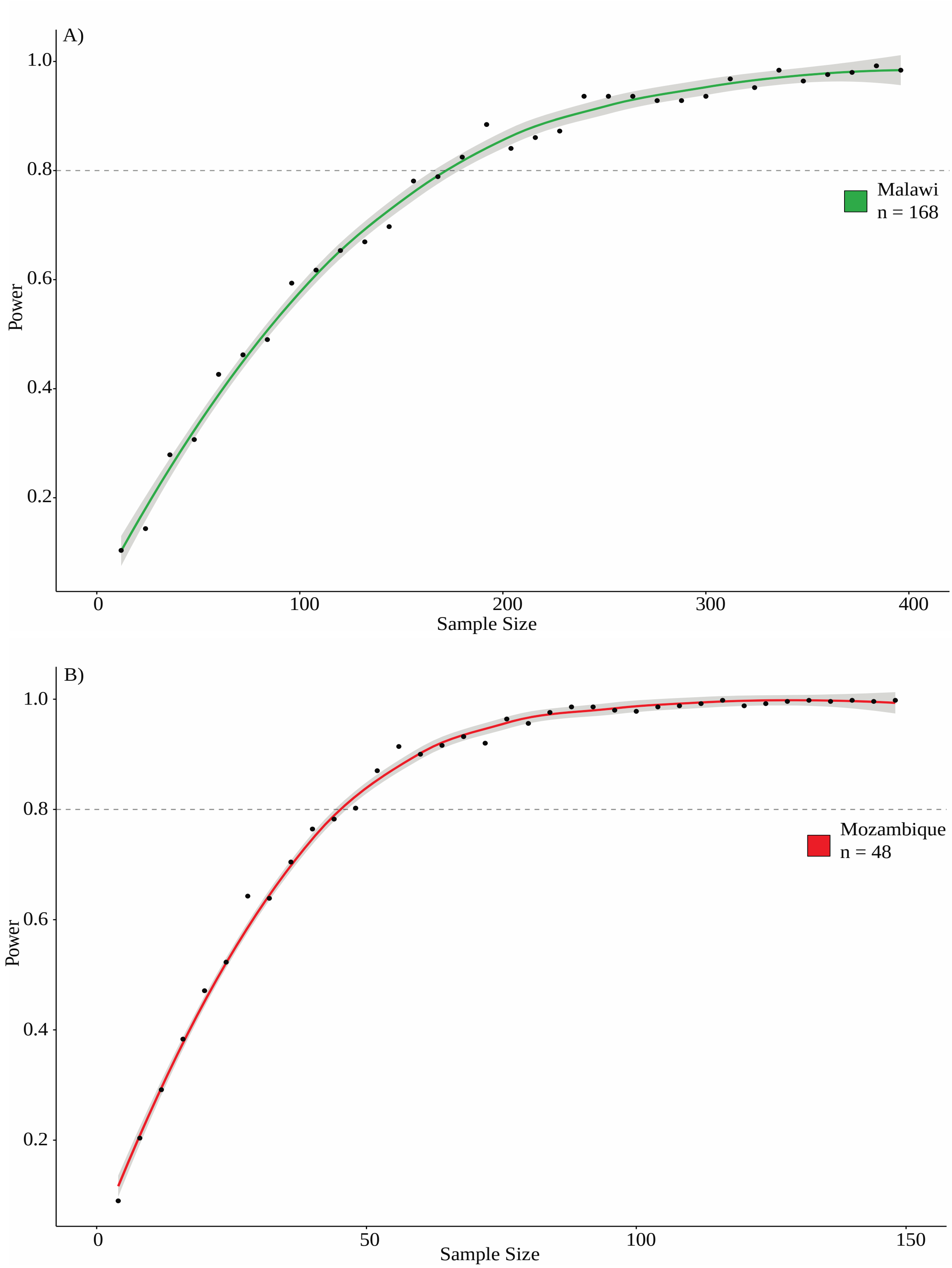


Figure 3.6: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Kampango*. Malawi (A) requiring 168 samples and Mozambique (B) requiring 48.

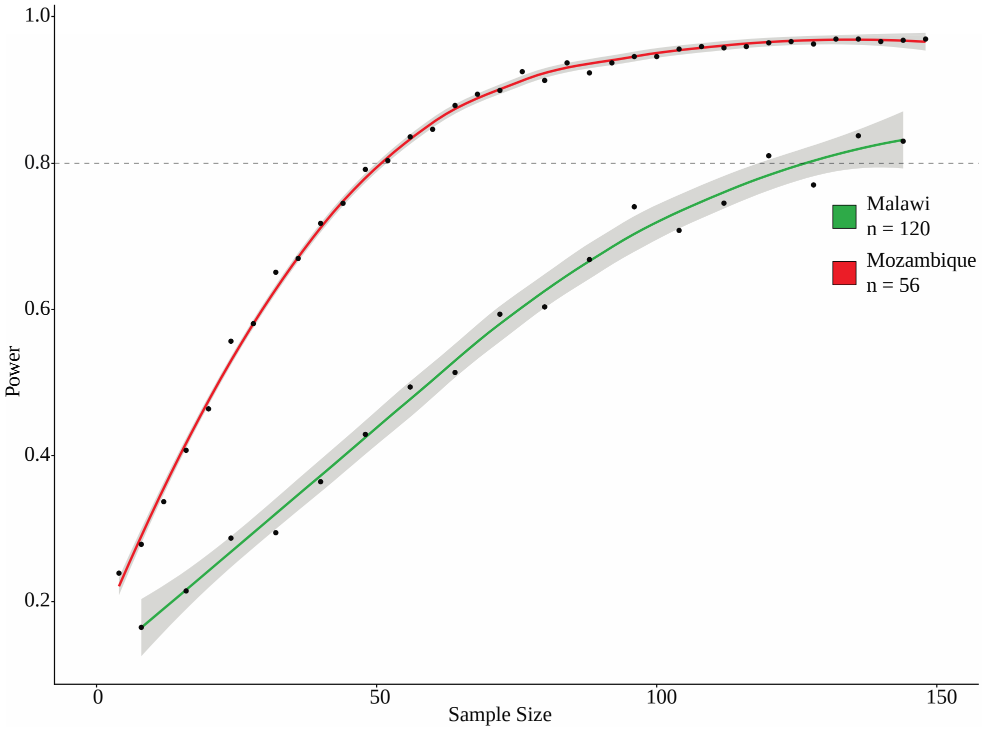


Figure 3.7: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Chambo*. *Malawi* requires 120 samples, and *Mozambique* requires 56 to reach a power of 80 %.

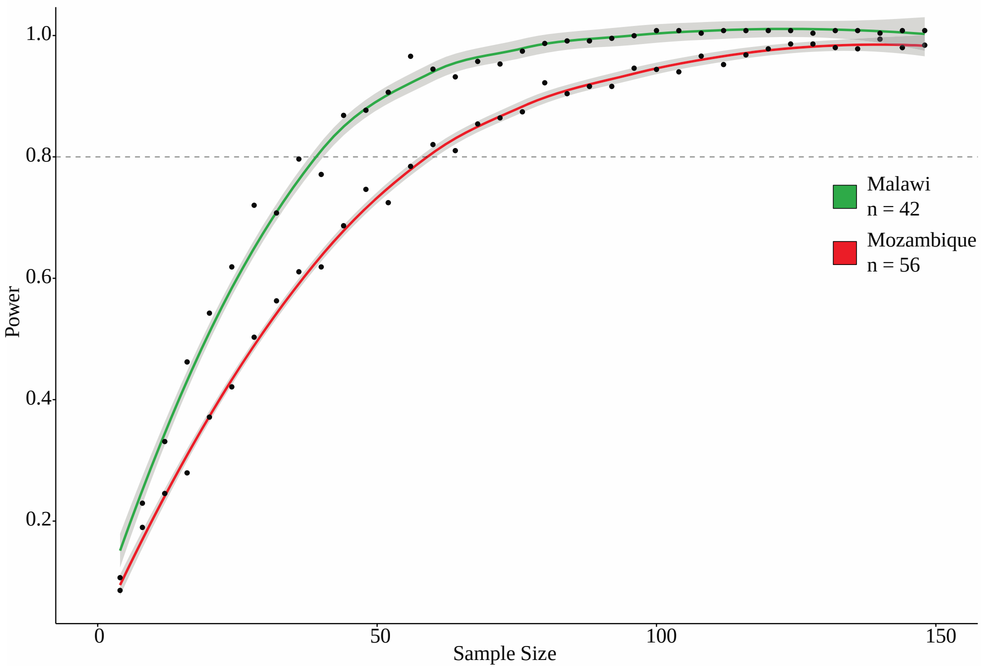


Figure 3.8: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Labeo*. *Malawi* requires 42 samples, and *Mozambique* requires 56 to reach a power of 80 %.

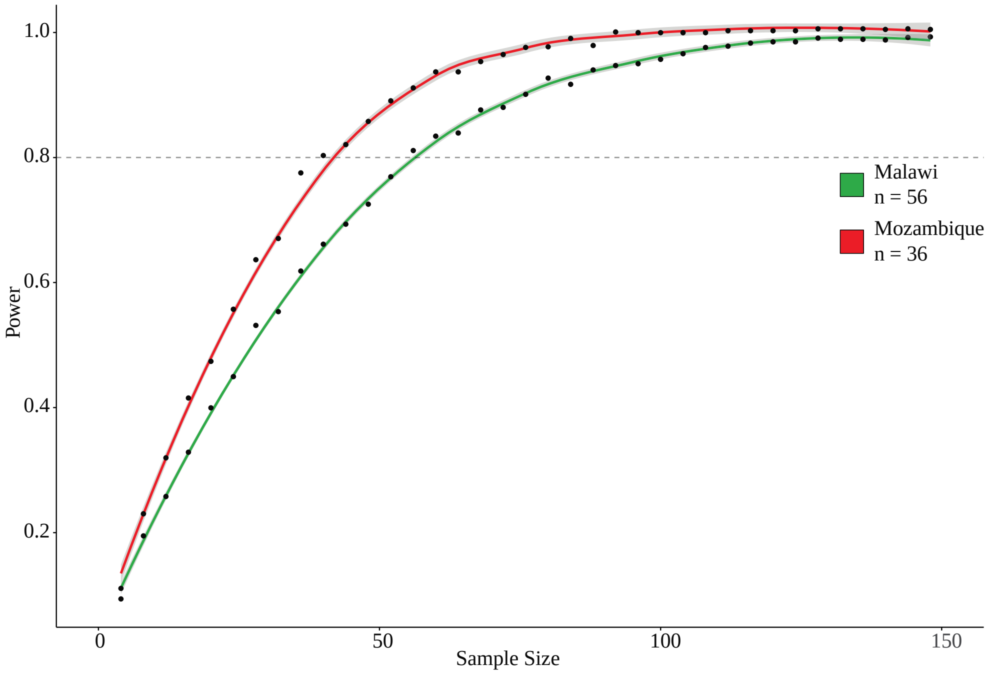


Figure 3.9: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Mlamba*. *Malawi* requires 56 samples, and *Mozambique* requires 36 to reach a power of 80 %.

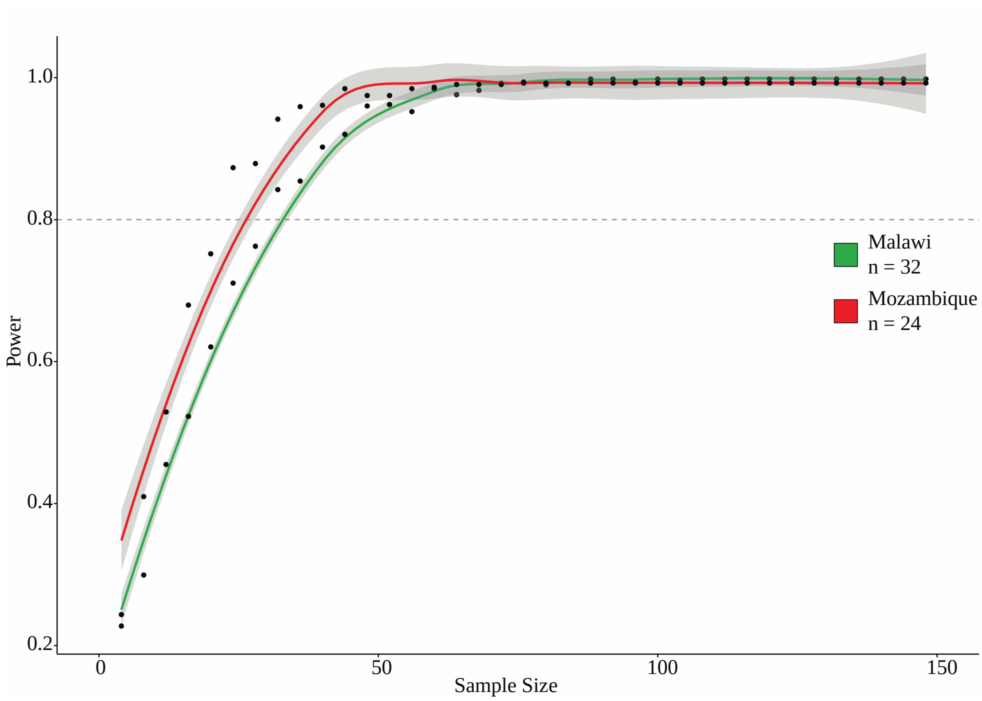


Figure 3.10: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Sleepers*. *Malawi* requires 32 samples, and *Mozambique* requires 24 to reach a power of 80 %.

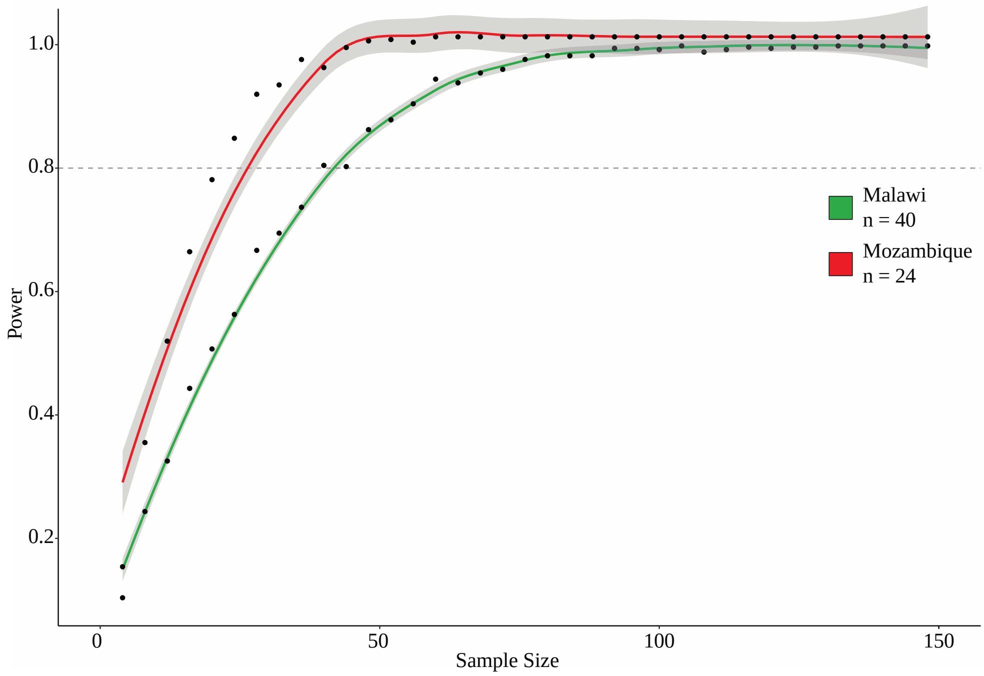


Figure 3.11: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Predatory haplochromines*. *Malawi* requires 40 samples, and *Mozambique* requires 24 to reach a power of 80 %.

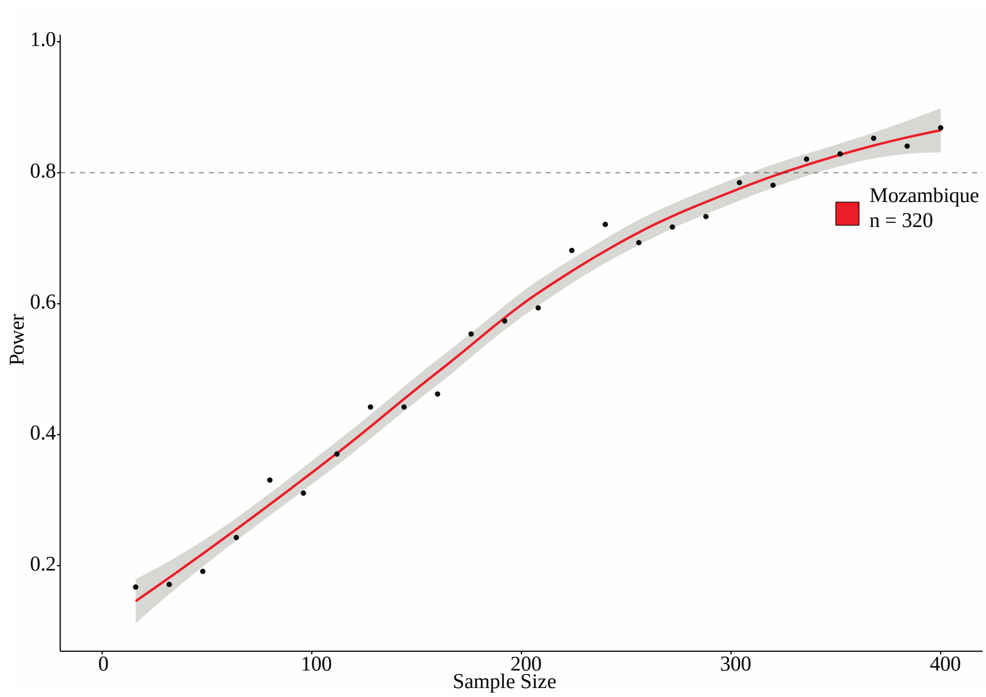


Figure 3.12: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Yellowfish*. *Mozambique* requires 56 to reach a power of 80 %.

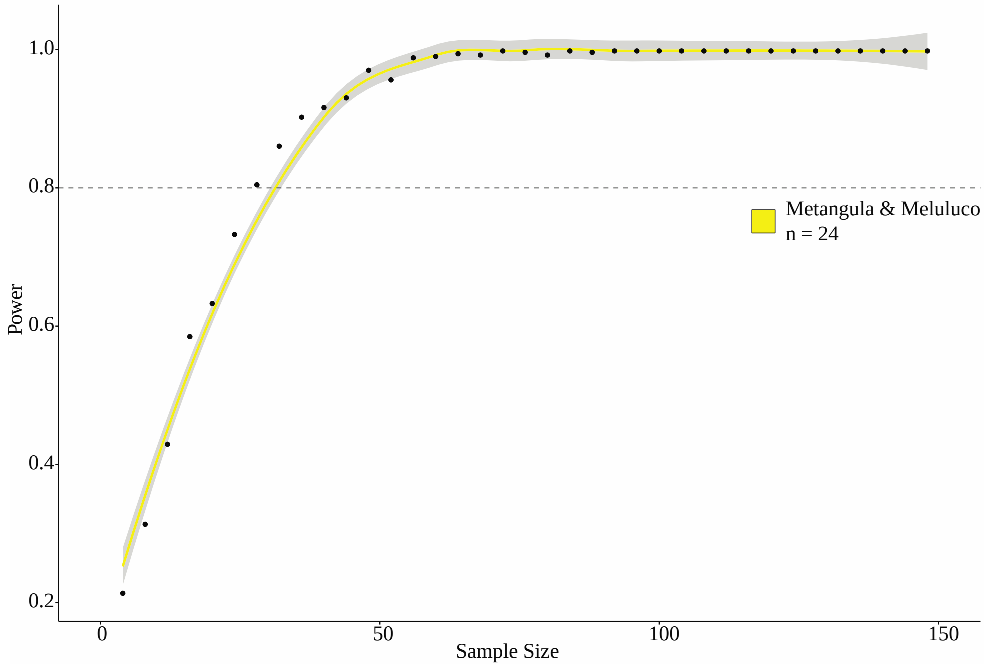


Figure 3.13: The effect of sample size on power to detect a 10 % change per year over ten years for the species group, *Metriaclima estherae*. *Metangula* and *Meluluco* requiring 24 to reach a power of 80 %.

Table 3.7: Dispersion or theta (σ) values of negative binomial models for selected species groups for Malawi, Mozambique as well as Metangula and Meluluco.

Species Group	Malawi	Mozambique	Metangula & Meluluco
Chambo	0.0775	0.1918	
Kampango	0.3578	1.7750	
Labeo	0.4137	0.3757	
Metriaclima estherae			0.4008
Mlamba	0.5664	1.1360	
Predatory haplochromines	0.2236	0.2525	
Sleepers	0.4056	2.375	
Yellowfish		0.0341	

DISCUSSION

In Chapter 2, I discussed the current applications of Baited Remote Underwater Video Systems (BRUVS) in the marine realm and proposed that the systems may aid in the monitoring of freshwater fish communities. There is, however, an evident paucity in information on how exactly this may be achieved. Freshwater-specific BRUVS design and standard operating procedures (SOPs) are critical to the advancement of the technology as a monitoring tool. There has, however, been no such research published to date. This chapter aimed to address this knowledge gap and take the first steps towards exploring the applicability of BRUVS in an African water body. Calculating both optimal deployment times and required sampling effort to detect minor changes in abundance over a long-term monitoring program.

Lake Malawi, supporting over 1.6 million people and labelled as the most species-rich lake in the world, is of both great scientific and economic value to its surrounding countries (Banda et al. 2005a). Various shoreline habitat types across a wide depth profile, adequate water visibility, and a need to monitor sensitive shallow water habitat presented the lake as a model study location. Being able to conduct non-destructive and standardised research in the lake will support current and future management of its fisheries resources. In this chapter, BRUVS were determined to be both effective and feasible for monitoring selected species groups in Lake Malawi. The BRUVS required a relatively short optimal deployment time of 15-minutes and a maximum annual sampling effort of 320 deployments to detect changes in abundance for the most undetected monitor species group.

A preliminary consideration for employing BRUVS as a long-term monitoring tool is their sampling success rate and potential of the data collected. The success of BRUVS is understandably affected by both the operator as well as the environment being sampled. Those environments which are characterised by excessive currents, dense vegetation, high water turbidity or steep bottom topography will increase the chance of the BRUVS failing to collect usable video footage. Such failures ordinarily occur due to the camera being completely obstructed or visibility being too poor to detect species adequately. The three-year sampling period of this study resulted in the collection of 1351 deployment of which, 1271 were successful (able to be analysed) and resulted in a sampling success rate of more than 94 %. This rate is a demonstrably adequate success ratio given that the BRUVS were deployed at a maximum depth of 64.9 m and 513 deployments were on structured habitats. All while being deployed entirely by hand and requiring - at a minimum, a two-person crew and an inflatable vessel powered by a single 5 HP motor.

Both configurations of the BRUVS (mono and stereo), were employed to collect abundance data for selected monitoring species in the Lake Niassa Reserve (LNR) and South East Arm of Lake Malawi. Calculating the abundance was achieved using a MaxN measure of relative abundance, as discussed by Ellis and DeMartini (1995). At the very least, the MaxN value recorded for each deployment may act as a grouping of observations for a species. This value may then be linked to a geographical location using the logged GPS waypoint. There were a combined total of 1129 confirmed MaxN recordings for the selected species groups over the three-year study period. This presence-absence or distribution data alone may prove to be useful in a system such as Lake Malawi where species-specific data are scarce.

Generating the optimal deployment time was based upon a declining rate of newly entering monitored species into the BRUVS field of view. The recorded 15-minute deployment time was based upon a 2015 pilot study in the LNR where Weyl et al. (2016) determined that BRUVS were not only more effective at observing significantly more species compared to non-baited systems but also that 15 minutes was an adequate recording time in both shallow water reef and sand habitats. The optimal deployment time for selected species was reviewed in this chapter given the inclusion of 3 years of MaxN data collected at greater depths, across different study sites and within a different country. Although a now much larger and encompassing dataset, the time taken to accumulate 95 % of MaxN abundance counts for selected monitoring species remained within the originally proposed 15-minute recording period. Times of 6.05 minutes (SD = 0.74) and 8.96 minutes (SD = 1.02) were identified for Malawi and Mozambique, respectively. This recording time - relative to marine estimates, is considerably shorter (Whitmarsh et al. 2017). The 15-minute deployment time may be more appropriate for this study given the monitoring of only a limited number of species groups as opposed to a complete species richness survey.

The final consideration for the applicability of BRUVS in Lake Malawi is the number of deployments required to detect changes in monitored species abundance. The "Save the Chambo Campaign" beginning in 2003 and spearheaded by the Malawi Fisheries Department laid out a 10-year plan to restore Chambo stocks to maximum sustainable yields recorded before the fisheries collapse (Banda et al. 2005a). Key objectives in the campaign's framework - which later proved to be challenging were points 3.1.2 and 3.1.3. These related to monitoring the success of Chambo restocking programs from the aquaculture industry and the formation of fisheries management and monitoring plans (Banda et al. 2005a). Monitoring the health of restocked habitats shall always present a challenge to fisheries management - particularly when the predominant fisheries monitoring technique involves netting of sensitive habitats and the somewhat counterproductive task of removing fish (Banda et al. 2005a). Although the initially proposed iteration of the restoration plan was completed in 2015, further 10-year monitoring programs of these restocking initiatives may benefit from the adoption of a less destructive technique such as BRUVS. Hence, a 10-year time frame to detect a minor change (10 %) in species abundance was selected not only for Chambo but all monitored species groups (Figures 3.6–3.13).

There are two key results from the power analysis on the BRUVS data. The first being the evident differences between the sampling effort required to detect the same changes in abundance between the countries, Mozambique and Malawi and second, the significantly higher sampling effort required for those species directly targeted by the countries' respective fishing sectors. There is a relationship between the probability of detection and recorded abundance of each species group and the sampling effort required. Those species groups which were detected less frequently and in varying abundances lead to greater variability in the model and subsequently, required a higher amount of sampling effort to achieve an 80 % power (Bausell and Li 2002). Those species groups directly targeted by the fisheries sector such as Kampango, Chambo, Mlamba and Predatory haplochromines all required a 50 % greater sampling effort in Malawi compared to Mozambique. Kampango in Malawi required the most considerable number of annual samples.

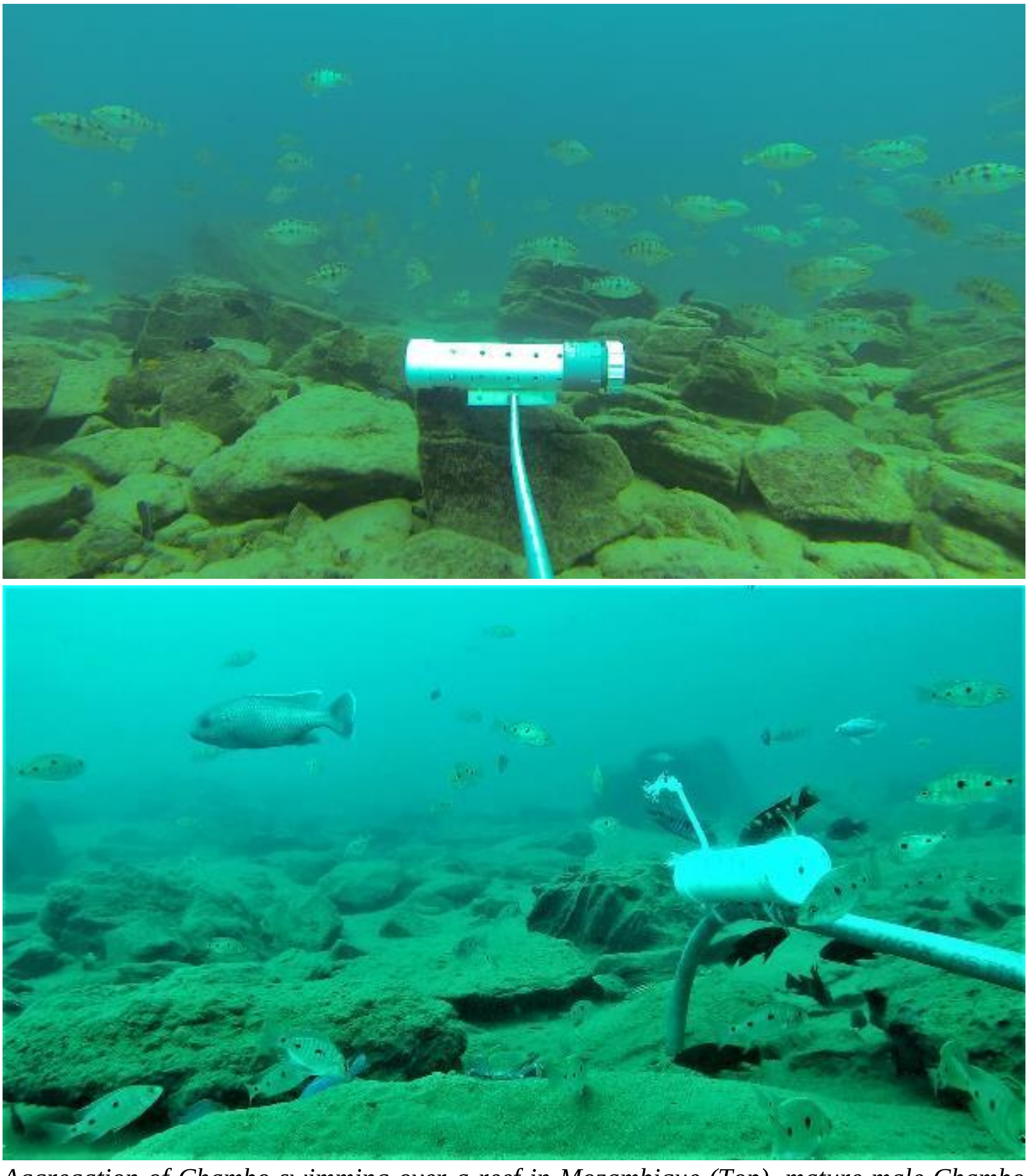
This difference in sampling effort can also be observed by the different dispersion values (σ) obtained (Table 3.7). Dispersion parameter estimates obtained from the negative binomial general linear models (GLMs) display the presence of over or underdispersion. This occurs when the observed variance is higher or lower than the variance of the model (Bolker et al. 2009). De Vos (2012) discusses that this variation may indicate how certain species' behaviours may affect their distribution within a study area. The less observed or uniformly distributed a species is within an area, the more the dispersion parameter will be skewed toward 0 and the higher the sampling effort requirement shall be (De Vos 2012). Conversely, overdispersion occurs when a species is highly abundant in some areas while absent in others. A dispersion parameter which conforms to the value 1, represents less variability in the model, a uniform distribution within space and ordinarily, lower sample size requirements. The most ubiquitously observed species group were the Mlamba. Often observed feeding vigorously on the bait container, this predatory species group displayed a relatively widespread distribution and a great attraction to the bait container (σ = 1.1360). Conversely, Yellowfish were the most under dispersed species group and required a significantly higher sampling effort (σ = 0.0341). This species is described as a potamodromous shoaler and omnivore. During the survey, the species was rarely detected and if observed the species were in relatively large schools and not feeding on the bait container. The species groups represent two distinctly different interactions with the BRUVS in space, and consequently, their sampling requirements differ significantly. Understandably, there could be an effect of predator-prey interaction upon the observation of species groups at the bait container. Mlamba, Kampango and Predatory haplochromines may influence the observation likelihood of smaller potential prey species such as *Metriaclima estherae*. Although undertaken in the marine realm, Phenix et al. (2019) investigated the effect of predators on mobile prey within BRUVS field of view. The authors elaborate upon how a prey species is more likely to display avoidance behaviours in the presence of significantly larger predators and in open-area habitats (i.e. sand expanses). Although no predation events took place during the video analysis of this study, further investigation into the interaction of bait, predator and prey should be considered in future studies.

Evidence of the BRUVS adequate sampling success rate, their ability to collect species-specific data and its direct contribution to the conservation of local fish species provides a platform from which one can motivate its employment as a long-term monitoring tool in the lake system. The effective use or applicability of BRUVS in the lake are, however, underpinned by well understood and robust SOPs. Understanding the fundamental aspects of BRUVS employment required for monitoring was investigated in this chapter. In an ideal hypothetical monitoring scenario, one would be able to deploy multiple BRUVS indefinitely within a study site. This is understandably not feasible in actual monitoring ventures operating with finite resources. With each BRUVS deployment and project, the employer must endure both monetary and logistical costs. Bernard (2012) discusses the actual running costs of deploying BRUVS and although documented for use in marine environments, many of the costs are mirrored when employed in freshwater and highlight the need for optimization of deployment SOPs. Another overlooked limitation of deploying BRUVS is both the storage requirements and analysis effort of the footage. The study's 1351 deployments generated over 338 hours of video footage. All of which was required to be analysed multiple times, linked to meta-data and distributed to multiple stakeholders.

In conclusion, BRUVS may perform effectively as a long-term monitoring tool in Lake Malawi. The systems in their current state, exist as a cost-effective and realistic non-extractive technique able to generate count and distribution data in the lake's sensitive lakeshore habitats. In a real-world scenario, the three-year LNR BRUVS data set and the collected presence-absence data was used to support the first delineation of International Union for Conservation of Nature (IUCN) Key Biodiversity Areas (KBAs) in the Mozambique waters of Lake Malawi. The IUCN Lake Malawi KBA workshop, hosted in 2018 used the species list generated for the LNR, the presence-absence data for selected monitoring species and attending researchers to confirm observations during this process. The presence of either endangered or range-limited species in an investigated area triggered the IUCN's guidelines for a KBA, and subsequently, the delineation of that area could be proposed. A total of 11 species were confirmed entirely from post-analysis of BRUVS footage in three of the LNR study sites. Nkwichi, Metangula and Minos-reef (Meluluco), with a combined area of 64.8 km^2 are to be the first recognised IUCN KBAs in Mozambique and highlight the potential of data able to be collected by BRUVS and its applicability in an African freshwater lake system (Sayer et al. 2019). Most importantly, BRUVS have the potential to contribute to long-term research and conservation efforts at a multispecies-level while limiting effects upon potential future recruitment. Further analysis of the complete BRUVS dataset and inclusion of species length data is, however, required to generate more informative models, perhaps shed light upon the effects of habitat or area upon species abundance and finally, explore the full potential of BRUVS in Lake Malawi.

CHAPTER 4

REMOTE UNDERWATER VIDEO MONITORING OF CHAMBO (OREOCHROMIS SPP.) IN LAKE MALAWI/NIASSA



Aggregation of Chambo swimming over a reef in Mozambique (Top), mature male Chambo sporting its darkened, breeding colouration (Bottom). P.C. Angus van Wyk (SAIAB)

INTRODUCTION

African freshwater fisheries provide a plethora of economic and societal benefits to the countries in which they exist, and for the communities that they support (Beeton 2002, Funge-Smith 2018). Lake Malawi (12.183° S, 34.367° E), with its multi-gear and multi-species fisheries, is no different (Tweddle and Magasa 1989). The lake supports the harvesting of live ornamental fish for international export, a developed tourism industry as well as industrial and artisanal fisheries sectors supporting over 1.6 million fishers (Banda et al. 2005a). The combined harvest from these fisheries has been stable since the mid-1980s with the total catch biomass fluctuating around 31,000 tons per annum (Weyl 2005, Weyl et al. 2010). The species composition of the total biomass has, however, changed and there has been a significant decrease in the catch of high-value species (Banda et al. 2005a, Banda et al. 2005b, Weyl 2005, Weyl et al. 2010). This phenomenon has since been documented by multiple fishing centres within both Malawian and Mozambique sections of Lake Malawi (Halafo et al. 2004, Weyl et al. 2010, Irvine et al. 2019). One such high-value species that has since met this fate is the local *Oreochromis* (*Nyasalapia*) species, Chambo (Turner and Mwanyama 1992, Palsson et al. 1999b, Weyl 2001, Banda et al. 2005b, Bulirani 2005, Bell et al. 2012, M'balaka et al. 2018).

Chambo is the collective term for the Oreochromis species flock and is comprised of three tilapiine species (*Oreochromis lidole*, *Oreochromis karongae* and *Oreochromis squamipinnis*). The Chambo stock has experienced a rapid decline in catch since its peak in the mid-1980s (Tweddle and Magasa 1989, Banda et al. 2005b). The period between 1985 and 1999 saw a decline of 85 % (from 9,400 t to 1,400 t) in caught biomass and a reduced contribution to Lake Malawi's annual landings from 49 % to a low of 7 % (Banda et al. 2005b). Consequently, all three species are currently listed as "Critically Endangered" by the International Union for Conservation of Nature (IUCN) Red List of Threatened Species (Sayer et al. 2019). The decreased Chambo catch, a high-value product in both Malawi and Mozambique, has had a significant socio-economic impact on the national economy and the Malawian fisheries sector experiences an estimated loss of over 21 million USD per annum as a result (Banda et al. 2005b, Weyl 2005). An area of particular importance to the fishery due to its relatively high productivity and available long-term catch data, is the South East Arm (SEA) of Lake Malawi (Tweddle and Magasa 1989, Banda et al. 2005b, Weyl 2005, Weyl et al. 2010). The SEA, while contributing only a fraction of the lake's total surface area, produced 70 % of the total Chambo catch in the fisheries peak year, 1985 (Banda et al. 2005b).

Human-induced factors have contributed significantly towards the observed collapse of the Chambo stock, and negative drivers such as gross overfishing, habitat destruction and general non-compliance have all played a pivotal role in the stock's demise (Weyl 2001, Banda et al. 2005b, Bell et al. 2012). This phenomenon is not, however, unique to Lake Malawi but is mirrored at a global scale in both freshwater and marine realms and has drawn the focus of many a national address and great scientific discourse (Dudgeon et al. 2006, Worm and Branch 2012). Subsequently, there exists a suite of tried-and-true management techniques available to management of state fisheries sectors which may be employed to address the resultant adverse effects. One such strategy which has been used to good effect for marine and terrestrial conservation is the closure of crucial areas to harvesting and the demarcating of protected areas. These protected areas act to reduce disturbance to local species and encourage naturally occurring processes to drive system-wide change (Roberts et al. 2001, Suski and Cooke 2007). The development and implementation of these reserves in freshwater habitats have however, been relatively slow-paced and whilst been recognised to have great potential, more working case studies are required (Suski and Cooke 2007).

Demarcation of protected areas - at a state policy level has been achieved in Lake Malawi. On the Malawian lakeshore, the Lake Malawi National Park (14.065° S, 34.885° E) was gazetted in the 1980s with the overarching aim to protect and better manage the 94 km^2 of lakeshore and the freshwater biodiversity found within the park's large bay and surrounding rocky island habitats (Bootsma 1992). The Lake Niassa Reserve (LNR) (12.696° S, 34.814° E) was announced in 2011 and exists as the first and only protected freshwater lake system in Mozambique with a total surface area of 478 km^2. The LNR extends from the northern fishing centre, Cobue (12.140°S, 34.760°E) to the southern Lussefa River mouth bordering the Meluluco fishing centre (12.907°S, 34.767°E). As of 2019, three IUCN Key Biodiversity Areas have been delineated in the LNR. These areas aim to safeguard the diversity of lakeshore fish communities (IUCN 2019).

Published fisheries data for the LNR are relatively scarce in comparison to the Malawian fisheries sector and the local fishery is predominantly characterised by artisanal fisheries. Weyl et al. (2017) discuss the catch, effort and species composition of the artisanal fisheries within the LNR. The local species composition of catches is dominated by usipa and pelagic haplochromines which, when combined, contribute to more than 80 % of the fishery's total landings (Weyl et al. 2017). There is also evidence of broad scale overfishing within the inshore and demersal fisheries. Local catch per unit effort in relation to gears used by the inshore and demersal fisheries showed a steady decline between the years 2006 and 2016 (Weyl et al. 2017). Regulatory measures initiated by both states to address overfishing focus mainly upon user licencing and gear regulations. Effort control, such as seasonal closures of the Chambo fishery by the Malawian state, are only included within legislation for the industrial fishing sector (Weyl et al. 2010).

Further efforts focussed primarily upon Chambo management and conservation have been undertaken in Malawi. These efforts included the establishment of a long-term habitat enhancement, monitoring and restocking program (Jamu et al. 2003, Banda et al. 2005b). Formalised by the "Save the Chambo Campaign" and spearheaded by the Malawi Fisheries Department, the Malawian program began in 2003 and laid out a 10-year plan to restore

Chambo stocks to maximum sustainable yields recorded before the fisheries collapse experienced three decades ago (Banda et al. 2005a). The program aimed to, among other objectives: restore, monitor and protect critical shoreline Chambo habitat, as well as utilise the local aquaculture industry to restock and bolster recruitment potential (Banda et al. 2005a).

Generating biomass or abundance through such strategies is a well-discoursed area of research and is generally achieved by the production or sustaining of habitat availability and food resources while simultaneously limiting fishing mortality and habitat destruction (Bohnsack and Sutherland 1985, Grossman et al. 1997, Banda et al. 2005b). Monitoring the success of these programs or determining what areas are optimal does, however, present a suite of challenges. This statement is particularly true when operating in sensitive or deep-water ecosystems and over a long-term study where traditional monitoring techniques such as netting, trapping and underwater visual consensus may prove to be logistically infeasible or ineffective (Bernard 2012).

Schools of both breeding and juvenile Chambo have been observed within such habitats of the Lake Niassa Reserve and observed at depths of 50 m as well as upon 1 m grass beds (van Wyk et al. 2017). Furthermore, the brushparks created during “Save the Chambo Campaign” were constructed purposefully out of local brush and wood to limit the effectiveness of traditional netting techniques and limit their exposure to fishing exploitation (Banda et al. 2005b). Effective monitoring of these habitats are however, imperative to the management of Chambo. Key objectives of the “Save the Chambo Campaign” framework were those related to monitoring the success of Chambo restocking programs from the aquaculture industry and the formation of fisheries monitoring plans (Banda et al. 2005a).

Non-extractive monitoring techniques may prove to be the most effective method for long-term monitoring of both the LNR and Lake Malawi National Park. Chapter 3 discussed the potential for Baited Remote Underwater Video Systems (BRUVS) to be employed as a monitoring tool for selected species and was determined to be both effective and realistic over a 10-year monitoring scenario in Lake Malawi. Developed initially as a mono-camera system, BRUVS have since been modified to utilise two cameras, in a stereo configuration (Cappo et al. 2003). The advancement permits an overlapping field of view and subsequently, stereo-vision (Cappo et al. 2003). Stereo-Baited Remote Underwater Video systems (Stereo-BRUVS) differ from traditional monitoring techniques as they present an opportunity to collect both relative abundance counts (MaxN in the case of this study) as well as fish length measurements in a non-extractive and relatively non-destructive manner (Willis et al. 2000, Watson et al. 2005, Cappo et al. 2006, Harvey et al. 2007, Langlois et al. 2010).

This chapter aims to utilise a three-year BRUVS dataset (2016-2018) and investigate the potential for BRUVS to effectively monitor and make inference upon Chambo distribution and size-structuring in both the SEA of Lake Malawi and the LNR. The objectives to be accomplished were, in both countries' waters to:

(1) identify key covariates which may affect both the presence and abundance of Chambo, and

(2) to determine possible differences in Chambo size structure between locations, habitat and depth profiles.

The working hypothesis for this chapter is that in those localities where Chambo are exposed to greater fishing pressure, the abundance and detection probability will be lower. Furthermore, the extensively fished populations will vary in size structure as larger, more valued individuals are disproportionately removed from the fisheries.

MATERIALS & METHODS

Study areas

Lake Niassa Reserve, Mozambique

The Study areas in Chapter 4 remain unchanged from Chapter 3. Both Chapters made use of similar Materials and Methods and are written in a publishable format.

The Lake Niassa Reserve (LNR) is situated on the eastern-central lakeshore of Lake Malawi, bordering the single country of Mozambique (Figure 4.1). The Mozambique shoreline is approximately 245 km in length. The selected study sites (Table 4.1) were characterized by three dominant shoreline categories namely: rocky, vegetated banks and sandy beaches (Halafo et al. 2004). The Niassa province of Mozambique, which borders the LNR and has an estimated population of 1,027,037 people, is the country's most sparsely populated province and experiences distinct rainfall seasons. The rainy season occurs annually between November and April and the dry season between May and October (Halafo et al. 2004). Although exposed to freshwater flooding and strong surface winds, Lake Malawi is a meromictic body of water and the oxic-anoxic boundary existing at ±250 m remains fixed due to a thermo-chemical gradient (Eccles 1974).

The LNR has an area of 478 km^2 and extends 4.83 km offshore from the Mozambique shoreline. The northernmost study site selected was offshore of the town Cobue, and the furthest south was offshore from Meluluco (Table 4.1). The LNR's delimited waters experience relatively warm surface temperatures ranging from 24-29°C and the lake's water is considered to be alkaline and ranges from pH 7.7–8.7 (Eccles 1974). It must be noted that relative to the productive waters of the southern Malawi lakeshore, the LNR is relatively data-poor. This paucity in data includes both long-term multispecies catches as well and community demographics data.

In the pioneering fisheries research by Bernacsek and Massinga (1983), a total of 42 fishing centres were identified on the Mozambique lakeshores of what is now known as the Lake Niassa Reserve. The researchers estimated the total annual catch of Metangula (the reserve's predominant fishing centre) was roughly 1690 tons/year and landed by approximately 3382 fisherman and 1203 operating fishing boats (Bernacsek et al. 1983). This insightful study was followed by a 15-year dark period whereby political unrest played a significant role in the lack of fisheries monitoring in Lake Malawi. Monitoring would continue in 1998 by the Instituto de Investigação de Pesquera (IIP) and later form the part of the research published by Halafo et al. (2004). The 2004 study reported that there had been a significant increase in fishing effort throughout the studied fishing centres between the years 1983 and 1999 (Halafo et al. 2004).

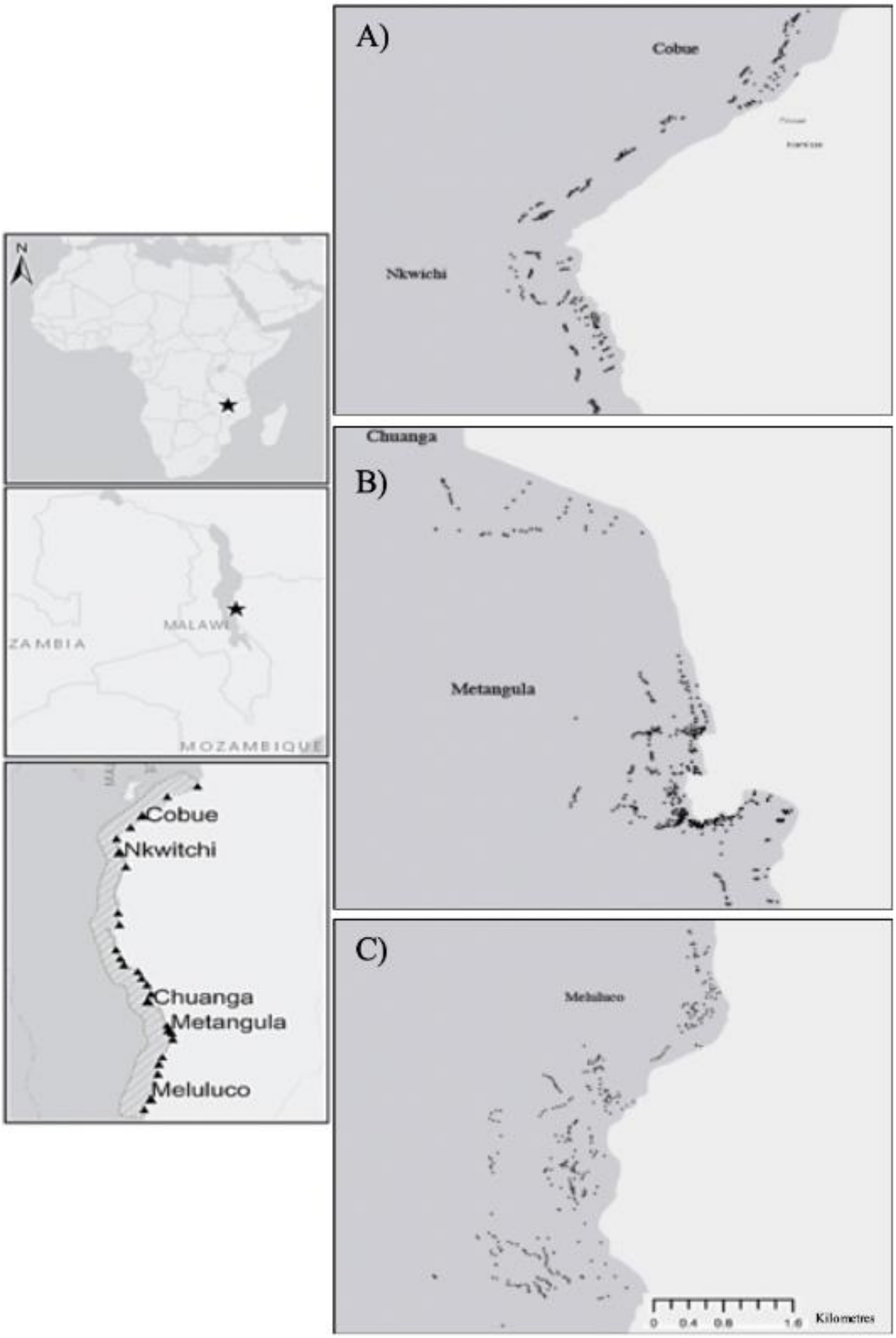


Figure 4.1: Map displaying baited remote underwater video systems (BRUVS) deployments in the Lake Niassa Reserve, Mozambique between years 2016-2018. A) Displaying the most northern study sites, Cobue and Nkwichi B) Chuanga and Metangula and C) the southernmost study site, Meluluco. Symbol (●) represents individual BRUVS deployments and the symbol (▲) represents recognised fishing centres.

Halafo et al. (2004) investigated the catch of six active fishing centres in the Lake Niassa Reserve, two of which are study sites in this study. Metangula (12.696°S, 34.816°E) and Meluluco (12.907°S, 34.767°E) are both active fishing centres (Table 4.1 & Figure 4.1). Although motorised vessels are in operation, the fishery is dominated by artisanal fishers who, employ a wide range of gears – including chilimera nets, long lines and gill nets (Halafo et al. 2004). These fishers target multiple rock-dwelling and pelagic cichlid species, local catfish (Mlamba) and a schooling pelagic cyprinid, Usipa (*Engraulicypris sardella*). Cobue (12.140°S, 34.760°E) is the most northern study site and similar to both Metangula and Meluluco in terms of fisher gears and target species. The scale of the fisheries is, however, smaller by comparison. Access to the fishing centre via road remains a challenge and the town only recently (in 2016) received a connection to the national power grid. South of Cobue is Nkwichi Bay. Nkwichi (12.209°S, 34.707°E) is de facto closed to fishing due to negotiations between a local lodge in the bay and the local community (van Wyk et al. 2017).

Table 4.1: Summary of selected study sites in the Lake Niassa Reserve, Mozambique. Displaying location, dominant bottom types, mean depth sampled, lake-wide fishing pressure relative to all study locations and total baited remote underwater video systems (BRUVS) deployments (Halafo et al. 2004, Weyl et al. 2010).

Site	Coordinates	Dominant bottom type(s)	Depth sampled (m) (*Min, Mean, Max*)	Lake-wide relative fishing Pressure	Total BRUVS deployments (2016-2018)
Cobue	12.140°S, 34.760°E	Sand (>60 %)	1.1, 13.0, 48.6	Low	178
Nkwichi	12.209°S, 34.707°E	Sand, Sand reef	1.5, 10.2, 42.5	Closed	152
Metangula & Chuanga	12.696°S, 34.816°E	Reef, Sand	0.8, 14.4, 63.1	Moderate	377
Meluluco	12.907°S, 34.767°E	Sand, Reef	1.0, 11.7, 64.9	Low-moderate	331

Lake Malawi South East Arm, Malawi

The southern region of Lake Malawi, with a surface area of ±3000 km^2 can be further divided into two distinct areas. Namely, the South Western Arm (SWA) (1 210 km^2) and the South Eastern Arm (SEA) (1 820 km^2). The Mangochi District, which borders the entirety of SEA is of great fisheries importance as it is the production powerhouse of Lake Malawi (Ngochera 2001, Weyl et al. 2004). This district alone has the potential to produce 60 % of the lake's total fish yield (Weyl et al. 2010). All Malawi study sites are within this district and the SEA (Table 4.2).

Table 4.2: Summary of selected study sites in the South Eastern Arm, Malawi. Displaying location, dominant bottom types, mean depth sampled, lake-wide fishing pressure relative to all study locations and total baited remote underwater video systems (BRUVS) deployments (Alimoso et al. 1990, Mdaihli et al. 1992, Weyl et al. 2010, M"balaka et al. 2018).

Site	Coordinates	Dominant bottom type(s)	Depth sampled (m) (Min, Mean, Max)	Lake-wide relative fishing Pressure	Total BRUVS deployments (2018)
Cape Maclear	14.014°S, 34.849°E	Reef, Sand	2.5, 17.9, 42.5	Moderate	104
Monkey Bay	14.071°S, 34.917°E	Reef, Sand	1.7, 14.4, 39.3	Moderate-high	73
Nkudzi Bay	14.181°S, 35.003°E	Sand	1.1, 9.2, 35.7	High	55

Cape Maclear (14.014°S, 34.849°E) is situated within the Lake Malawi National Park and was declared a protected area in the 1980s. The study site has the lowest experienced fishing pressure of all Malawi study sites. The protected areas within the Lake Malawi National Park are, however, primarily positioned to protect the shoreline reef fish communities within the large bay. Subsequently, vast expanses of deep-water sand habitat near to the protected areas remain exposed to the fishery (Banda et al. 2005a). Monkey Bay (14.071°S, 34.917°E) is a state military harbour and commercial port south of Cape Maclear. Vessels entering the bay are strictly monitored, and the setting of nets in the bay is challenging due to the high number of operating motorised vessels (Figure 4.2). Reef and sand habitats directly outside of the bay are, however, targeted by the fishery. The most exposed study selected site in Malawi was Nkudzi Bay (14.181°S, 35.003°E). Relatively small protected areas of habitat are present in the bay. Large gill nets were, however, witnessed being set daily across the opening of the bay during the sampling period.

The shoreline population of the SEA is 404,850 and has a reported annual growth rate of 3.16 %. A contributing factor to the district's relatively high population density and growth rate is the migration of people into the district with intentions to participate in the fishing and agricultural sectors (Donda et al. 2014). The SEA is subjected to similar rainfall patterns to that of the Lake Niassa Reserve (LNR), and the study sites share the same dominant shoreline categories. A differing physical feature between the LNR and SEA is the level of productivity. The SEA has a relatively high productivity level and vast expanses of lake bottom dominated by diatom ooze (Weyl et al. 2010). The three sites (Table 4.2) were selected, similarly to the LNR as to encompass multiple bottom types (i.e. sand, reef and grass) at varying depths as well as including areas of different fishing pressures.

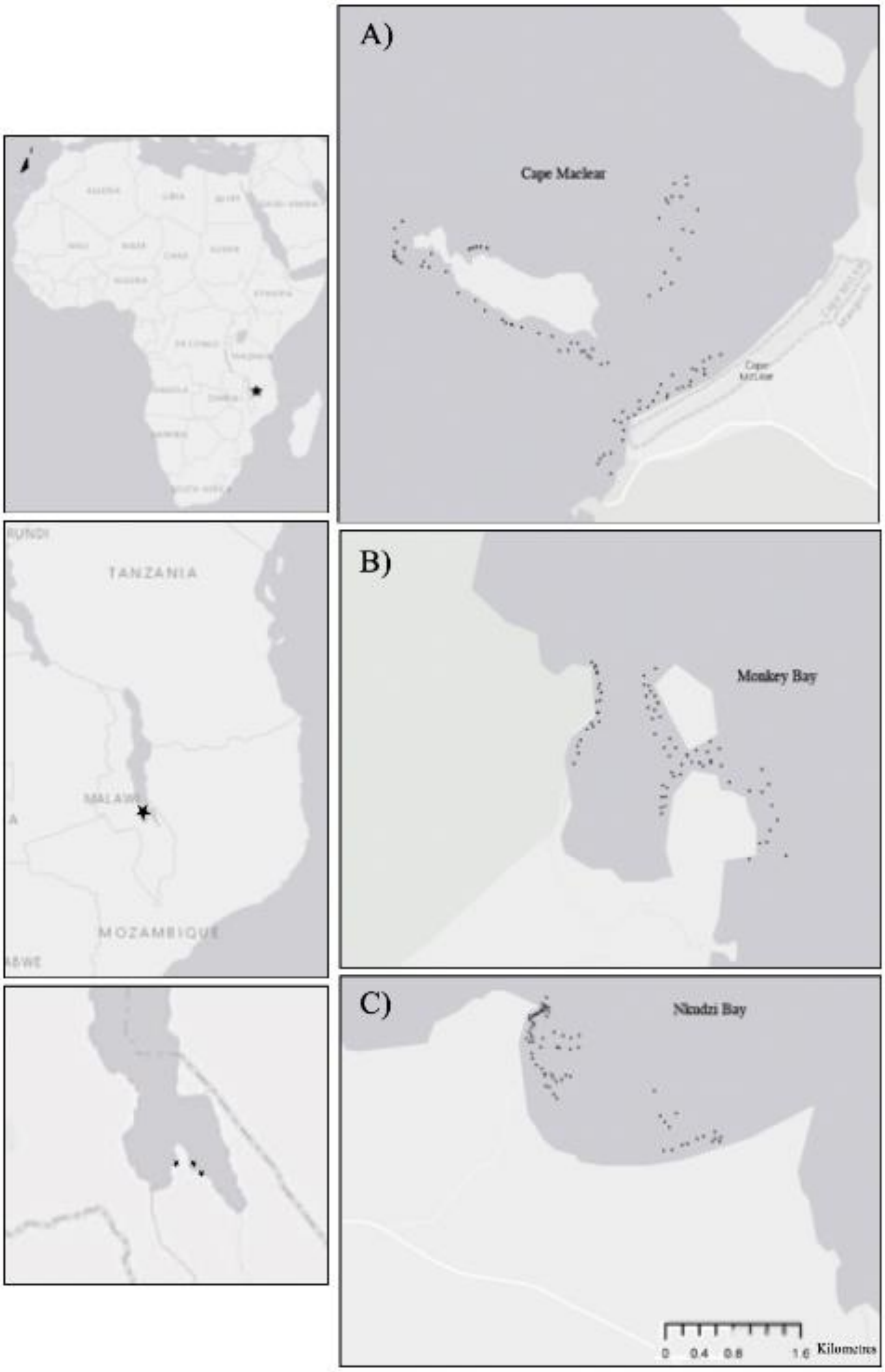


Figure 4.2: Map displaying baited remote underwater video systems (BRUVS) deployments in the South Eastern Arm, Malawi in year 2018. A) the only study location within the Malawi National Park, Cape Maclear B) Monkey Bay and C) the southernmost study location, Nkudzi Bay. Symbol (●) represents individual BRUVS deployments and the symbol (★) represents recognised fishing centres.

Sampling strategy

Mono-BRUVS description

Ten mono-BRUVS were employed for sampling over the years 2016-2017. The systems had a standardised construction and were initially designed to be cost-effective, lightweight and able to operate at depths shallower than 70 m (Figure 4.3). A single GoPro Hero 3 action camera, inside a stock waterproof housing was mounted upon a triangular plastic base plate. The GoPro Hero 3 was set to record with a medium field of view, at 1080p with autofocus disabled and recording at 60 frames per second. The base plate is the central component upon which a pair of steel legs, a mooring line and a single 1 m steel bait arm attaches. The mono-BRUVS were deployed and retrieved by hand from a single research vessel via a mooring line attached to both the system and surface buoy. A single perforated PVC container was positioned on the bait arm directly within the camera's field of view and acted as a bait container.

Before each deployment, a standardised 100 g of crushed Usipa (*Engraulicypris sardella*) was placed into the bait container. Usipa is an affordable and of-least-concern oily-pelagic species native to Lake Malawi (IUCN 2019). A pilot study in the Lake Niassa Reserve was conducted in 2015 and determined that 100g of Usipa was an adequate amount of bait to monitor fish species on both shallow sand and reef habitats and was found to be more effective at observing species than non-baited systems (Weyl et al. 2016).

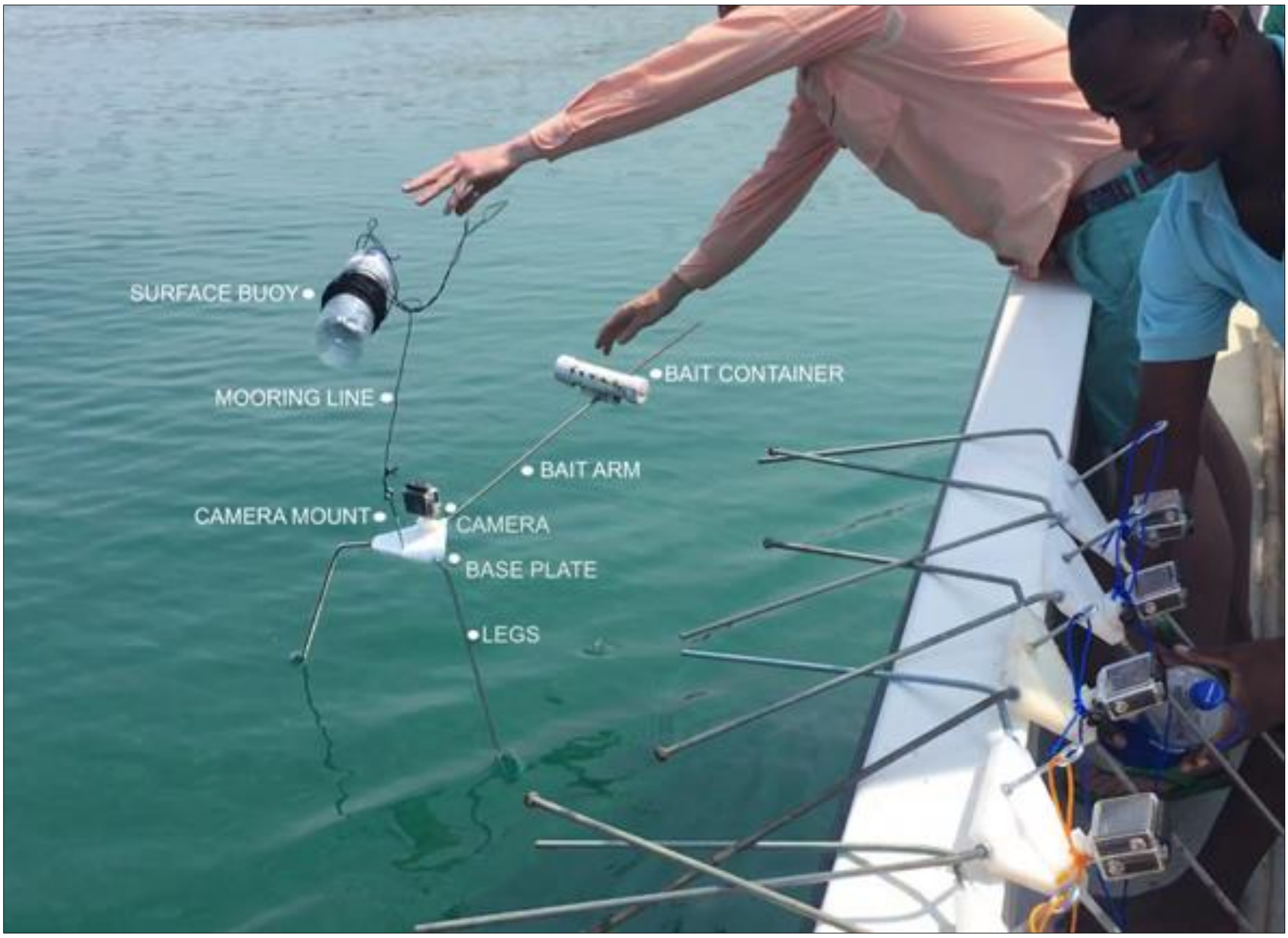


Figure 4.3: Annotated photograph of mono- Baited Remote Underwater Video Systems (BRUVS).

Stereo-BRUVS description

Sampling in the year 2018 employed stereo-Baited Remote Underwater Video Systems (stereo-BRUVS) only. The stereo-BRUVS were constructed of a stainless-steel frame, two GoPro Hero 5 action cameras (enclosed within aluminium housings), a single PVC bait attached to a 1 m bait arm and an Onset HOBO Water Temp Pro v2 logger (Figure 3.4). Each GoPro Hero 5 was set to record with a medium field of view, at 1080p with autofocus disabled and recording at 60 frames per second. Two systems were deployed and retrieved by hand from a single research vessel. The stereo-BRUVS were heavier in construction and remained attached to a surface marker buoy for retrieval via a length of reinforced mooring line (4 mm Dyneema gp12 floating rope).

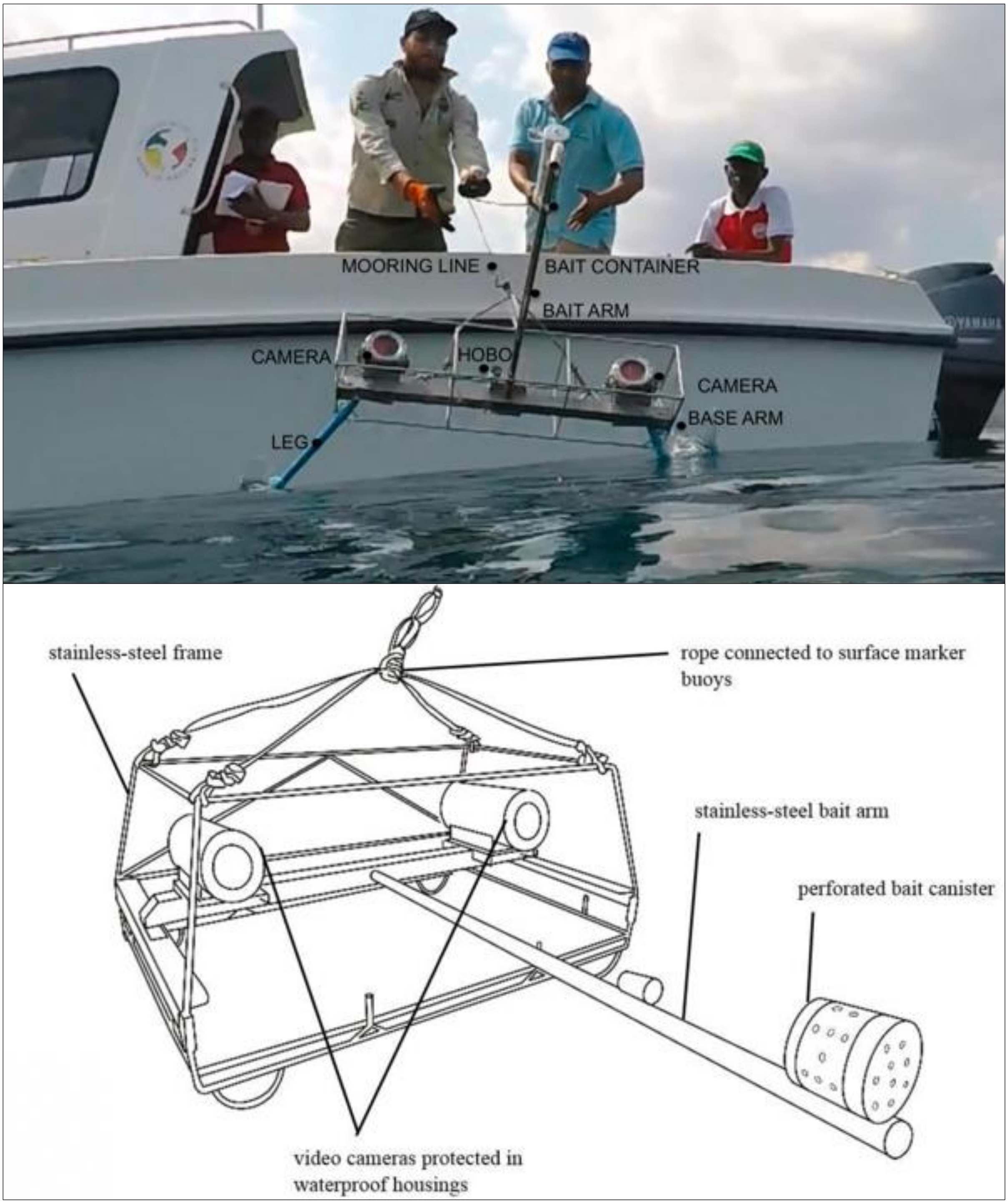


Figure 4.4: Annotated photograph of stereo-baited remote underwater video systems (BRUVS) deployment and annotated diagram of general stereo-BRUVS design (Harvey and Shortis 1995, Heyns-Veale et al. 2016).

The two cameras (left and right), were mounted 0.7 m apart upon a steel base arm and the cameras on both systems face inward at a standardised angle of 8^{o}. The camera lenses and positioning of the housings allow for an overlapping stereo field of view and subsequently, the capability to collect length measurements during video analysis (Harvey and Shortis 1995). Calibration of the camera systems was conducted both before and following the completion of the field trips and conducted using CAL Software and proprietary equipment (SeaGIS Pty. Ltd.). The bait container was positioned directly within the cameras field of view, and all deployments were filled with 100g of crushed Usipa (*Engraulicypris sardella*).

Sampling approach

Selection of deployment sites was achieved using a stratified random sampling design. Selected study sites were initially explored using a motor vessel and an echo-sounder. Results from the exploration were then fit to a local map layer using ArcMap 9 and 10 (Environmental Systems Research Institute, Redlands, CA, U.S.A.). Both depth profiles and habitat type were noted on the map and broad sampling zones within each study site were demarcated. Within each of these broader zones, random deployment points (a minimum of 50 m apart) were then selected

The stratified random sampling design was constructed to ensure an even allocation of sampling sites within the level of the habitat strata. The four habitat types classified were reef, sand-reef, sand and grass. The duration of each deployment was 15 minutes and was taken from the time that the system settled on the benthos of the lakebed. The deployment duration was achieved by noting the location of the system from the vessel's echo-sounder. The sampling approach was standardised for all study sites (Malawi & Mozambique) and completed between months September to November.

Video analysis

All video data was analysed using EventMeasure Software (SeaGIS Pty. Ltd.). The software, as discussed in Chapter 2, is a paid-for software specifically designed to collect data from video footage – including stereo video. The software's period creator function was used to create a watching period of 15-minutes and named "*analysis*". The creation of the period was to ensure all the observer's collected data remained within a standardised time interval. In the instances of BRUVS being rotated by an underwater current or visually obstructed, the deployments were included in the analysis and observed for as long as feasible.

Using available literature, all selected monitoring species matching their respective descriptions and able to be positively observed within the standardised time frame were included in the analysis. Individual image frames of clearly presented individual fish were also collected during analysis. These images were stored and later used to develop a working species identification table.

Estimated species abundance was calculated using a MaxN approach. As discussed in Chapter 2, MaxN as an abundance estimate is relatively easy to attain from the footage and decreases the probability of the observer counting individuals more than once (Ellis and DeMartini 1995). The approach selects a single frame from a video sample which captured the highest number of individuals per species. MaxN is therefore considered to be a conservative index or a precautionary approach and is suited to conservation science as it is less likely to overestimate the true abundance (Ellis and DeMartini 1995). Explanatory environmental variables considered to influence species identification and MaxN abundance counts were recorded both before deployment as well during video analysis. The recorded covariates are represented in Table 4.3 and included: depth, locality, habitat type and are ordered by either Continuous or Factorial data.

Table 4.3: Summary of continuous and factorial covariates for *Malawi* (2018) and *Mozambique* (2016-2018). Including number of baited remote underwater video systems (BRUVS) deployments per analysis level and combined total number of Chambo MaxN counts.

				Continuous covariates					
Mozambique					Malawi				
Name	mean	SD	min	max	Name	mean	SD	min	max
Depth (m)	12.58	0.36	0.80	64.90	*Depth* (m)	13.13	0.74	1.10	42.50
Temperature (°C)	25.32	0.11	21.10	28.10	*Temperature* (°C)	25.86	0.07	21.70	28.84

Factorial covariates			
Name	levels	Number of BRUVS deployments	Total Chambo MaxN counts
Country			
	Malawi	232	677
	Mozambique	1038	2773
Year			
	2016	342	692
	2017	533	1420
	2018	395	1338
Location			
	Cape Maclear	104	411
	Chuanga	30	45
	Cobue	178	490
	Meluluco	331	924
	Metangula	347	993
	Monkey Bay	73	262
	Nkudzi Bay	55	4
	Nkwichi	152	321
Habitat			
	Grass	115	92
	Reef	398	1072
	Sand	601	1950
	Sand-reef	156	336

Following MaxN analysis, the observer loaded footage from both cameras in the EventMeasure Software (SeaGIS Pty. Ltd.) and fit length data to each individual within the MaxN frame. The observer measured the fork lengths of recorded fish. These measurements were achieved and performed by following the guidelines provided by the EventMeasure Software (SeaGIS Pty.

Ltd.). To limit error, the default error warnings in the software were adhered to at all times – discarding length measurements which were flagged by the software. These warnings included error generated by a fish being measured at too great of an angle, too far away or obstructed due to visibility or obstructions.

Data analysis

Descriptive statistics

Microsoft Excel 2016 was used for the compilation of all metadata and R commander used for the running of all descriptive statistical summaries and tests (Fox and Bouchet-Valat 2019). These statistical tests included calculating the means, counts and standard deviations of the factors and covariates as well as the running of statistical tests to determine the significance or lack thereof between selected means - included the use of t-tests during data exploration and determining possible differences between recorded variable means such as *Depth* and *Temperature* by study sites.

Abundance and detection probability modelling

Generalised linear models (GLMs) were used to investigate the effect of selected predictor variables upon both Chambo abundance as well as the species' detection probability. These predictor variables (covariates) included: *Country*, *Location*, *Habitat*, *Depth* and *Temperature*. GLMs are considered useful for the analysis of ecological count data (such as the data in this study) which tends to contain non-normal response data in relation to the selected predictor variables (Zuur et al. 2009).

Before applying the GLMs to either the abundance or detection probability data, both datasets underwent thorough data exploration following the guidelines discussed by Zuur et al. (2009).

The exploratory steps included identification and removal/retention of impactful outliers and assessing potential collinearity between continuous covariates as well as between factorial and continuous covariates (Zuur et al. 2009). The initial fully saturated models for both datasets were as follows **(4.1)**:

$$(Ey) = \alpha + ß2\ (\mathit{Country}) + ß3\ (\mathit{Location}) + ß4\ (\mathit{Habitat}) + ß5\ (\mathit{Depth}) + ß6\ (\mathit{Temperature}) + \varepsilon$$

Where *(Ey)*, the expected response was assumed to follow a Poisson distribution family for the abundance model and a binomial distribution family assumed for the detection probability model. The predictor function $\alpha + ß\ldots$ consists of multiple factorial and continuous covariates which may explain a proportion of variability observed in the response variable *(y)* and ε is the modelled error term (Zuur et al. 2009).

Model validation for over-dispersion was then performed on the abundance model count data. The MaxN count data collected for Chambo contained numerous zero detections, and subsequently, the data failed to meet the assumptions of a Poisson distribution. Such that, the variance of the distribution was not equal to the mean. Instead, the data was fit to a quasi-poisson and a negative binomial distribution model, which incorporates an overdispersion parameter, k into the fit model and is deemed more appropriate for overdispersed count data (Zuur et al. 2009). The parameter estimates and standard error for both fit models were compared, and each model led to a similar outcome. A negative binomial distribution model was then selected based upon the functionality of the MASS R Package (RStudioTeam 2019).

Finally, model selection was conducted by fitting the fully saturated model for both Chambo detection probability and abundance (Equation 8) and comparing the recorded Akaike Information Criterion (AIC) score. A stepwise model selection process was then carried out, which compared all possible covariate combinations. By performing a stepwise model selection, one attempts to retain a biologically relevant, best fit model while limiting model complexity. The stepwise model selection recorded the AIC scores at each step, and the model combination with the lowest AIC score was selected.

Length data analysis

Length frequency distributions for Chambo were generated from analysed fork lengths and collected by stereo-Baited Remote Underwater Video Systems (stereo-BRUVS) in the sampling year *2018*. Using Rcmdr and R Studio, the length-frequency histograms of Chambo were plotted as a function of covariates *Country*, *Location*, *Habitat* and *Depth* (Fox and Bouchet-Valat 2019, RStudioTeam 2019). Distributions of each length-frequency underwent data exploration before the running of statistical tests. The data exploration included the performing of a Shapiro-Wilk test to analyse the distribution of each generated length-frequency distribution as well as investigating the level of Skewness or Kurtosis present.

The Shapiro-Wilk test for normality is used to investigate the null hypothesis that a distribution is normally distributed (Ghasemi and Zahediasl 2012). P values < 0.05 attained for a distribution provide evidence to reject the null hypothesis, and there is evidence that the data are not normally distributed. All plotted length-frequency distributions in this study rejected the null hypothesis distributed ($p < 0.05$) and are assumed to be non-normally distributed. Following the Shapiro-Wilk test, measures of Skewness and Kurtosis were performed upon each distribution. Skewness, a measure of symmetry implies how "off-centre" the data in the distribution appears. This measure of symmetry about the mean can provide an insight into whether the data curve is skewed either positively or negatively and the amount skew (Ghasemi and Zahediasl 2012). Kurtosis, however, measures the relative sharpness of the data peak and can provide information upon how the length data is clustered about the mean. Abnormally high peaks or low, excessively dispersed peaks may provide evidence of a non-normal distribution (Ghasemi and Zahediasl 2012).

Given the non-normal distributions of the length-frequency data, a Levene's Test for Equality of Variances was performed to test the assumption of equal variances. The homogeneity of variance is of particular importance when determining what statistical test to employ for determining a difference between means. The Levene's Test may be used as an alternative to the Bartlett test. Both of which, test for the homogeneity of variance. Levene's Test is, however, less sensitive to departures from normality (Garson 2012). The null hypothesis of the test assumes that the variances of the data are equal. P values < 0.05 attained for a dataset provide a reason to reject the null hypothesis and that there is evidence that the data do not have equal variances (Garson 2012). All plotted length-frequency distributions in this study rejected the null hypothesis ($p < 0.05$) and displayed no homogeneity of variance. Therefore, Kolmogorov–Smirnov two-sample tests were used to compare mean length differences.

The Kolmogorov-Smirnov two-sample test is a non-parametric test that compares the distributions of two input data sets (Garson 2012). Which in the case of this study, were the length-frequency distributions as a function of *Country*, *Habitat* and *Depth*. Each covariate was further separated into two levels namely: *Malaw*i vs *Mozambique*, *Sand* vs *Reef* structure and *Shallow* vs *Deep*. The null hypothesis of a Kolmogorov-Smirnov two-sample test is that the distributions of each level are identical. The test is indiscriminate of what parameters may lead to a determined difference (i.e. median, variance or distribution shape) and reports only a D test statistic and a P value (Garson 2012). If the P value is < 0.05, the null hypothesis may be rejected, and there is evidence that the two distributions differ significantly. All distributions tested in this study, *Malaw*i vs *Mozambique*, *Sand* vs *Reef* structure and *Shallow* vs *Deep* rejected the null hypothesis and their length-frequency distributions were assumed to differ significantly ($p < 0.05$).

Graphical representation

The data for all models were visually presented in R Studio (RStudioTeam 2019) using a combination of the following installed packages: Lattice, LatticeExtra, Jtools and ggplot2. The General Linear Model coefficient plot presented in this study was created using jtools and the frequency histograms created using Rcmdr and Lattice. Adobe Illustrator 2019 was employed for all colour enhancements on selected graphs.

RESULTS

Abundance of Chambo in Southern Lake Malawi

Following the data exploration outlined by Zuur et al. (2009), those General Linear Models (GLMs) which failed the overdispersion test (dispersion value > 1) were alternatively fit with negative binomial GLMs. This occurred in all Chambo abundance models. Negative binomial GLM coefficient plots for Chambo abundance are represented in Figures 4.5–4.8. The highlighted horizontal lines are those covariates or factors, selected from the model of best fit, which significantly explain a portion of the variance observed in the Chambo abundance data. Negative binomial GLM coefficient plots are a graphical representation of standard GLM output whereby the model Estimate is represented on the x-axis, and Standard Error, z-values and Pr(>|z|) values represented for significant covariates or factors.

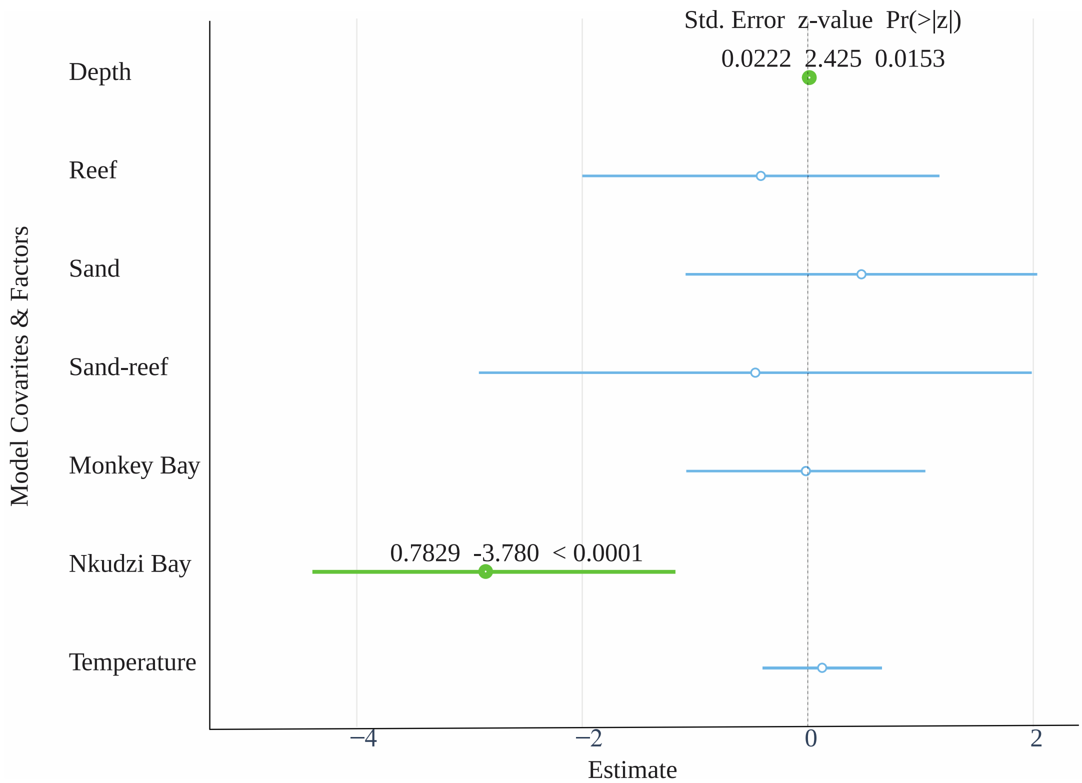


Figure 4.5: Negative binomial general linear model coefficient plot for Chambo abundance in *Malawi*, including plotted Estimate points, Standard Error, z-value and Pr(>|z|). Highlighted points display those covariates and/or factors which significantly affect the abundance model ($p < 0.05$).

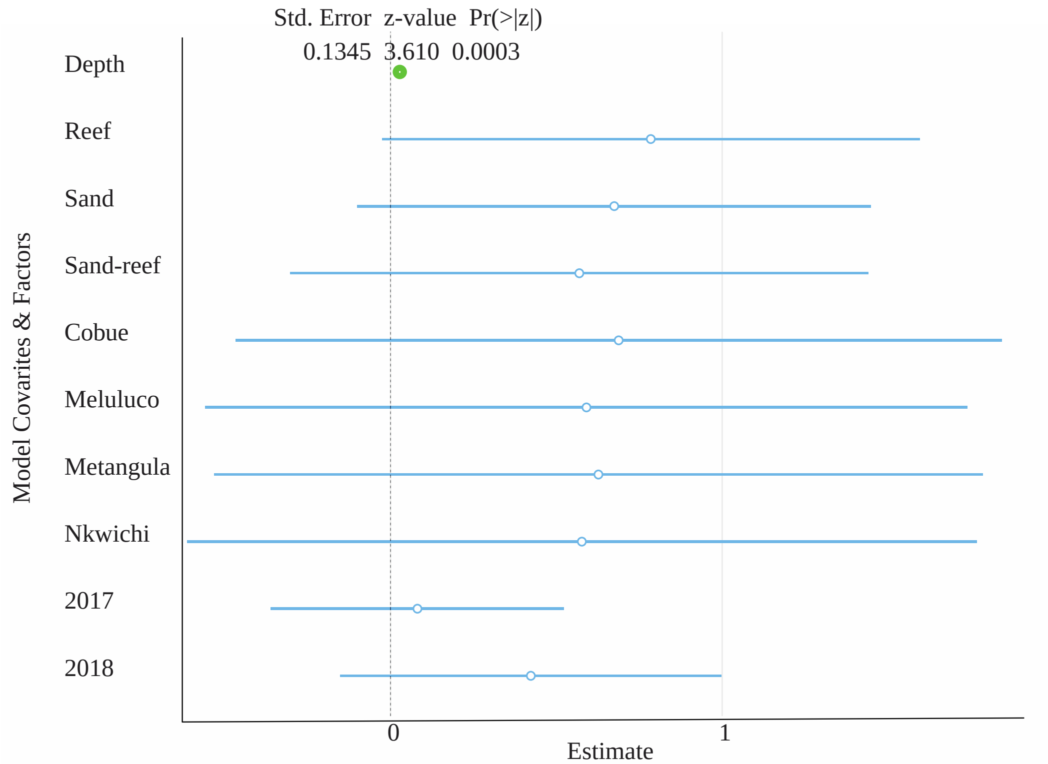


Figure 4.6: Negative binomial general linear model coefficient plot for Chambo abundance in *Mozambique*, including plotted Estimate points, Standard Error, z-value and Pr(>|z|). Highlighted points display those covariates and/or factors which significantly affect the abundance model ($p < 0.05$)

The saturated models fit for those deployments taking place in 2018 only (Figure 4.5 & 4.7) were as follows:

(*Ey*) = α + ß2 (*Country*) + ß3 (*Location*) + ß4 (*Habitat*) +ß5 (*Depth*) + ß6 (*Temperature*) + ε **(4.2)**

Abundance models fit to data from all three years (Figure 4.6 & 4.8) had the following saturated model prior to AIC model selection:

(*Ey*) = α + ß2 (*Country*) + ß3 (*Location*) + ß4 (*Habitat*) +ß5 (*Depth*) + ε **(4.3)**

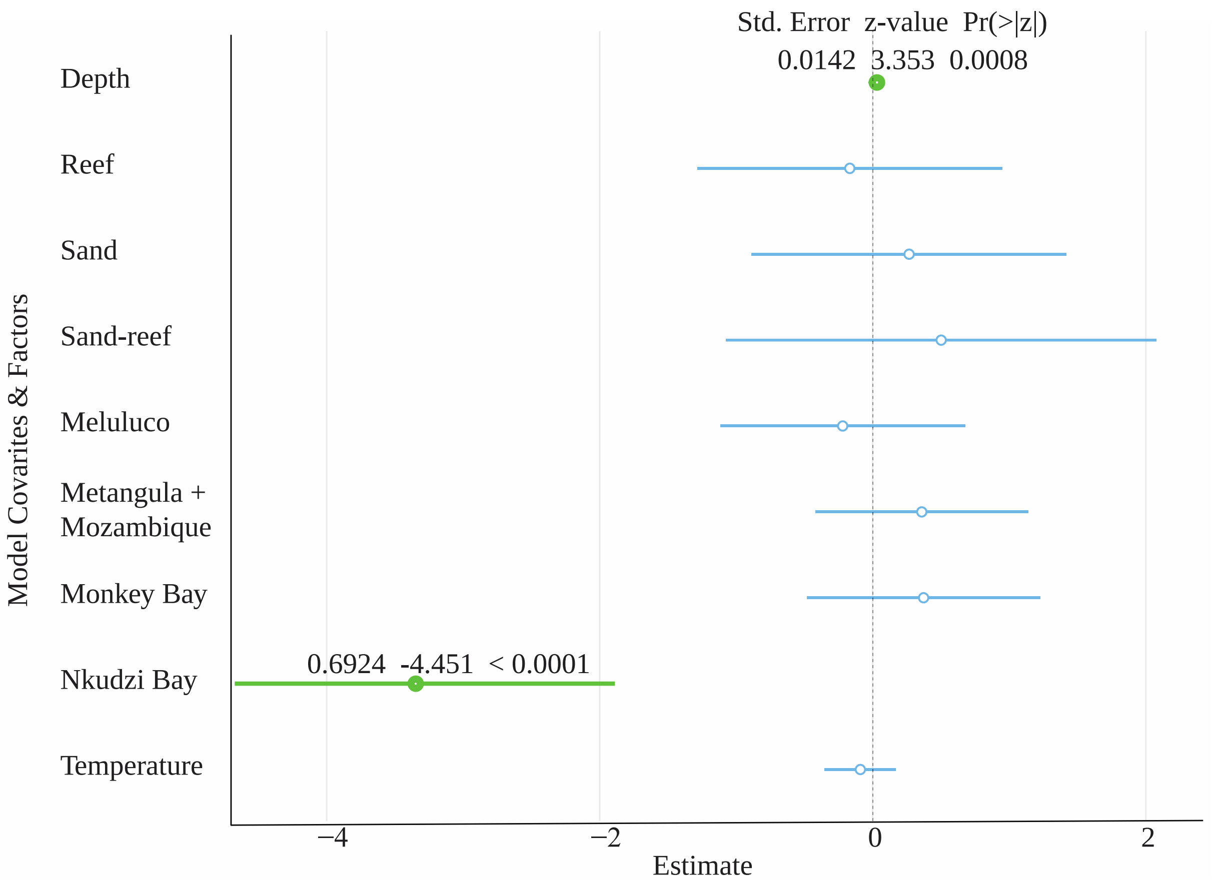


Figure 4.7: Negative binomial general linear model coefficient plot for Chambo abundance in *Malawi* and *Mozambique* (2018 deployments only), including plotted Estimate points, Standard Error, z-value and Pr(>|z|). Highlighted points display those covariates and/or factors which significantly affect the abundance model ($p < 0.05$).

The negative binomial GLM for Chambo abundance in *Malawi* (Figure 4.5) indicated that *Location* ($p < 0.001$) and *Depth* ($p < 0.015$) significantly explained a portion of the variance observed in the Chambo abundance data. *Depth* (Estimate = 0.054) having a positive significant effect and *Nkudzi Bay* (Estimate = -2.969) having a negative effect. In *Mozambique* (Figure 4.6) however, only the continuous covariate *Depth* ($p < 0.001$) had a significant and positive affect (Estimate = 0.034) upon the abundance model. Figures 4.6 and 4.8 included abundance data collected overall of the three years: *2016*, *2017* and *2018*.

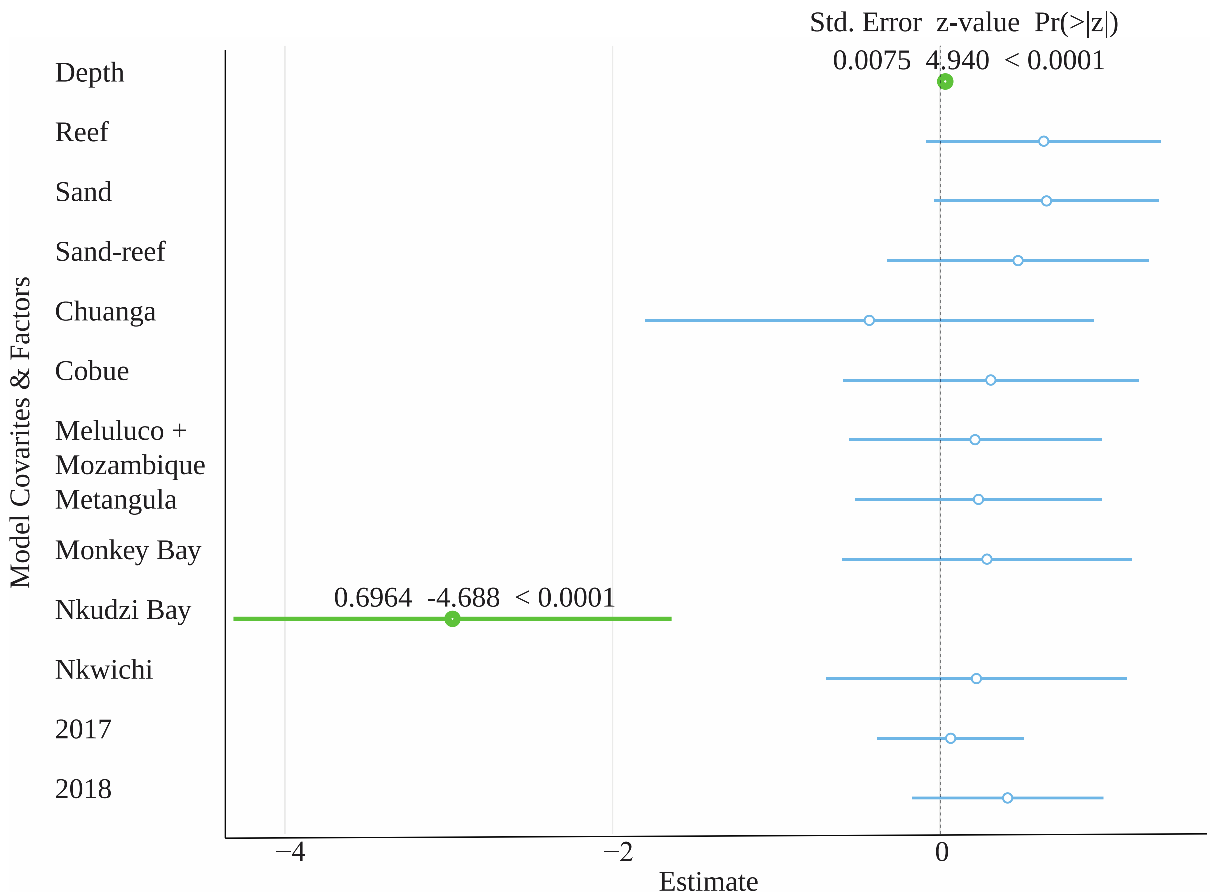


Figure 4.8: Negative binomial general linear model coefficient plot for Chambo abundance in *Lake Malawi* (all Malawi & Mozambique study sites), including plotted Estimate points, Standard Error, z-value and Pr(>|z|) values. Highlighted points display those covariates and/or factors which significantly affect the abundance model ($p < 0.05$).

Figure 4.7 displays the model fit to abundance data collected in the year *2018* only and included both *Malawi* and *Mozambique*. *Depth* ($p < 0.001$) and *Nkudzi Bay* ($p < 0.001$) were the only covariates significantly explaining a portion of the variance. *Depth* having a positive effect (Estimate = 0.048) and *Nkudzi Bay* (Estimate = -3.082) a negative effect. Similarly, the complete model of *Lake Malawi* (i.e. all deployment *Years*, *Countries* and *Locations*) was only significantly affected by *Depth* ($p < 0.001$) and *Nkudzi Bay* ($p < 0.001$). *Depth* having a positive effect (Estimate = 0.037) and *Nkudzi Bay* (Estimate = -3.265) a negative effect.

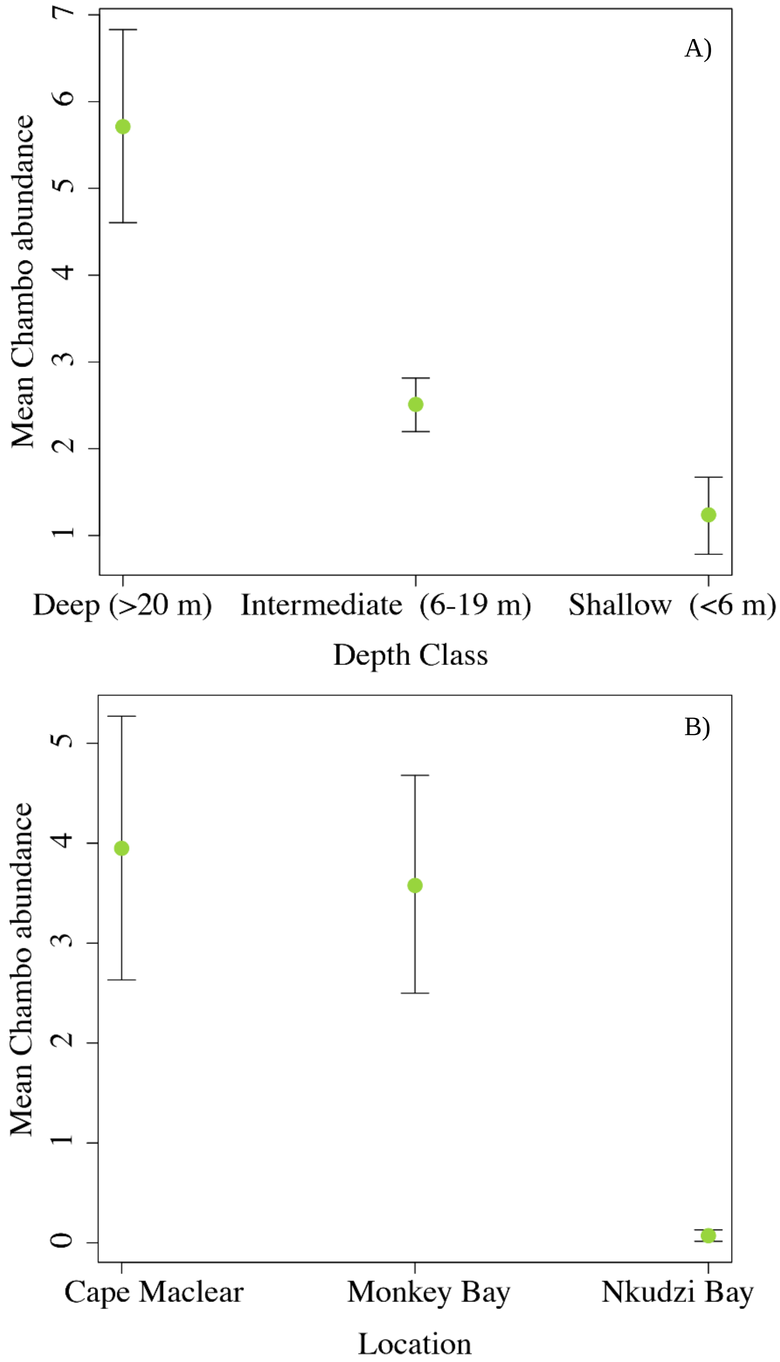


Figure 4.9: Plotted means for Chambo abundance in relation to (A) recorded depth classes and (B) study locations within Malawi. The dashed lines represent the Standard Error of each mean.

Mean Chambo abundance plots were fit to the abundance data to explore further both the significant effect of *Depth* and *Location* upon the abundance model (Figure 4.9). Deepwater habitats (> 20 m) are driving the positive effects of *Depth* upon the abundance of Chambo. Conversely, *Nkudzi Bay* has a much lower mean Chambo abundance in comparison to *Cape Maclear* and *Monkey Bay* (Figure 4.9).

Detection probability of Chambo in Southern Lake Malawi

Unlike the Chambo abundance models, the detection probability data is derived from a binomial distribution. A General Linear Model (GLM) was, therefore, fit to the data assuming a binomial distribution and following the data exploration technique outlined by Zuur et al. (2009). Fully saturated models (i.e. all possible covariates) were fit to the abundance data before undergoing AIC model selection and the model of best fit selected. Similar to the abundance modelling, GLM coefficient plots were fit to the Chambo detection probability data and presented in Figures 4.10–4.13.

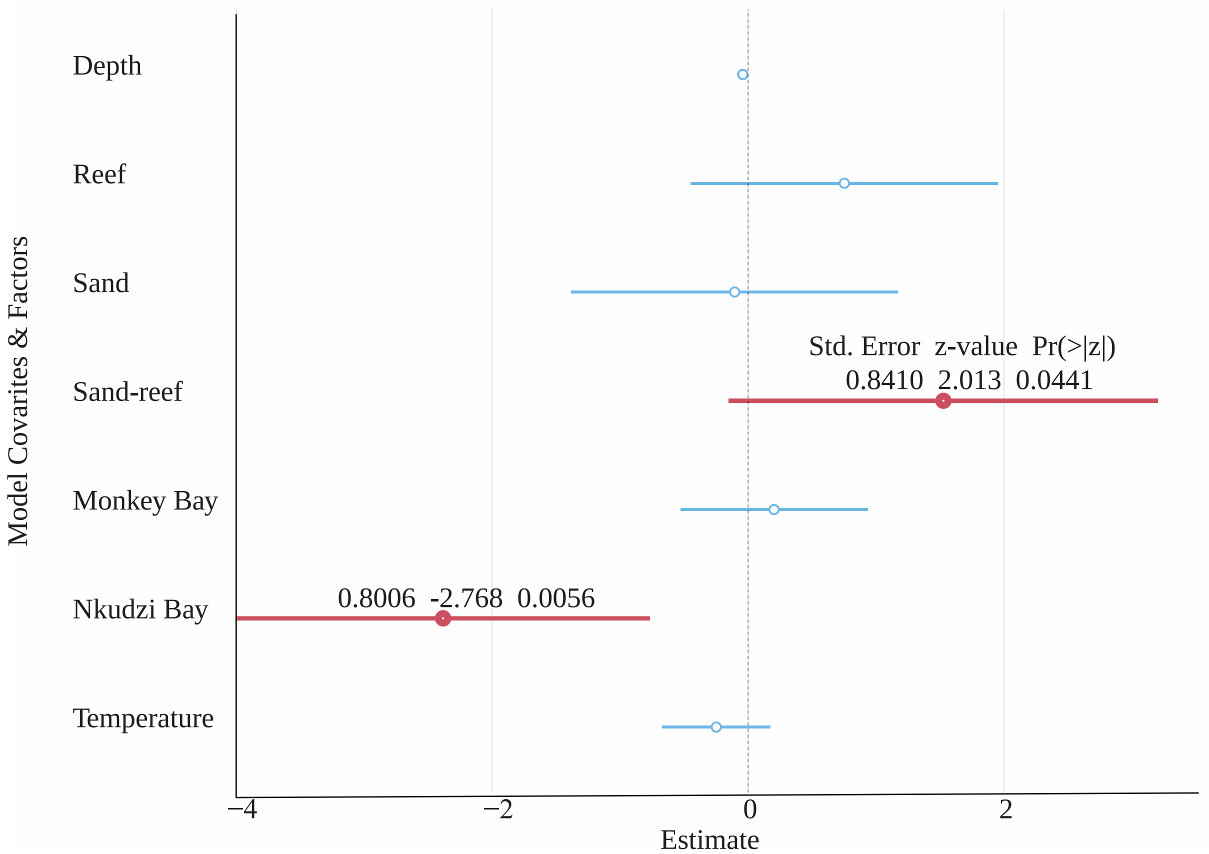


Figure 4.10: General linear model coefficient plot for the detection probability of Chambo in *Malawi*, including plotted Estimate points, Standard Error, z-value and Pr(>|z|) values (2018 deployments only). Highlighted points display those covariates and/or factors which significantly affect the model ($p < 0.05$).

The saturated models fit to the Chambo detection probability data for those deployments taking place in 2018 only (Figure 4.10 & 4.12) were as follows:

$(Ey) = \alpha$ + ß2 (*Country*) + ß3 (*Location*) + ß4 (*Habitat*) +ß5 (*Depth*) + ß6 (*Temperature*) + ε **(4.4)**

Detection probability models fit to data from all three years (Figure 4.11 & 4.13) had the following saturated model prior to AIC model selection:

$(Ey) = \alpha$ + ß2 (*Country*) + ß3 (*Location*) + ß4 (*Habitat*) +ß5 (*Depth*) + ε **(4.5)**

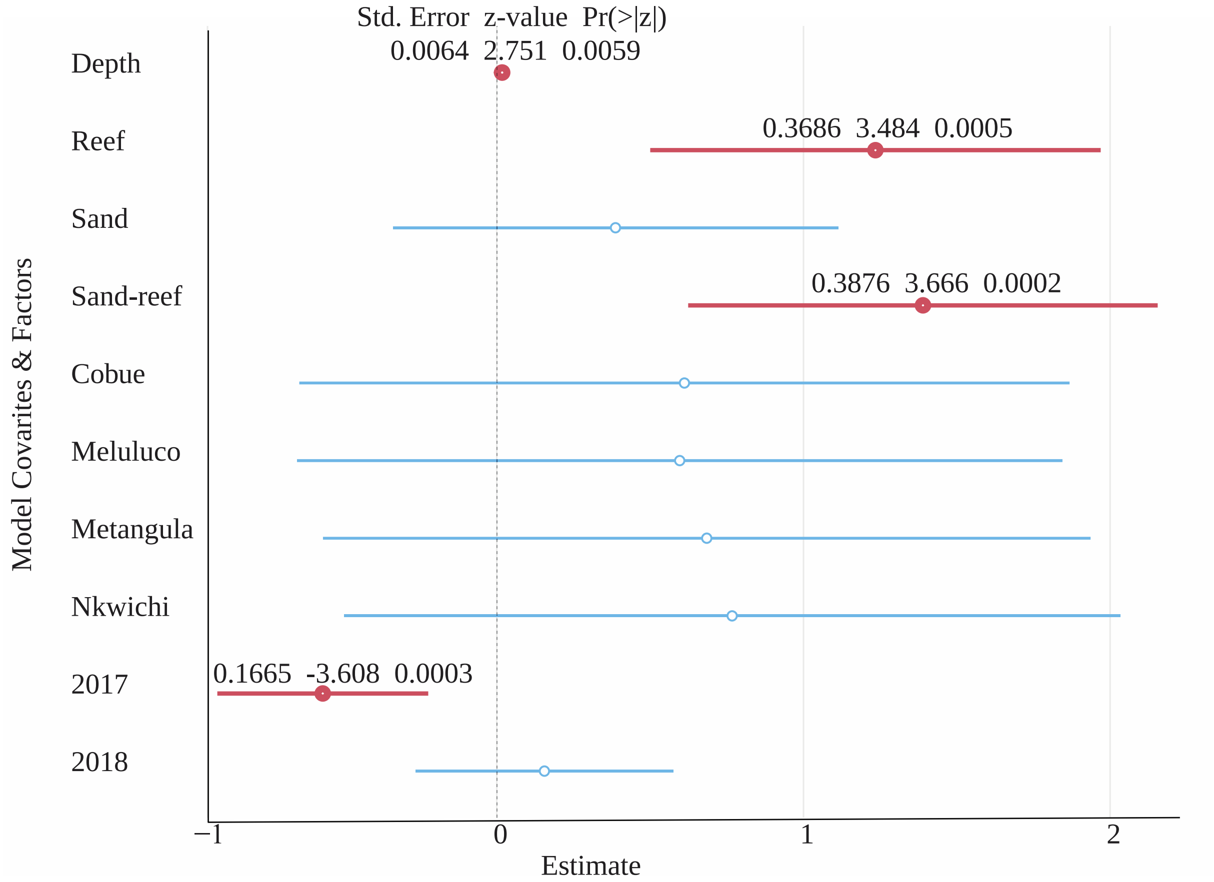


Figure 4.11: General linear model coefficient plot for the detection probability of Chambo in *Mozambique*, including plotted Estimate points, Standard Error, z-value and Pr(>|z|) values. Highlighted points display those covariates and/or factors which significantly affect the model ($p < 0.05$).

The GLM for Chambo detection probability in *Malawi* (Figure 4.10) indicated that *Location* ($p < 0.005$) and *Habitat* ($p < 0.044$) significantly explained a portion of the variance observed in the model. *Nkudzi Bay* (Estimate = -2.216) having a negative significant effect and *Sand-reef* (Estimate = 1.693) having a positive effect.

Although *Sand-reef* ($p < 0.001$) also had a significant effect upon the detection probability of Chambo in *Mozambique* (Estimate = 1.421), *Depth*, the presence of *Reef* and *Year 2017* also had significant effects with p values < 0.001 and Estimate values of 0.017, 1.284 and -0.600 respectively (Figure 4.11)

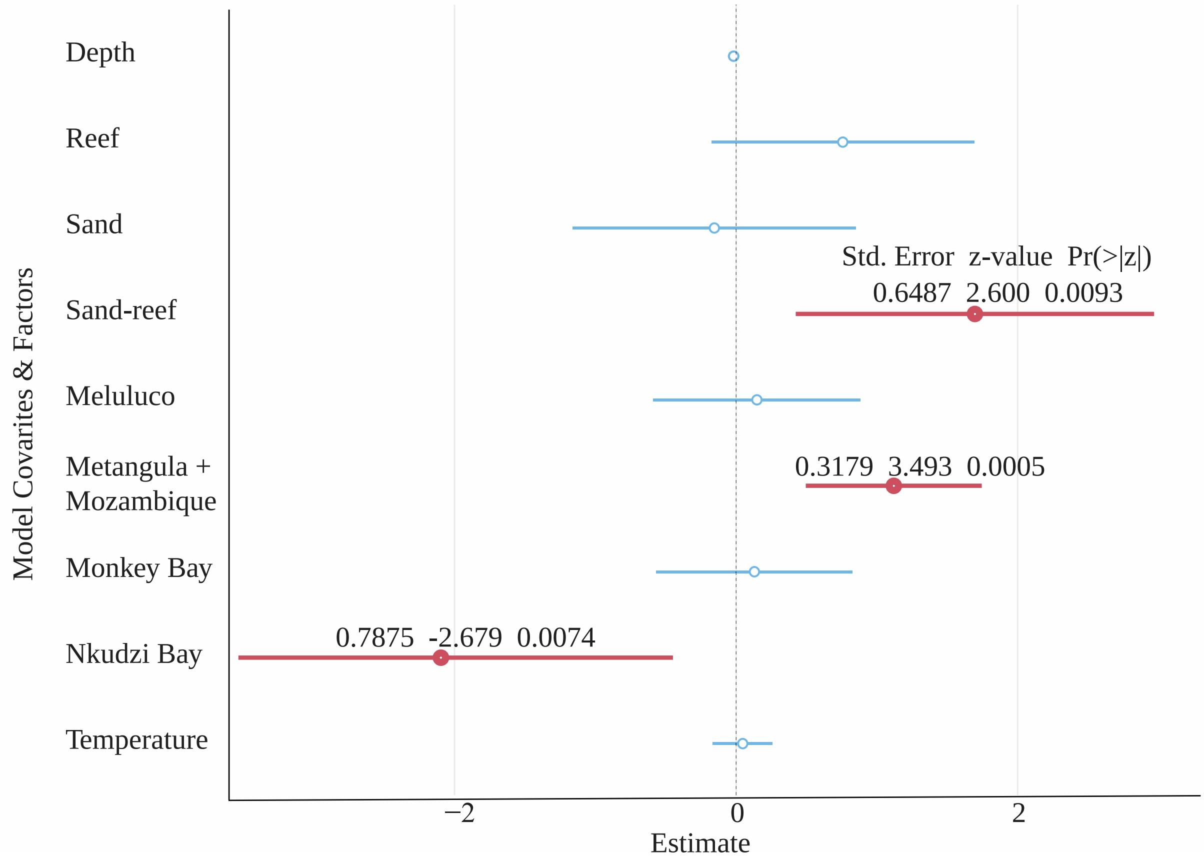


Figure 4.12: General linear model coefficient plot for the detection probability of Chambo in *Malawi* and *Mozambique*, including plotted Estimate points, Standard Error, z-value and Pr(>|z|) values (2018 deployments only). Highlighted points display those covariates and/or factors which significantly affect the model ($p < 0.05$).

Figure 4.12 displays the GLM fit to data collected in the year *2018* only and included both *Malawi* and *Mozambique*. The country *Mozambique* had a significant positive effect upon detection probability of *Chambo* ($p < 0.001$) with an Estimate value of 1.111. The habitat *Sand-reef* and study location *Nkudzi Bay* both had a significant affect upon the model ($p < 0.01$) with respective Estimate values of 1.687 and -2.109. The final fit GLM displays the complete model of *Lake Malawi* (i.e. all deployment *Years*, *Countries* and *Locations*) (Figure 4.13). Again, the country *Mozambique* had a significant positive effect upon detection probability of Chambo ($p < 0.01$) with an Estimate value of 0.826. Both habitats, *Reef* and *Sand-reef* had a positive and significant effect upon the model ($p < 0.001$) with Estimate values of 1.087 and 1.286, respectively. Multiple study locations had a significant effect on the detection probability of Chambo in *Lake Malawi*. All those locations in the country *Mozambique* (*Cobue*, *Meluluco*, *Metangula* and *Nkwichi*), had positive effects upon detection probability of Chambo where *Nkudzi Bay* was the only location with a negative Estimate value of -1.643. The year *2017* was the only year which significantly affected the model ($p < 0.01$) and resulted in a negative Estimate value of -0.0510.

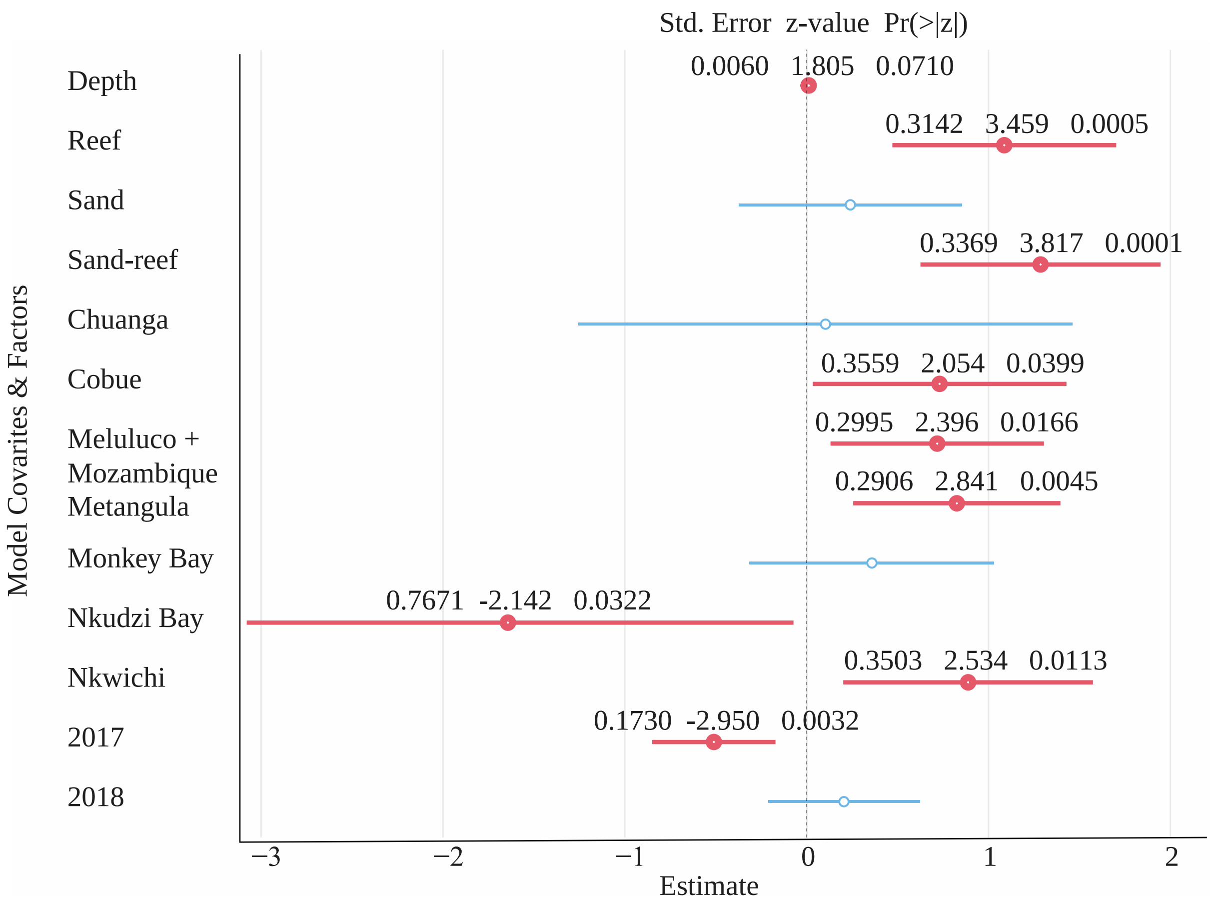


Figure 4.13: General linear model coefficient plot for the detection probability of Chambo in *Lake Malawi* (all Malawi & Mozambique study sites) including plotted Estimate points, Standard Error, z-value and Pr(>|z|) values. Highlighted points display those covariates and/or factors which significantly affect the model ($p < 0.05$).

Plotted means for the detection probability of Chambo were similarly fit to the abundance data to further explore the effects of significant GLM covariates and factors. *Depth*, *Habitat*, *Country* and *Location* all explained a significant proportion of the variance of the detection probability model (Figure 4.14). Deepwater (> 20 m) and Intermediate (6-19 m) habitats are driving the positive effects of *Depth* upon the detection probability of Chambo. Similarly, structure habitats such as *Reef* and *Sand-reef* as well as the country *Mozambique* also display a higher mean probability of detection. *Nkudzi Bay* has a much lower mean Chambo detection probability in comparison to *Cape Maclear* and *Monkey Bay* (Figure 4.14).

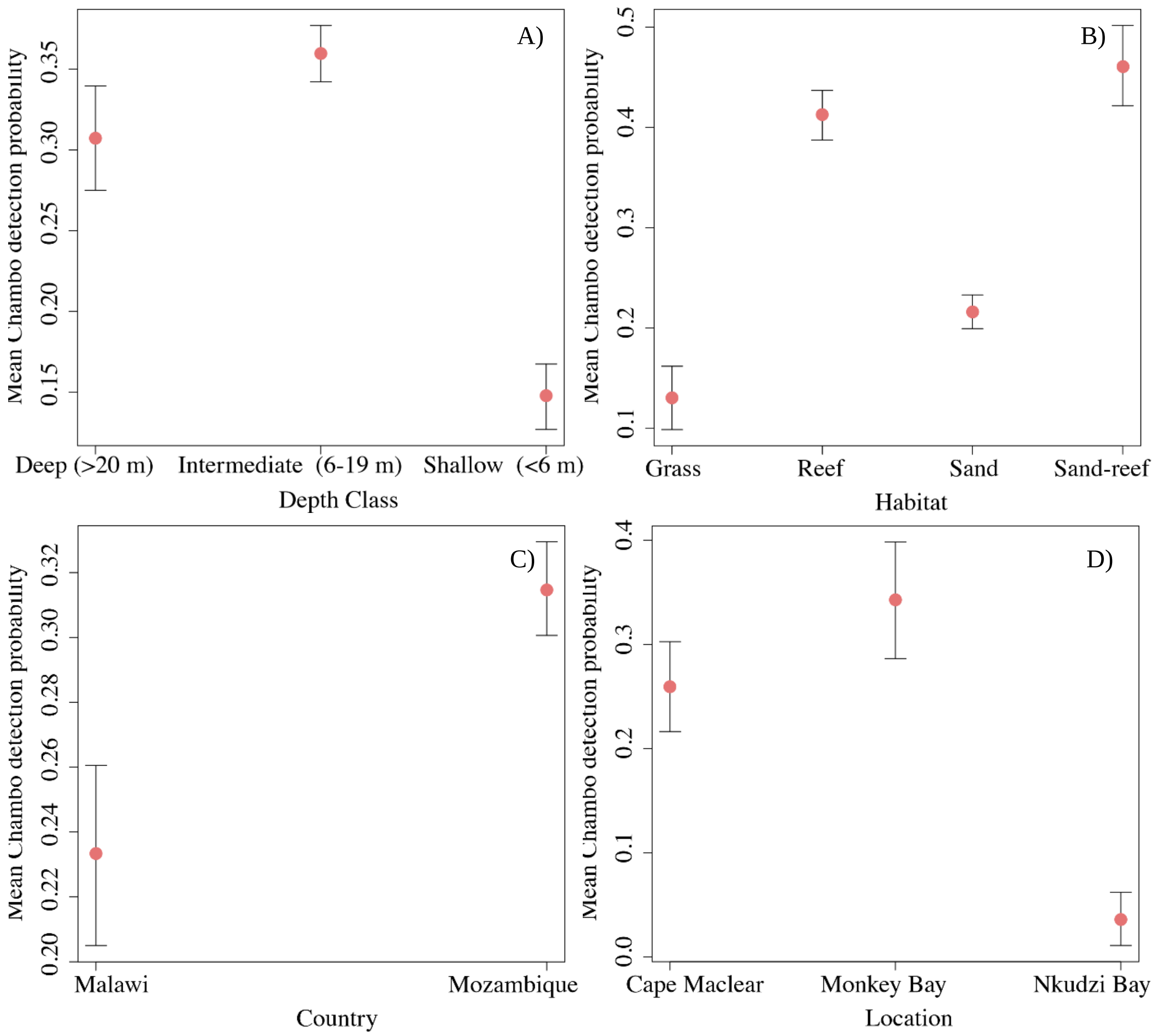


Figure 4.14: Plotted means for Chambo detection probability in relation to (A), recorded depth classes (B), Habitat type (C), Country and (D), study locations within Malawi. The dashed lines representing the Standard Error.

Length frequencies

Fork lengths of Chambo were fit to length-frequency histograms by *Country, Location, Depth* and *Habitat* (Figure 4.15 – 4.18). These lengths were attained from stereo-BRUVS footage during the 2018 sampling year. The complete summary of length data for Chambo is displayed in Table 4.4 and included: depth, locality, habitat type and are ordered by either Continuous or Factorial data. Unlike the analysis of General Linear Model data, *Depth* in this analysis is a factorial covariate. A total of 1,261 lengths were able to be obtained during the sampling period and the average size of Chambo recorded was 164.9 mm (min = 69.5, max = 355.6). The presentation of results followed data exploration, including Shapiro-Wilk tests, Levene's Tests for Equality of Variances and a Kolmogorov-Smirnov two-sample test to examine differences in length distributions further.

Table 4.4: Summary of recorded length data covariates (including mean, minimum and maximum) and total recorded length counts by levels for Chambo in Lake Malawi.

	Length data summary		
	Continuous covariates		
Name	mean	min	max
Length (mm)	164.9	69.5	355.6
Precision (mm)	3.8	0.9	22.7
Range (m)	2.05	0.79	4.92
	Factorial covariates		
	Length counts		
Name	levels	n lengths	
Country			
	Malawi	825	
	Mozambique	436	
Location			
	Cape Maclear	556	
	MB & NKB	269	
	Meluluco	120	
	Metangula	316	
Depth			
	Deep (>20 m)	719	
	Shallow (<20 m)	542	
Habitat			
	Sand	863	
	Reef	370	

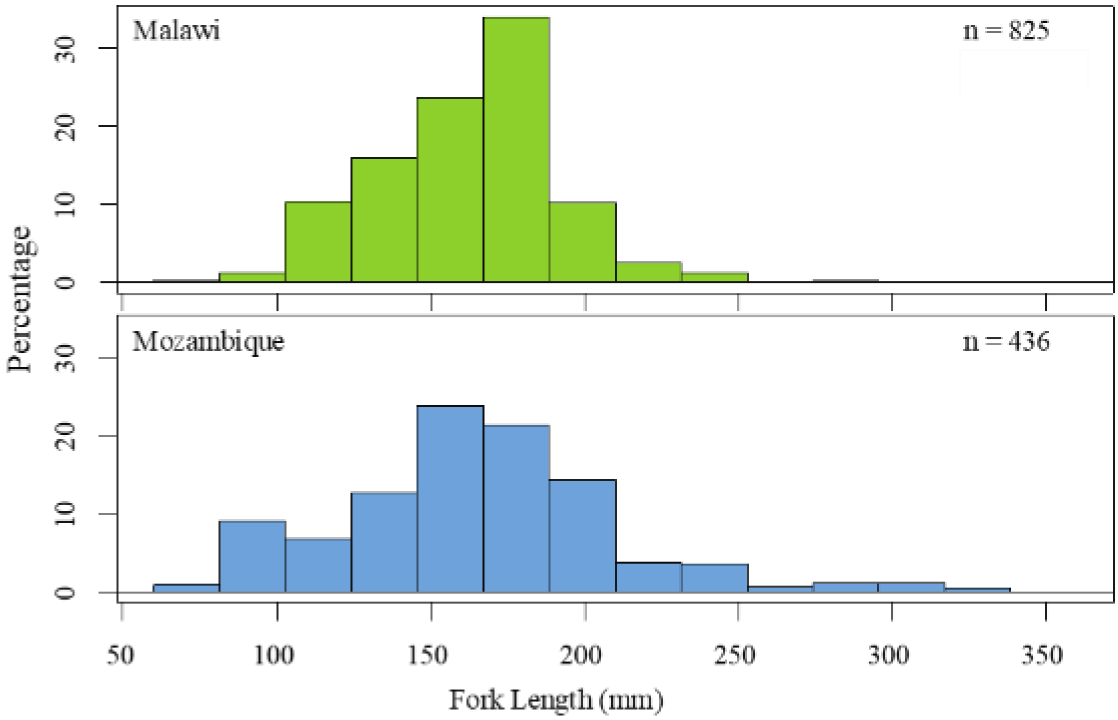


Figure 4.15: Length frequency histograms for Chambo sampled in *Malawi* and *Mozambique*, including the total number of lengths captured (n).

Figure 4.15 presents the length distributions of Chambo by countries *Malawi* and *Mozambique.* The Chambo population of *Malawi* is considered to be exploited at much greater levels than in *Mozambique* (Halafo et al. 2004). A Kolmogorov-Smirnov two-sample test determined that there was a significant difference in the length distribution of Chambo observed between the two countries (D = 0.154, $p < 0.001$) with *Mozambique* having a greater average length of 168. 70 mm vs *Malawi* of 162.82 mm. Figure 4.16 was the only length distribution fit to more than two *levels*. The Figure displays *Cape Maclear*, *Outside National Park* (*Monkey Bay* and *Nkudzi Bay*), *Meluluco* and *Metangula*. The *Outside National Park* Chambo population remains unprotected and being located in Malawi, is exposed to the greatest levels of exploitation. The length distribution of *Cape Maclear* differs significantly from *Outside National Park* (D = 0.641, $p < 0.001$), *Meluluco* differs significantly from *Metangula* (D = 0.186, $p < 0.004$) and *Cape Maclear* differs significantly from *Metangula* (D = 0.244, $p < 0.001$).

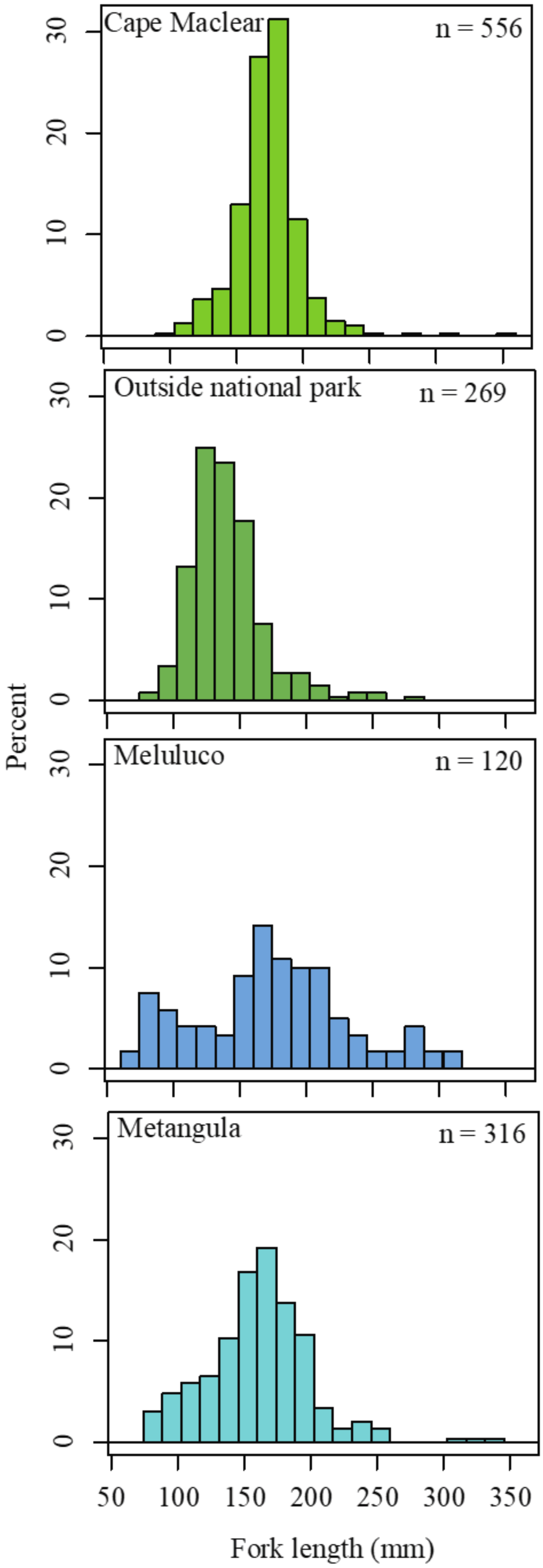


Figure 4.16: Length frequency histograms for Chambo sampled within *Cape Maclear*, *Monkey Bay* (MB) and *Nkudzi Bay* (NKB), *Meluluco* and *Metangula*, including the total number of lengths captured (n).

Figure 4.17 and 4.18 present the size-frequency distributions for Chambo by *Depth* and *Habitat* in *Lake Malawi*, respectively. The D*eep* (> 20 m) size-frequency distribution of Chambo differed significantly for *Shallow* distributions according to a Kolmogorov-Smirnov two-sample tests (D = 0.435, p < 0.001) and *Sand* distributions differed significantly from *Structure* habitat (D = 0.229, p < 0.001) according to run K-S tests.

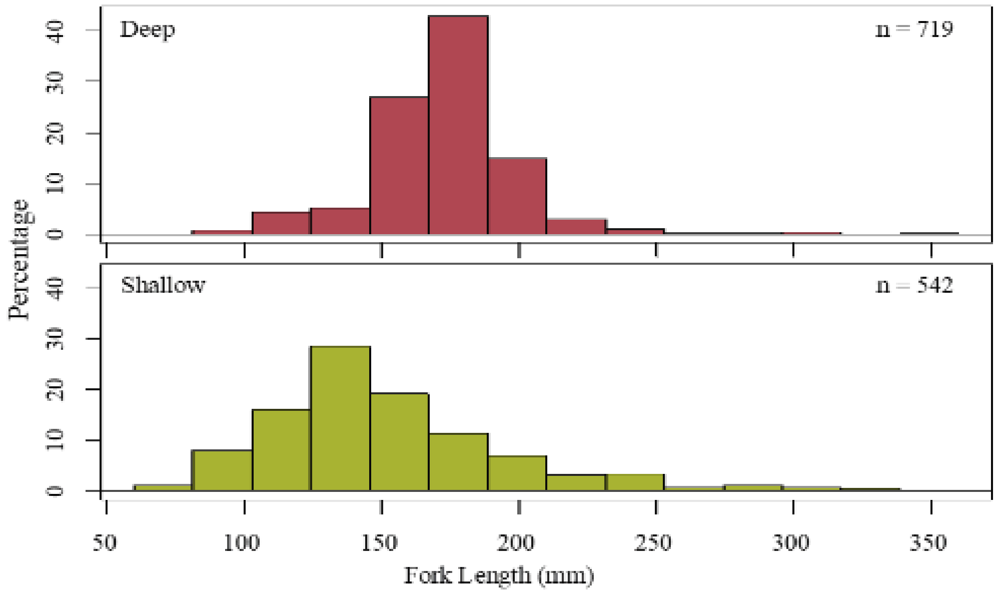


Figure 4.17: Length frequency histograms for Chambo sampled in *Deep* (> 20 m) and *Shallow* (< 20 m) waters of *Lake Malawi*, including the total number of lengths captured (n).

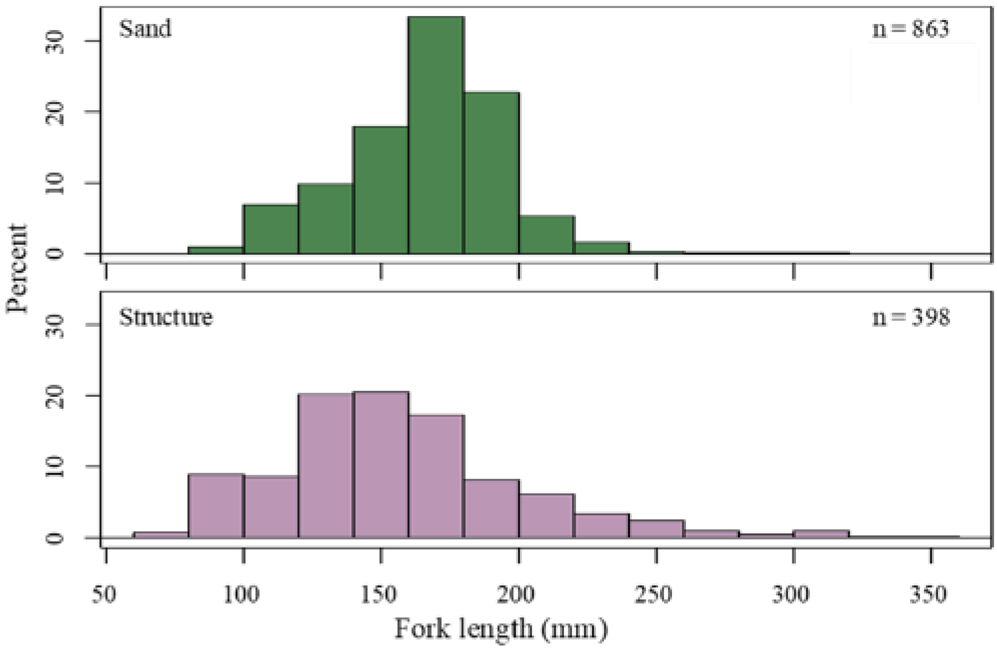


Figure 4.18: Length frequency histograms for Chambo sampled in *Sand* habitat and those habitats where a proportion of *Reef* was present (Structure) in *Lake Malawi*, including the total number of lengths captured (n).

DISCUSSION

The working hypothesis for this chapter was that in those localities where Chambo are exposed to greater fishing pressure, the abundance and detection probability would be lower. Furthermore, the extensively fished populations will vary in size structure as larger, more valued individuals are disproportionately removed from the fisheries. The findings from this chapter – for the most part, support the working hypothesis. Chambo was significantly more abundant at greater depths and was significantly less abundant in Nkudzi Bay – the most heavily exploited study site (Figures 4.7 - 4.9).

Similarly, Chambo had a significantly higher probability of being detected at greater depths and a lower probability in Nkudzi Bay. The species group was also significantly more detectable in Mozambique compared to Malawi (Figure 4.13 & 4.14). Observations of Chambo in Malawi and in the study location, Nkudzi Bay were less frequent in shallow waters. The abundance of Chambo did not, however, align with this hypothesis. Although observed infrequently in Malawi, when Chambo was detected, they were observed in relatively large schools - displaying strong aggregating behaviour.

Chambo in Lake Malawi underwent a significant species-specific depletion of biomass two decades ago (Turner and Mwanyama 1992, Turner 1995, Weyl 2001, Banda et al. 2005b, Bell et al. 2012). In many, once productive areas, such as Salima (13.779° S, 34.459° E) on the Malawian lakeshore, Chambo catches have dropped to only 1 % of total catch as a result of gross overfishing (Bell et al. 2012). The species flock – now classified as Critically Endangered, exists as the most valuable constituent to the fisheries sector and subsequently, is of great socio-economic importance at both a household level as well as to the national economy (Banda et al. 2005a, IUCN 2019). Weyl et al. (2010) discuss the consequence of fisheries growth and development preceding that of robust scientific knowledge upon the exploited species. The resultant effect being the adoption of adaptive management measures (Weyl et al. 2010). One such measure in both Southern Lake Malawi and the Lake Niassa Reserve (LNR) is the proposal and advocacy for closed sanctuary areas and the implementation of monitoring or restocking programs. The simplified aim of this endeavour is to demarcate those areas of great importance to Chambo, close them to fishers and provide both juvenile and mature stocks with a means to recover (Banda et al. 2005a). A key challenge to the undertaking of this proposal is the selection of adequate and realistic protected sites and monitoring potential biomass growth. This study proposed the use of remote video monitoring as a tool to investigate both the applicability of stereo- Baited Remote Underwater Video Systems (BRUVS) and shed light on potential areas of importance for Chambo.

Key to the employment of stereo-BRUVS as a long-term monitoring tool is the determination of standard operating procedures, including optimal deployment times and sample size requirements (Harvey et al. 2013). Prior to this study, no complete operating procedures for the use of stereo-BRUVS in an African Great Lake had been formulated. Determining an adequate deployment length (i.e. video recording) was achieved in Chapter 3 and a 15-minute deployment time was determined to be an adequate recording length and importantly, remained a conservative and manageable length of time to capture a relative abundance estimate (MaxN) for Chambo. Requiring, upon average, 5.9 minutes (SD = 0.5, min = 0.01, max = 19.9) to be recorded in the BRUVS field of view at a lake-wide scale. Similarly, the number of annual samples required to detect a 10 % change in Chambo abundance over a period of 10 years was determined to be manageable in both Malawi and Mozambique. Malawi did, however, require almost double the annual sampling effort (n = 120) compared to Mozambique (n = 56). This sampling effort was as a result of increased variability in the run model for Malawi due to fewer detections recorded and larger variation in school sizes – indicative of potential differences in fishing exploitation rates.

A key objective of this study was to determine how Chambo may be distributed across different habitat types, depths and localities in Lake Malawi and make inference upon critical areas in the lake which may be of importance to the species. The sampling period each year (from September-November) was selected as to avoid the rainy season but also had the benefit of aligning with the onset of the species' breeding season. The distribution of the species flock during these months are believed to be relatively localised, demonstrating limited dispersion across the lake (Palsson et al. 1999b). The modelling of the Chambo data collected by BRUVS provides multiple interesting observations and highlights the complexity of the issue faced by fisheries managers.

The species flock utilises a range of different habitat types. During the species' breeding period, large aggregations of mature individuals occur on expanses of sand as the males begin to construct large, conical-shaped nests (Turner and Mwanyama 1992). Following a successful spawning event, the mouthbrooding females then transport young to shallow nursery habitats (which include grass beds, sheltered sandy bays and rocky shores) where the independent fry develop into juveniles and reach sexual maturity (Lm50) at ± 230 mm (Palsson et al. 1999b). The BRUVS data collected in this study suggests the addition of structured habitat as potential areas of importance for maturing Chambo. The detection probability of Chambo on both reef and sand-reef habitats was significantly higher in all models, whereas sand had little effect (Figures 4.10-4.14). This result was not however, mirrored in the abundance modelling. Less frequent but larger aggregations of Chambo were detected on sand expanses, lacking the somewhat ubiquitous distribution of Chambo observed on structured habitat and may explain this occurrence.

Another notable result was the significant effect of depth upon the abundance of Chambo observed in all models during the sampling months as well as its significant effect upon the detection probability of Chambo at a lake-wide scale. The BRUVS were deployed at a depth range between 0.8 m to 64.9 m. The importance of shallow habitat for Chambo has been discussed in multiple studies and these areas - as previously discussed support juvenile fish as well as large, brooding females (Palsson et al. 1999b, Banda et al. 2005a). Deeper waters may evidently, prove to be equally as important to Chambo - particularly when they undergo depth-specific migrations. Such as aggregating to spawn on constructed sandy nests during critical spawning months from September to December (Palsson et al. 1999b). Demonstrated by the habitat and depth-effected size structuring in figures 4.17 and 4.18. This observation then raises the question as to what areas or habitat types are of importance to the Chambo flock in Lake Malawi? According to previous literature, which suggests that Chambo make use of a variety of habitats during its full life history and taking into consideration the results of this chapter, there is no single habitat type or depth range that should be singled out and reasoned to be independently protected (Turner and Mwanyama 1992, FAO 1993, Banda et al. 2005a). Rather, broader areas containing multiple habitat types and varying depth profiles - whilst being able to encompass depth-specific migrations should be considered.

Perhaps the most telling results, are the apparent and significant differences in Chambo data between both the two studied countries, Malawi and Mozambique and areas of differing fishing pressure. The Malawi Chambo fisheries are considered collapsed and the deep-water stocks heavily exploited (Banda et al. 2005a, Weyl et al. 2010). The devastating crash, although also attributed to a drop in lake water levels, is mainly recognized to be as a result of gross overfishing (Palsson et al. 1999b, Bell et al. 2012). In an attempt to preserve the natural biome of the Malawian lakeshore, the Lake Malawi National Park was gazetted in the 1980s (Bootsma 1992). It is within this National Park that the study area, Cape Maclear is located and contains multiple habitat types required by the different Chambo life-history stages as well as a broad depth profile. On the Mozambique lakeshore, the Lake Niassa Reserve was declared in 2011 in a similar attempt to better manage the county’s natural fisheries resource. Chapter 3 discussed the difference in fishing pressure experienced by the two countries and that fishing limitations in Mozambique such as relatively low population densities, weak market forces and over a decade a civil war meant that the fishing pressure along Mozambique's shoreline remained relatively low (Halafo et al. 2004).

The BRUVS data collected for Mozambique differs significantly from that collected in Malawi. The probability detection probability of Chambo is significantly higher in Mozambique, and subsequently, the required sample size to detect minor changes in abundance is approximately half that of Malawi. Furthermore, there was a difference in the mean length of Chambo observed between the two countries with Mozambique having a higher average length versus that of Malawi. The two length distributions also differed significantly according to the Kolmogorov-Smirnov two-sample test ($D = 0.154$, $p < 0.001$) (Figure 4.15). The apparent lack of individuals larger in size than 230 mm FL in Malawi and a general narrowing of the length distribution are two factors contributing to this observed result and are positively evident in the non-protected areas of Malawi, Monkey Bay and Nkudzi Bay (4.16). This narrowing or truncation of the distribution has been discussed to occur in areas exposed to high fishing pressure due to both the removal of largest individuals through size-selective fishing and the evolution towards smaller size-at-age due to selective harvesting of faster-growing individuals (Audzijonyte et al. 2013). Positively, the length frequency of Chambo analysed for Cape Maclear differs significantly from its non-protected surrounding bays and more individuals over 230 mm FL were detected (Figure 4.16).

An applied use of the BRUVS abundance and length-frequency data may be presented, taking into consideration the study completed by Turner and Mwanyama (1992). The 1997 Malawian Fisheries and Conservation Act put in place both seasonal and area catch limitations for the Malawi Chambo stock - closing the fisheries annually from the 1st of November to 31st of December (Donda 2003). The relatively large aggregations of Chambo measured by BRUVS in deeper waters during the sampling period, September to October and the presence of mature, breeding coloured males in these deeper, sandy area suggests that (A), the spawning period extends beyond the initially proposed closure months or (B), that the species undergoes a pre-spawn staging event (Turner and Mwanyama 1992). Staging events observed in other freshwater aggregating species refers to the aggregation of mature individuals in suitable habitats prior to the building of nests and actual spawning (McCleave 2013). This theory was initially proposed following the post-analysis of stereo-BRUVS video footage and the lack of any sand nests observed in the presence of large aggregations of maturely coloured Chambo. Evidence that staging is occurring shall, however, require greater examination of sand habitat for the presence of both mature individuals and the iconic conical-shaped nests.

Turner and Mwanyama (1992) discussed the size-specific depth preference of the species as well as its extended breeding season and its subsequent implications for the management of the fisheries. If Chambo have a more extended breeding season than initially assumed, the authors state that there is little biological justification for the timing of a closed season. Furthermore, the size-specific depth preference of the species (particularly larger, fecund females) means that the entire flock is at risk of being exploited by all scales of the fishery – artisanal to industrialised throughout their breeding cycle (Turner and Mwanyama 1992). Proposed then, is a closed season based upon social considerations, strict gear regulations and the advocacy for protected areas able to encompass the full range of Chambo habitat.

In Conclusion, BRUVS are demonstrably a single monitoring technique able to be realistically employed for the monitoring of Chambo in Lake Malawi. Being able to effectively identify, count and measure the species within all potential habitat types and depth profiles may be the key to the successful long-term monitoring of the species flock. Addressing the paucity of species-specific data and taking steps towards a more proactive management paradigm. It must be noted, however, that the strength of the BRUVS data is underpinned by the inclusion of more study sites, repeated sampling and the standardisation of all operating procedures. This study has taken the first steps towards understanding the use or appropriateness of employing BRUVS for *in-situ* monitoring of fish species in Lake Malawi. Although objectively efficacious, further studies would benefit from the inclusion of more study sites and greater sampling effort at a wider depth profile.

IUCN workshop participants where Key Biodiversity Areas in the Lake Niassa Reserve were proposed using results from BRUVS surveys (Top) and scenes from a presentation in the Meluluco fishing village about "where their local underwater life and our research meet" (Bottom). P.C. Angus van Wyk (SAIAB)

"Living wild species are like a library of books still unread. Our heedless destruction of them is akin to burning the library without ever having read its books."

John Dingell

This single quote taken from Balancing on the Brink of Extinction, authored by John Dingell, may best describe what this thesis aimed to address (Dingell 1991). Human actions, driving system-wide change, in freshwater systems we are yet to fully understand or have the chance to appreciate. One such measure that may offer sanctuary to species within the biodiverse library of Lake Malawi, are protected areas and apt long-term monitoring tools – such as Baited remote underwater video systems (BRUVS). The applicability of BRUVS in freshwater, although discussed to be relatively cost-effective and efficient in the marine realm, are not well understood in African freshwaters (Bernard 2012). The results of this study suggest, however, that BRUVS are a feasible long-term monitoring technology for Lake Malawi.

Eight selected species groups could be identified, and their respective optimal deployment times were less than 15-minutes. Similarly, the number of annual deployments required to detect a minor change in the species' groups abundance also remained feasible. Interestingly, most of the species groups required more significant sampling effort in Malawi compared to Mozambique – particularly those species groups targeted by local fisheries. The effect of differing fishing pressures between the two studied countries was also observed in the Chambo abundance data. The high-value species group was less abundant in shallow water habitats and within the most heavily exploited study site, Nkudzi Bay. The probability of detecting Chambo was also significantly lower in shallow waters, Nkudzi Bay, in Malawi and upon sand habitat. The size-structure of Chambo between the two countries also differed significantly. Malawi, and particularly those study locations outside of the Malawi National Park having size structures typical of fish populations exposed to overexploitation.

The synthesis of key thesis findings, in relation to the proposed aims of each data chapter, is presented in Figure 5.1. The overarching aim of this thesis was to investigate the potential to employ a fisheries independent monitoring technique in the form of BRUVS in Lake Malawi. The initial step towards this aim demanded a systematic review of current BRUVS employment (Chapter 2). This review was undertaken with the intention to better understand how BRUVS were being employed on a global scale and determine their current applications in African freshwater systems. There is currently no published research (within the scope of the systematic review) whereby BRUVS are employed in any capacity in African freshwater systems. Furthermore, there is an evident dominance of marine BRUVS applications, and only 1 % of studies were performed in the freshwater realm. This is interesting, given the recommendations by marine researchers who advocate for the employment of BRUVS as a non-extractive monitoring tool and highlight their potential benefits over traditional techniques – including cost (Cappo et al. 2001, Bernard 2012, Bernard et al. 2014, Cundy et al. 2017, Logan et al. 2017, Roberson et al. 2017).

Many freshwater bodies, particularly African Great Lakes, should provide ideal systems for BRUVS employment. Most of the Great Lakes are characterised by clear waters, low currents, manageable operating depths and a perceived need to monitor fish biodiversity. Nevertheless, BRUVS have not being used to any significant effect for research or monitoring prior to this dataset. Perhaps the logistical and technological costs required to initially employ BRUVS presents too significant an investment for institutes in developing nations - 83 % of studies employing BRUVS were published by developed nations. Another barrier for the adoption of

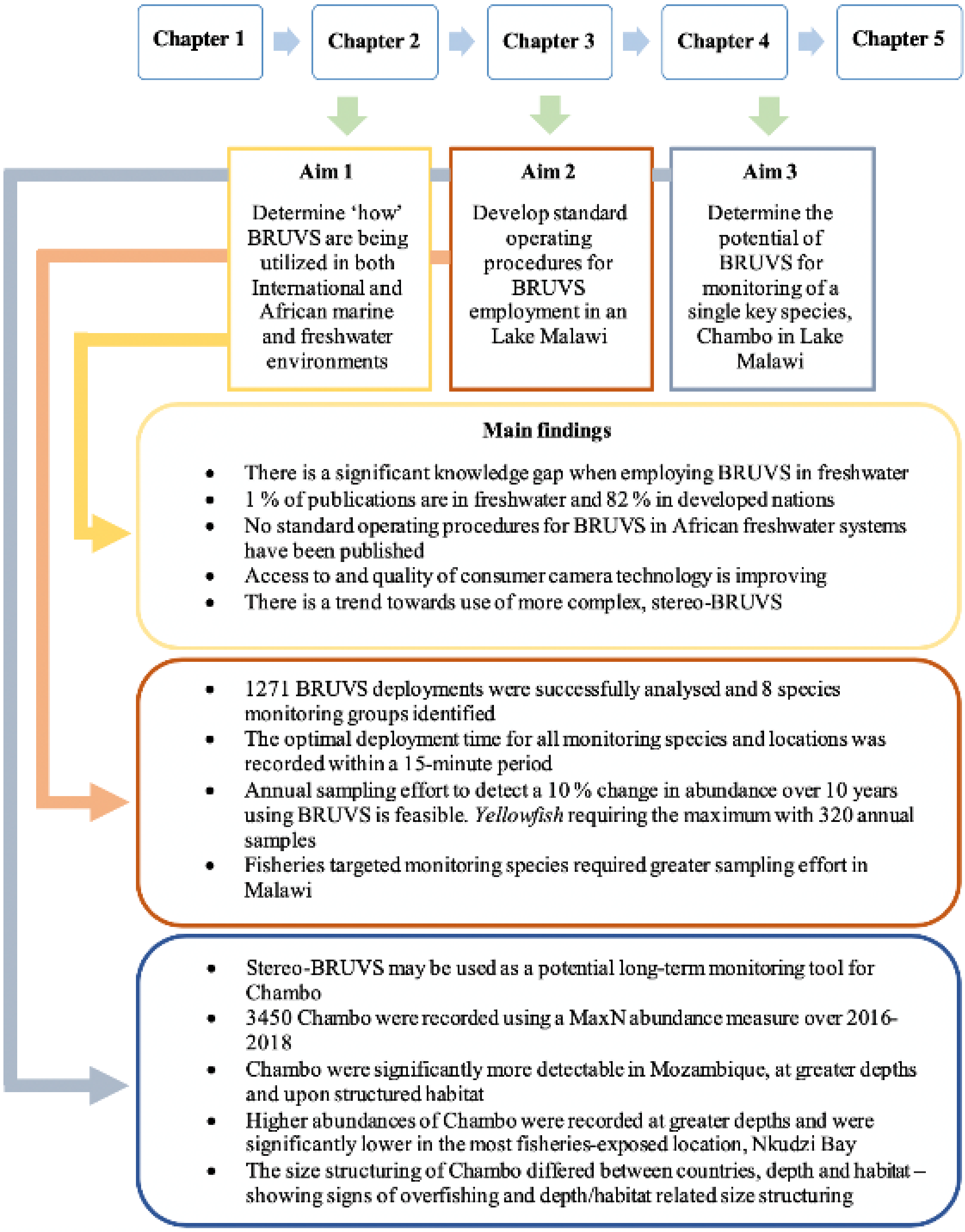


Figure 5.1: Synthesis of thesis finding in relation to proposed aims and chapter outlines.

BRUVS in African freshwater bodies may be the lack of published standard operating procedures (SOPs) and guidelines. Understanding how to employ BRUVS – regarding both their limitations and benefits may prove to be important for the initial adoption of the technology and its inclusion into ecological research as a beneficial monitoring tool.

Chapter 3 contributes to the development of standard operating procedures for BRUVS employment in Lake Malawi. This was achieved by identifying key monitoring species, determining an optimal deployment time and lastly, to determine if BRUVS could be logistically feasible if employed in a long-term monitoring program by calculating sampling effort requirements. Identifying species and calculating species richness using BRUVS is challenging – particularly in Lake Malawi, where researchers believe the number of actual species maybe double that what is currently described (Turner et al. 2001). A similar challenge was faced by Joo et al. (2013) when the researchers attempted to identify Lake Malawi cichlid species using computer vision. Even with high-quality imagery and post-image analysis, the researchers could not conclusively identify all species because of the apparent phenotypic similarity between genetically distinct species (Joo et al. 2013). For that reason, key monitoring species were selected that could be identified using camera footage only and were representative of the Lake Malawi shoreline fish communities. Eight species groups could be identified and were arranged by their respective local fisheries management groupings. These were namely: Chambo (Oreochromis spp.), Kampango (*Bagrus meridionalis*), Labeo (*Labeo cylindricus*), *Metriaclima estherae*, Mlamba (Bathyclarias & Clarias genera), Predatory haplochromines (Dimidiochromis, Tyrannochromis & Rhamphochromis genera), Sleepers (Nimbochromis spp.) and Yellowfish (*Labeobarbus johnstonii*).

The optimal deployment time for the selected monitored species groups fell within the 15-minute recording period used throughout the study. Malawi requiring 5.7 minutes and Mozambique, 8 minutes (Table 3.5). In an attempt to better understand the optimal deployment time for each monitored species group, an average time to record a MaxN abundance estimate was calculated. Similarly, neither species group had an average time over 15-minutes (Table 3.6). The relatively short optimal deployment time required for the monitored species groups improves the potential for BRUVS employment in Lake Malawi. With a shorter deployment time, fewer resources and time may be dedicated to video analysis, backup and storage per single sample collected. Furthermore, days sampling on the lake and the costs endured when using a vessel may be reduced.

In Chapter 3, I also used a power analysis to determine the amount of sampling effort required to detect a 10 % change in species abundance over a 10-year period and for each species group. Similarly to shorter optimal deployment times being logistically beneficial, so too are fewer required deployments. A lower annual sampling requirement to detect changes in species abundance may reduce the operating costs of employing BRUVS and reduce the number of boat days in a protected area. There was a relationship between the variability within the abundance data of each species group and the sampling effort required (Table 3.7 & Figures 3.6-3.13). Those species groups which were detected less frequently and in varying abundances lead to greater variability in the model and subsequently, required a greater amount of sampling effort to achieve an 80 % power.

Yellowfish, for this reason, resulted in the species group requiring the most significant amount of sampling effort (Figure 3.12). The species group was rarely detected, and if observed, large schools were encountered. Subsequently, the species group required a sample size of 320 annual BRUVS deployments to detect a minor change in abundance over a 10-year period. Those species groups Chambo, Kampango, Mlamba and Predatory haplochromines which are directly targeted by various fisheries required more than 50 % greater sampling effort in Malawi in comparison to Mozambique. Lower abundances or less frequent detections of these species groups in Malawi – resulting in greater sample size requirements may be as a result of different levels of fishing pressure between the two countries (Halafo et al. 2004, Weyl et al. 2010).

In Chapter 4, BRUVS were employed to investigate the relative abundance and size structuring of Chambo in Lake Malawi. Analyses demonstrated that BRUVS data supported findings of current fisheries generated catch data (Halafo et al. 2004, Weyl et al. 2010). That is, Malawi and particularly the South Eastern Arm (SEA), exploits Chambo stocks at a higher level compared to the Mozambique shoreline. Consequently, the SEA Chambo stock collapsed in the 1980s (Palsson et al. 1999b, Weyl 2001, Banda et al. 2005b, Bulirani 2005, Bell et al. 2012). The significant amount of fishing effort in the SEA fishery responsible for driving the collapse was not, however, experienced in Mozambique (Halafo et al. 2004). This phenomenon generated the working hypothesis for Chapter 4. Malawi, which experiences significantly higher levels of Chambo exploitation, would result in lower detections using BRUVS and size structures likened to exploited fish species. Chambo were significantly less likely to be detected by BRUVS in Malawi as well as Nkudzi Bay, the most exploited study site (Figures 4.12-4.14). Chambo were also significantly less abundant in shallow, easily accessible waters and within Nkudzi Bay (Figure 4.7-4.9). Interestingly, habitat type also played an important role in the detection probability of Chambo as those habitats with a portion of reef present resulted in a significantly higher detection probability (Figures 4.13 & 4.14).

The length data I collected using stereo-BRUVS during the year 2018 also supported this working hypothesis. The size structure of Chambo in Lake Malawi differed significantly by Country and Location (Figures 4.15 & 4.16). The length-frequency distribution of Chambo in Mozambique was less truncated than that of in Malawi, and less truncated in those locations within the protected Malawi National Park compared to outside the park. Both cases display a greater contribution to the size structure by those individuals over 230 mm – the species' length of first maturity (Lm50) (Palsson et al. 1999b). The effects of overfishing may not only affect the number of fishes observed during a monitoring program but also the species' size structure. Overexploitation often results in the removal of larger, fitter and more fecund individuals from the population and selection pressure for the species to mature at a younger and smaller size class (Beamish et al. 2006). This can have a detrimental effect upon future recruitment rates of Chambo and consequently, the fishery's future.

Considering the generated results and discussion from this thesis, BRUVS may be used as a long-term monitoring tool in Lake Malawi. The technology behind the construction and employment of BRUVS is more accessible and efficient than it has ever been, the length of recording time and sampling effort required are objectively feasible, and the systems can generate useful multi-species modelling data – all while remaining independent of fisheries and extracting no live specimens. This research is, however, only the first step towards optimising BRUVS for employment in African freshwater bodies, and there is still much work to be done. Future research will benefit from further developing the current Lake Malawi BRUVS dataset of 1271 deployments.

Bettering the dataset should involve the inclusion of more study sites across the Lake as well as including habitats at a greater depth range. The stereo-BRUVS employed in 2018 are rated to be operational at depths of 300 m. The BRUVS may also be configured to be suspended in the water column to monitor pelagic species. Increasing the time series and the number of monitored species groups for BRUVS data within current study sites would prove to be highly beneficial. Particularly in the Malawi study sites which were only included in the dataset in the year 2018. Although including new data into existing BRUVS dataset may advance the technology, so too will the continued analysis of the current 318 hours of captured video footage. The existing BRUVS dataset may be used to answer many more relevant ecological questions and perhaps shed light on the wealth of biodiversity in Lake Malawi. In conclusion, there are many more critical areas in Lake Malawi and upon the African continent that could serve as useful study locations and ultimately benefit from the employment of BRUVS – bolstering local management capacity and attempting to avoid the "burning" of Africa's incredible freshwater species library.

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APPENDIX

Supplementary data link: **https://drive.google.com/file/d/1vWjCB1PkBGMD-r8aB1h5YlkD087KkLo1/view?usp=sharing**