

**DEVELOPMENT OF TILAPIA CAGE CULTURE ON LAKE VICTORIA,
UGANDA**

**A THESIS SUBMITTED TO THE UNIVERSITY OF STIRLING FOR THE
DEGREE OF DOCTOR OF PHILOSOPHY**

BY

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DECLARATION

I declare that this thesis contains original research work made by myself and that where works of others have been used, acknowledgment by reference has been made. This research has not been submitted for any other degree.

Faridah Bukirwa

DEDICATION

To my children, Waldah, Roshan and Rayaana, may you be inspired. Thank you for weathering the storms with me.

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ABSTRACT

Lake Victoria is the largest freshwater lake in Africa with significant social, ecological, and economic importance. In the face of declining fisheries, Tilapia cage culture was sought to bridge the demand gap and create employment opportunities. However, in a eutrophic lake with a large population dependent on it, fish farming should be supported by evidence-based decision-making. While used in systems elsewhere, existing tools will need adaption to address freshwater, context-specific challenges. This research aims to contribute towards evidence-based decision support for the planning and management of Tilapia cage culture in Lake Victoria.

To set the context, farm production practices, existing regulatory frameworks and stakeholders' views were explored. Spatial planning and prioritization tools for efficient and equitable resource use were investigated. Considering the complex dynamics in the lake ecosystem, monitoring techniques for two objectives; rapid farm impact assessment or on-farm licensing were evaluated. The research investigates different aspects of cage farming in Lake Victoria. However, the findings will be applicable to several other tropical lakes.

Results show that Tilapia cage culture is growing rapidly, dominated by large farms. Use of trash feed among smallholders poses a significant environmental risk and hence, strategies targeting them will be required. Using the Q methodology, stakeholders' views on the growth of cage farming on Lake Victoria revealed optimism for economic growth but concerns around institutional gaps and aquaculture negative externalities. A review of the regulatory framework revealed EIA as a major management tool, but in its guidelines, cumulative organic loading from cages was overlooked. To contribute to protocols for environmental management, benthic macroinvertebrate assessment coupled with nutrient balance modelling were identified as robust tools for on-farm licensing of cage farming in eutrophic freshwater lakes. Other techniques like regression statistical modelling generated simple models that farmers could use to rapidly assess environmental performance. Site hydrodynamics were key in model performance.

Using spatial planning, only 9.7% and 3% of the study area was found moderately to highly suitable for Tilapia farming in medium-large cages and small cages respectively. Current production is concentrated in eutrophic, shallow enclosed bays, in which other ecosystem services happen as well. In a spatial optimization scenario of cage culture growth, there were minimal trade-offs between ecosystem services that could occur in deeper waters. By contrast, the greatest compromises were made between recreation

services and fishing nurseries. As technology advances, deeper parts of the lake will need exploration for aquaculture expansion.

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CHAPTER 1

GENERAL INTRODUCTION

1.1 Background

Global aquaculture production is growing and substantially contributes to aquatic food supply and nutrition security (Zhang *et al.*, 2022). However, the contribution of sub-Saharan Africa to global volumes is still less than 1% (Ragasa *et al.*, 2022), despite the immense water resources and relatively cheap labour available in the region. Increasingly, aquaculture production is relying on freshwater systems (Belton *et al.*, 2020), generating 54.4 million tons of fish in 2020 (FAO, 2022). With this growth comes expansion and intensification which pose socio-economic, nutritional, and environmental risks (Bush *et al.*, 2019; Heilpern *et al.*, 2021). It is these negative externalities that undermine the remarkable contribution aquaculture makes within global food systems (Naylor *et al.*, 2021). Examples of such negative externalities of aquaculture growth include organic enrichment; adverse impacts on livelihoods; and incompatibility with other Ecosystem Services (ES) (Aguilar-Manjarrez *et al.*, 2017). Although some of the adverse consequences of aquaculture development are being addressed by researchers, government, private and public sector players and donor organizations, there remains a long way to go.

The Food and Agriculture Organisation of the United Nations (FAO) recommended an Ecosystem Approach to Aquaculture (EAA) (FAO, 2010) to ensure sustainable sector development. Central to the EAA is a recognition of the need for wider frameworks for planning, development and management while considering aquaculture's effects on other sectors and the ecosystem (FAO, 2018). The EAA brings together a common framework for stakeholder involvement in aquaculture planning: the different uses of the ecosystem (social, ecological, environmental and a consideration of trade-offs); the use of scientific research in decision making; formal management plans for specific developed areas; adaptive management and institutional development (FAO, 2018). As presented in Ross, *et al.* (2013), the three principles of an ecosystem approach to aquaculture are:

- Principle 1: *Aquaculture development and management should take account of the full range of ecosystems functions and services and should not threaten the sustained delivery of these to society.*
- Principle 2: *Aquaculture should improve human well-being and equity for all relevant stakeholders.*
- Principle 3: *Aquaculture should be developed in the context of other sectors, policies, and goals.*

As a result of relying solely on smallholder pond farms, aquaculture production in sub-Saharan Africa had stagnated for decades (Isyagi, 2017; Odende *et al.*, 2022). The remarkable growth in the recent past has been attributed to private sector investment, with most farming done in inland cage systems (Ragasa *et al.*, 2022). In the East African region, Lake Victoria provides an opportunity for increasing fish production and indirectly reducing the pressure on open access fish stocks that are on the decline (LVFO, 2016). Shared among Uganda, Kenya and Tanzania, Lake Victoria is one of the largest African Great Lake (AGL) systems with high social, environmental, political, and economic importance within the region (UNEP, 2006). Directly, the lake supports a population of over 42 million people (Awange, 2020) while indirectly, millions of people living in the five lake basin countries including Rwanda, and Burundi, depend on it. Lake Victoria supports fishing, tourism, navigation, water abstraction for domestic and agricultural use (El-Noshokaty, 2017). Other uses include flood control, sediment trapping, filtration of raw or partially treated municipal and industrial waste and climate moderation. To explore the potential to contribute to food security and employment, the three riparian countries of Uganda, Kenya and Tanzania have earmarked it for commercial Nile Tilapia (*Oreochromis niloticus*) cage farming.

Developing aquaculture in common pool resources like lakes (Ostrom, 2008) that support a large population is more prone to dispute due to the implications of the negative socio-economic and ecological externalities involved (Keys *et al.*, 2019; Asche *et al.*, 2022). Progressively, these challenges become more critical as population growth, rural to urban migrations and uncertainty in agriculture production driven by climate change (Thornton *et al.*, 2019) force people towards exploiting resources on the lake as a source of their livelihood. Hence, while cage culture on Lake Victoria is promoted as a policy option, attention should be drawn towards the consequences of commodity widening (expansion) and deepening (intensification) (Saguin, 2016), where private enterprises take advantage of the provisions of a capitalist economy to maximise profits

out of a common good. Consequently, this could foster displacement of livelihoods, and irreversible damage to the ecosystem.

Extensive research efforts have focused on mariculture of high value species compared to freshwater aquaculture (Naylor *et al.*, 2021). Organic enrichment was reported to drive the prohibition of cage culture in freshwater systems in China (Naylor *et al.*, 2021), and its restriction on the Damietta and Rosetta branches of the Nile River in Egypt (El-Sayed, 2006). New evidence needs to be generated to fill gaps in knowledge in the Lake Victoria and similar AGL systems. Understanding is needed of how developments can contribute to the livelihoods of the population without compromising existing uses, or the environment. If not properly planned and regulated, the reduction in space available for other lake users could be a source of conflicts (Musinguzi *et al.*, 2019).

Several decision support frameworks that could be leveraged for planning and management exist but need to be tailored to the context of the Lake Victoria system. Furthermore, a misunderstanding of the spatial extent suitable for cage farming and an assumption that by virtue of having the lake, there is potential for fish culture is a misnomer. Poor cage installation including placing cages in shallow waters reduces resource-use efficiency (Musinguzi *et al.*, 2019; Ragasa *et al.*, 2022) in addition to increasing vulnerability to the impacts of organic enrichment. Njiru *et al.* (2018) found that on the Kenyan side of the lake, over 45% of the cages were within 200m of the shoreline. In Uganda, spaces within 200m of the shoreline are restricted as lake buffer zones and protected from encroachment.

Stakeholder engagement is crucial for aquaculture development (Stankus, 2021). Perceived benefits play a role in determining acceptance of aquaculture (Hishamunda *et al.*, 2014). Understanding stakeholder perceptions towards aquaculture aids identification of key concerns and potential conflict areas (Kluger *et al.* 2019) allowing for early mitigation to lower negative impacts and enhance social acceptability (Brugere, Troell and Eriksson, 2021). In the earlier stages of aquaculture development, stakeholder engagements provide an opportunity to generate evidence on what is crucial. In addition, they provide an opportunity to integrate their knowledge, expertise, and judgment into planning. As Lake Victoria serves as a major source of livelihood to a wide population, holistically planning for lake use is a means of ensuring that equity is upheld. This is a means of covering the third principle of the EAA.

Of the major externalities of fed cage farming is the addition of organic waste into the water column (Beveridge, 2008; Gondwe, *et al.*, 2011; Karikari *et al.*, 2020), accelerating

eutrophication in freshwater bodies (Wang *et al.*, 2019; Taskov *et al.*, 2021). For a lake that is already heavily polluted from catchment runoff and atmospheric deposition (Scheren, *et al.*, 2000), licensing to ensure farms stay within the site assimilative capacity (Weitzman and Filgueira, 2020) is needed. It is vital to monitor waste produced from cages using robust protocols to set production limits in licensing.

Considerably, national governments within sub-Saharan Africa will need to re-orient themselves to adopt or revise their legislation to incorporate approaches for aquaculture that ensure equity, efficiency, and environmental integrity. Efforts should focus on sustainable exploitation for improved nutrition, incomes, and livelihoods. This calls for utilisation of evidence-based decision support tools for spatial planning, resource use prioritisation, and impact monitoring to inform development and management of cage production on the lake. In Uganda, significant progress is being made towards upgrading the legislation. However, the absence of evidence-based research in policy making (Goldman and Pabari, 2020) will only limit the contribution of such efforts towards sustainable use of the lake. Research needs to venture into generating new robust tools specific to the lake, or reexplore already existing ones, adapting them to meet objectives in novel contexts.

1.2 Study area

1.2.1 Geographical context

The total surface area of Lake Victoria is 68,800km², making it the 2nd largest freshwater lake in the world and the largest in Africa (Awange, 2006). Formed as a shallow depression between the Eastern and Western arms of the Great East African Rift Valley (Kendall, 1969; Nyamweya *et al.*, 2020), the lake has been referred to as a small sea (Kendall, 1969) shared among Uganda (45%), Kenya (6%) and Tanzania (49%). The Lake is characterised by shallow bays, gulfs, and archipelagos (pockets of islands) mostly in the northern part, an average depth of 40m and a maximum of 80m, and a surface elevation of 11354m above sea level (Awange, 2021). With a long residence time of approximately 23 years (Deirmendjian *et al.*, 2021), the lake is the source of the river Nile, the longest river in the world, and the lake's single outlet controlled by the Nalubaale dam (Beverly *et al.*, 2020) in Jinja, Uganda. Catchment runoff and precipitation account for 80% of inflows to the lake (Khaki and Awange, 2019) while the remaining 20% is from rivers; Kagera, Katonga, Simiyu, Gurumeti, Mara, Gucha,

Nyando and Nzoia (Beverly *et al.*, 2020). With its wide surface area, 80% of the lake outflow is accounted for by evapotranspiration (Deirmendjian *et al.*, 2021) while the river Nile accounts for the remaining 20%.

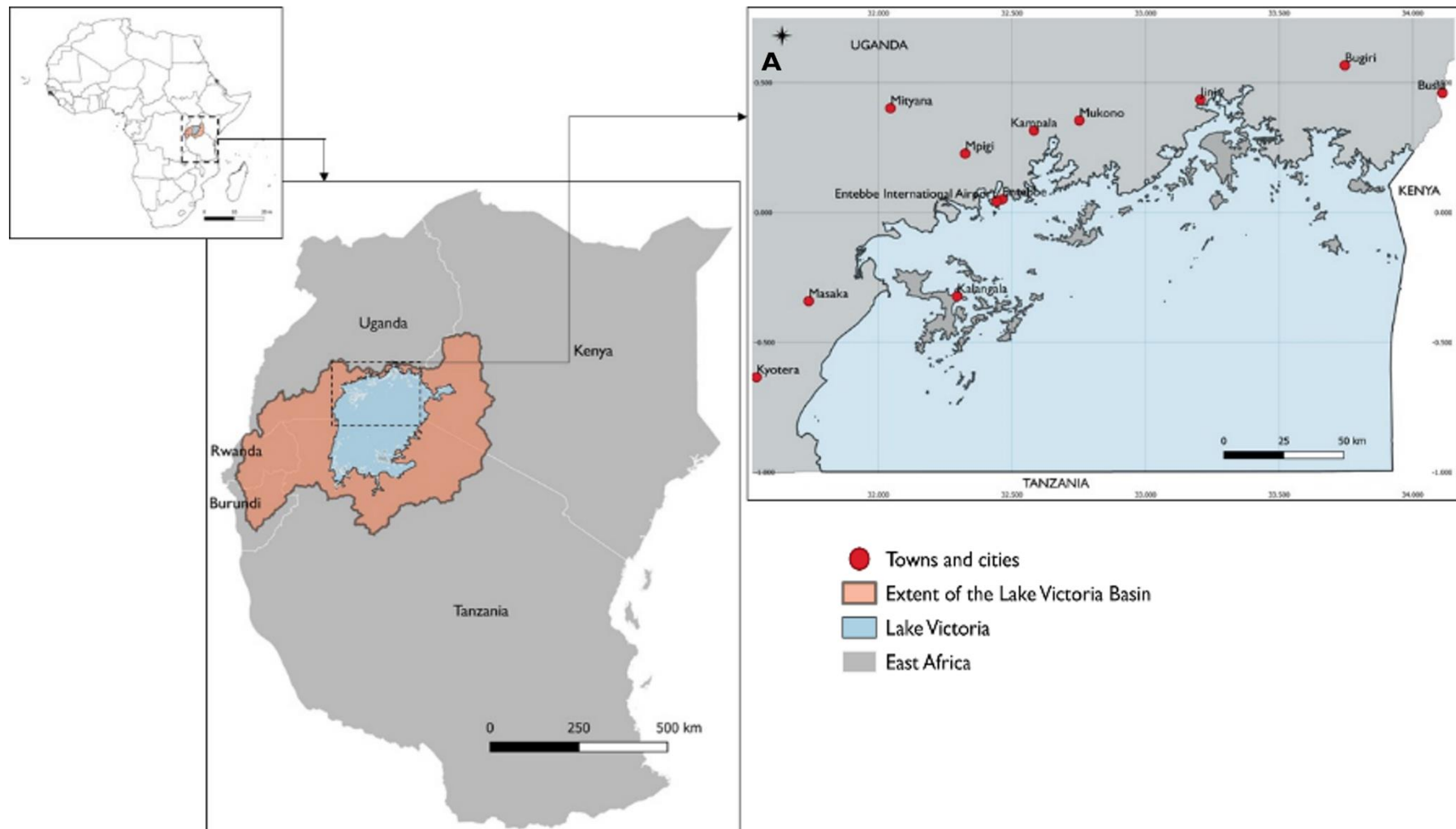
Lake Victoria has substantial influence over regional climate conditions, controlling fluxes of energy, and the hydrological cycle (Thiery *et al.*, 2010). Major influencing factors are the intercontinental Tropical Convergence Zone (ITCZ), Indian ocean South-East trade winds, among other thermodynamic conditions determining diel, seasonal, and annual energy distributions (Nyamweya *et al.*, 2016) to cause rainfall formation over the catchment. Short rains are experienced in the months of September, October and November, while the longer rainy season occurs in March, April and May (Awange *et al.*, 2013, Nyamweya *et al.*, 2016). Precipitation over the lake surface and catchment inflows are the major inflow sources into the lake contributing (80%) and 20% respectively (Awange *et al.*, 2008).

Lake water levels have been reported to fluctuate over the years. So far, historic highs were recorded in the 1990's and 1997 following periods of El Nino rains. Similar trends were reported by Okungu *et al.*, (2005) who stated that low water levels in 1994, 1997, 2005 and 2007 and the highest for the period captured in late 1997 to early 1998 were following El Niño rains. These peaked at 11.52 cm above mean sea level, reported between April and June 1998 (Okungu *et al.*, 2005). Increased rainfall patterns in 2010-2011 (Awange *et al.*, 2013) could be responsible for the increase in lake water levels seen in the period 2011 onwards. The remarkable decline in water levels in 2001-2005 was attributed to a changing climate characterized by La Niña droughts of late 1998 and 2000 (Awange *et al.*, 2013) and changes in the water release policy for hydropower generation at the river Nile. The later led to more water being released from the lake, while rainfall patterns weren't compensating for the outflows. Commissioned on the Ugandan part of the lake in 1954, the Owen falls dam had always been operated on an Agreed curve (Okungu *et al.* 2005; Vanderkelen *et al.*, 2018) but following the commissioning of a second outlet at the Kiira, parallel to the first dam (Swenson and Wahr 2009; Vanderkelen *et al.*, 2018), releasing twice as much is reported to partially have contributed to a decline in lake water levels. These hydrodynamic, morphometric characteristics and the water budget are important for planning and management of cage farming because of the direct influence they have on water quality and water levels i.e., influencing cage locations and waste dispersal.

The focus of the study was the political boundary of the Ugandan sector of Lake Victoria (Figure 1.1) lying between longitude 31° 60'E and latitude 1° 0'S, stretching a surface area of 20,837km² with a maximum water depth of 79m. This is the 2nd largest part of lake after Tanzania. The catchment area covers two more countries; Rwanda (11.4%) and Burundi (7.2%) with a total expanse of 193,000km² of which 44% is in Tanzania, 15.9% in Uganda and 21.5% is in Kenya (Awange, 2006).

1.2.2 Fish production and socio-economic situation in the region

Fish production significantly contributes to the national Gross Domestic Product (GDP) of Uganda, Kenya, and Tanzania by approximately 2-3% (Kimani *et al.*, 2018). There are three known commercial species of Lake Victoria: Nile Perch (*Lates niloticus*), Tilapia, and Silver Cyprinid *Daaga* (*Rastrineobola argentea*) (Kashindye, 2016). Until recently, the fisheries sector was the only contributor to the GDP, with aquaculture done mostly at substance scale in ponds. However, unsustainable exploitation led to a decline in wild stocks (Njiru *et al.*, 2018). Capture fisheries began to decline in about 2014, while aquaculture grew steadily from 2005 through 2018 (Figure 1.2). Aquaculture production in 2019 was reported at 110,000 tonnes, with Nile Tilapia making up 70% of the catch and catfish making up 30% (FAO, 2020b).



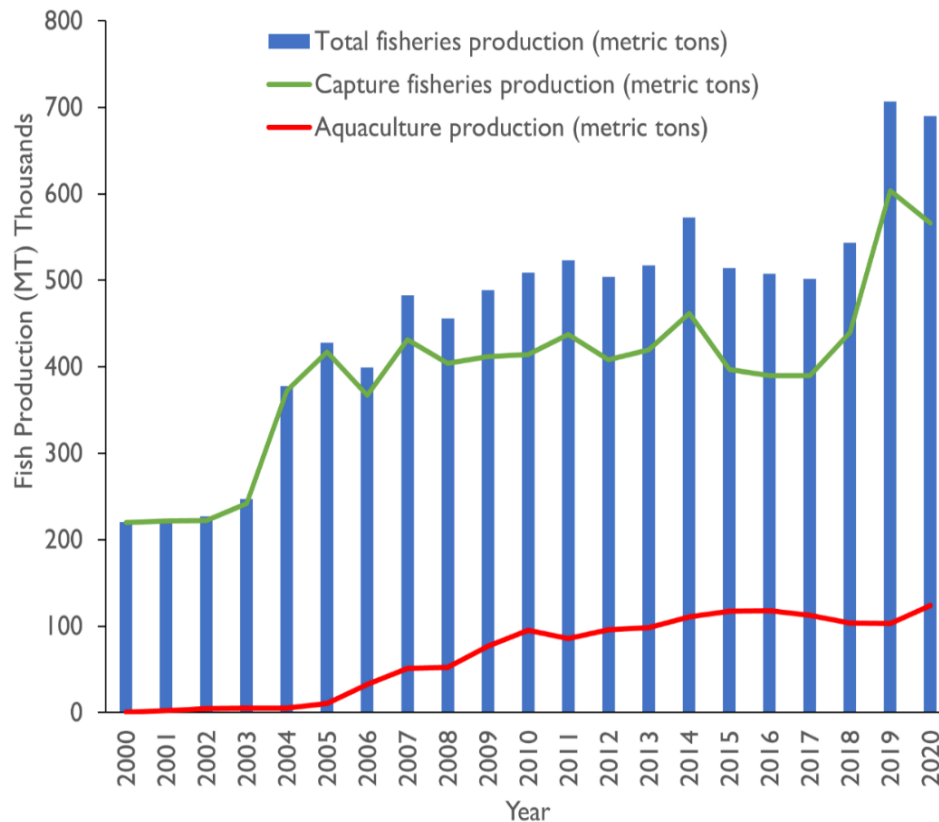


Figure 1.2. Total fish production with capture and aquaculture in Uganda for the period 2000 to 2020 (Data source: World Development Indicators; (World Bank, 2022) last updated 16/09/2022.

O. niloticus species were introduced into Lake Victoria between the 1950s and 1960s (Bwanika *et al.*, 2006; Hecky *et al.*, 2010) outcompeting the two endemic species, *O. esculentus* (Graham) and *O. variabilis* (Boulenger) (Njiru *et al.*, 2006). This ability to outcompete other fish species was attributed to several factors including ability to survive in poorly oxygenated waters; an increase in water hyacinth that provided refuge for feeding and breeding; and an increase in fishing pressure on the predatorous Nile perch (*Lates niloticus*) (Njiru *et al.*, 2008). The lake shoreline vegetation acted as their nursing grounds (Balirwa, 1998). However, due to increased demand and degradation of the wetlands, wild stocks kept declining. This created a large gap between the demand and supply and because tilapia has a high preference in the region as food fish (Njiru *et al.*, 2008), it made it the most suitable species for cage farming.

Meanwhile, most recent estimates put the East African region at 30.6% and 27.3% of the 2015-2017 undernourished and severely food insecure respectively (FAO, 2020a). The same statistics put Uganda's per capita fish protein intake in 2013 at

3.7g/capita/day, which was above the sub-Saharan Africa fish protein intake of 2.8g/capita/day (FAO, 2020a). Urban population and GDP of Uganda have steadily increased from the 1990s to 2020 (Figure 1.3) with a drop in GDP in 2016, attributed to the global credit crisis in the same year. Kuteesa *et al.*, 2009 attribute the growth in GDP in the 1990's to the security and stabilization that allowed businesses to thrive.

Urban population growth rate for the past ten years since 2012 was estimated at an average of 6% (World Bank, 2022) (Figure 1.3). Traditionally, urban dwellers are net consumers of most agricultural produce (including fish), an opportunity for cage farmers (Obiero *et al.*, 2019). Referred to by OECD, (2016), as megatrends, the increase in urban population coupled with access to information and communication technologies, and changes in food consumption trends could lead to increased demand for fish as it is a better protein source compared to other meats (Balami, Sharma and Karn, 2019). This will continue to grow the market for aquaculture products. To contribute to food and nutrition security and poverty eradication, both the fisheries and aquaculture of Lake Victoria need equal consideration.

Amidst these developments, OECD, (2016) highlight that a lack of knowledge, evidence-based policies, weak scientific research, and weak and uncoordinated institutions as major reasons as to why the fisheries and aquaculture sectors have limited contribution to the GDPs of most sub-Saharan Africa countries. Aquaculture must strive to transform the status quo if it is to surpass its current contribution of less than 1% to global aquaculture production (Ragasa *et al.*, 2022). Effective controls and proper planning, governance and management must be established to ensure both sectors are sustainable.

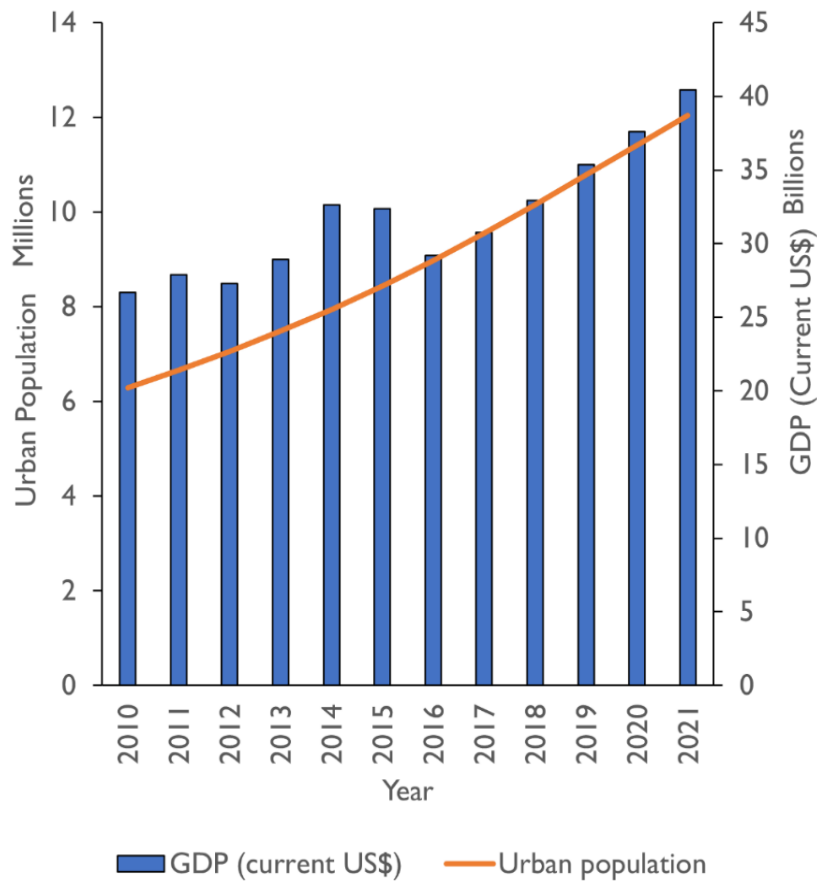


Figure 1.3: Urban population and GDP per capita (current US\$) for Uganda between 2010 and 2021 (Data source: World Development Indicators (World Bank, 2022), last updated 16/09/2022).

1.3 Research Aims

This research focuses on the utilisation of the concept of an ecosystem approach to inform cage aquaculture growth on Lake Victoria, Uganda. Decision support tools for spatial and ecological modelling to inform better planning and management of Tilapia cage farming on the lake are explored. The goal is to contribute to evidence-based planning for the development of cage culture on the Ugandan side of Lake Victoria. Specifically, the aims of the research are;

- To establish the status of cage culture on Lake Victoria, Uganda
- To establish stakeholder's perspectives towards cage culture growth in Lake Victoria, Uganda
- To assess suitability for Tilapia cage culture in Lake Victoria, Uganda.

- To evaluate trade-offs and synergies between cage farming and other ecosystem services on Lake Victoria
- To evaluate approaches for organic waste impact assessment at cage sites
- To assess the applicability of statistical modelling in developing protocols for monitoring cage impacts
- To make recommendations for policy intervention

Following this introductory chapter, Chapter 2 presents a situation analysis of cage fish farming on the lake. Organized into two parts, the first is an assessment of the status of fish farming. An overview of growth and farm operations is provided. In the second part, a review of the institutional and regulatory framework available at regional and local level and the gaps therein are presented. Perceptions of stakeholders towards the growth of cage aquaculture are explored in Chapter 3. This gives us evidence on what aspects of aquaculture development in the lake stakeholders think need attention. Both Chapter 2 and Chapter 3 describe the context under which cage farming on Lake Victoria is currently happening. The findings provide the basis for subsequent chapters. Chapter 4 presents the suitability for cage production using small and large cages. While acknowledging that the lake serves other users apart from cage farming, in Chapter 5, spatial prioritization is utilized as a tool for trade-off analysis among the various ecosystem services the lake offers. An assessment of the impact of cages on sediment and water quality in a typical cage farm is presented in Chapter 6. Methods available for on-farm organic waste impact assessment are evaluated for limitations of use in licensing decision support specific to eutrophic freshwater systems. Acknowledging complex ecosystem dynamics, technical and financial challenges for water quality monitoring in Lake Victoria, in Chapter 7, statistical regression models are used to reduce the parameters monitored. The resulting regression equations are tested for different hydrodynamic conditions. Chapter 6 and 7 explore robust monitoring protocols that could be used on-farm for licensing or on-farm rapid impact assessment within a technically and financially constrained context. Finally, suggestions are made for policy interventions. Chapter 8 presents conclusions, policy implications, and limitations of the research.

CHAPTER 2

STATUS OF CAGE CULTURE IN LAKE VICTORIA (UGANDA)

2.1 Introduction

A potential source of employment, income, and fish protein, aquaculture has long been a priority for the Ugandan government and development partners (NPA, 2020). This sector can improve the livelihoods of millions. For years, production depended upon ponds owned by smallholder farmers producing primarily for subsistence use and only selling the surplus. Pond production in Uganda, as was the case in some other sub-Saharan Africa countries (e.g., Kenya, Malawi), had stalled for decades. This was linked to the level of production done by small scale farmers scattered in rural areas; the high investment and operational costs; a lack of extension service and a general failure to benefit from economies of scale (Isyagi, 2017; Odende *et al.*, 2022). A dependence on government stimulus packages for fingerling and feed supply still exists among rural farmers today (Netherlands Enterprise Agency, 2022), a vice that was equally responsible for stagnating pond production.

Since the 2000's the government has shifted towards large scale production with the aim of making aquaculture contribute more to the sector's GDP. Uganda has been involved in tilapia cage culture on Lake Victoria since 2007 with support from the United States Development Agency (USAID) (Mbowa *et al.* 2017). This continued effort and private sector involvement along the value chain have started to yield results. In terms of quantity, Uganda ranked 3rd on the list of top farming countries in Africa in 2017 (FAO, 2020a), albeit amidst challenges of feed, quality seed and technical assistance (Adeleke *et al.*, 2021). The government and other partners committed funds to establish aquaculture parks (Isyagi, 2017). Aquaparks were primarily intended to fill the supply gap created by dwindling wild stocks by moving from rural subsistence production to a more consolidated commercial industry (Mugabira *et al.* 2013). Initial interventions into development of aquapark by the government started in 2017, under the project "Promoting Environmentally Sustainable Commercial Aquaculture in Uganda (PESCA)" (MAAIF and EU, 2018) funded by Nature Conservancy and European Union with plans for a cage and pond-based aquapark. Production estimates for 2020 indicate that of the average 35,250MT produced, pond production made-up 2-6% (Tilapia and catfish) while

the rest was done in cages (Netherlands Enterprise Agency, 2022). Increasing cage production is accompanied by intensification and expansion. This calls for proper planning and monitoring to ensure a sustainable industry with no compromise on other lake uses or the integrity of the ecosystem.

The aim of this chapter was to present the status of cage culture on Lake Victoria, Uganda. The study was conducted in two stages. First, an assessment of the farm production and management practices implemented, followed by a review of the existing institutional and legal framework for aquaculture and environmental management.

2.2 Methodology

2.2.1 Data Collection

Primary data was collected from fish farmers in a field survey. Secondly, feed samples were taken from the same farms for proximate analysis. A review of grey and published literature on existing institutional and regulatory framework pertaining to environmental management of cage farms was done. The literature reviewed included government reports, legislation, environmental impact assessment reports, and reports from regional governing bodies. Focus was drawn to specific legislation and guidelines that addressed cage production. A detailed evaluation was done to identify gaps in the regulations and guidelines.

2.2.2.1 Farm Survey

Google EarthPro (Google Earth, 2018) image dated 09/10/2019 was used to locate cages that were visible in the Ugandan part of Lake Victoria. Ground truthing was done between October and December 2019. First, a list of farms and their locations was obtained from the Department of Aquaculture Development and Management (DAMD), Ministry of Agriculture, Animal Industry and Fisheries (MAAIF). This list was not exhaustive and once fieldwork commenced; farmers were asked to suggest others they knew. At the farms, a structured survey was administered to gather information on farm production and management practices. Farm practices, infrastructure, and management practices were observed. Pictures were taken with permission from the farmers. Farms visited were selected based on accessibility and the farmer's willingness to allow a site

visit. Apart from the farms on the islands, most farms were accessible by road albeit with challenges in the rainy season. On average, smaller farms were less accessible because of their remote locations.

2.2.2.2 Feed sampling and Analysis

During the farm visits, feed samples were taken for analysis of the nutrient content to ascertain proximate composition. Different feed sizes available on the farms (powder, 0.3mm, 1mm, 3mm, 4mm, 5mm) were sampled and stored in universal tubes and analysis was done at the Nutrition Analysis laboratory, University of Stirling. Standard methods were employed to analyse for moisture, fat content, crude protein, lipids, energy, and phosphorus.

For protein determination, Kjeldahl method using the Opsis LiquidLINE system was used. 200mg of portions of the samples were put on a 42.5mm filter paper disc in duplicates and placed into a 250ml Kjeldahl digestion tube. 4cooper Kjeltabs and 12mls of concentrated sulphuric acid were added to each sample. The rack holding the tubes was put into the digestion block at 420°C for an 1hour. The tubes were then put aside in the fume cupboard and allowed to cool for 40mins. Distillation was then done using Opsis LiquidLINE KjelROC Analyzer and Opsis LiquidLINE KjelROC Sampler KD-525. Moisture content was determined by first grinding the samples to a powder using a grinder. It was weighed off and placed in a pre-weighed dish, put in the drying oven at 103°C for 4hours. The sample was removed from the oven, cooled in a desiccator for 30mins and reweighed. For Ash, 1g of the same was weighed into a pre-weighed crucible and placed in a muffle furnace at 600°C for 20 hours. They were removed and left to cool at room temperature in a desiccator and reweighed. Crude lipid was determined using Soxhlet extraction with petroleum ether. Gross energy was determined with a calorimeter bomb.

To compare results from the laboratory to what the bags in the field indicated, the chi-square statistical test to check for homogeneity between details of feed proximate values was carried out.

2.3 Results

2.3.2 Farmer Survey

Farms were classified into two categories: small and large. This classification was based on number of cages owned and production volume. Large farms were those with a production greater than 5 tons per cycle and importing own fish feed. The farmers visited were in the districts of Wakiso, Jinja, Buikwe and Kalangala (Figure 2.1), on basis that the farmers allowed a site visit. Seven large and three small farms were visited. Other farmers corresponded by email.

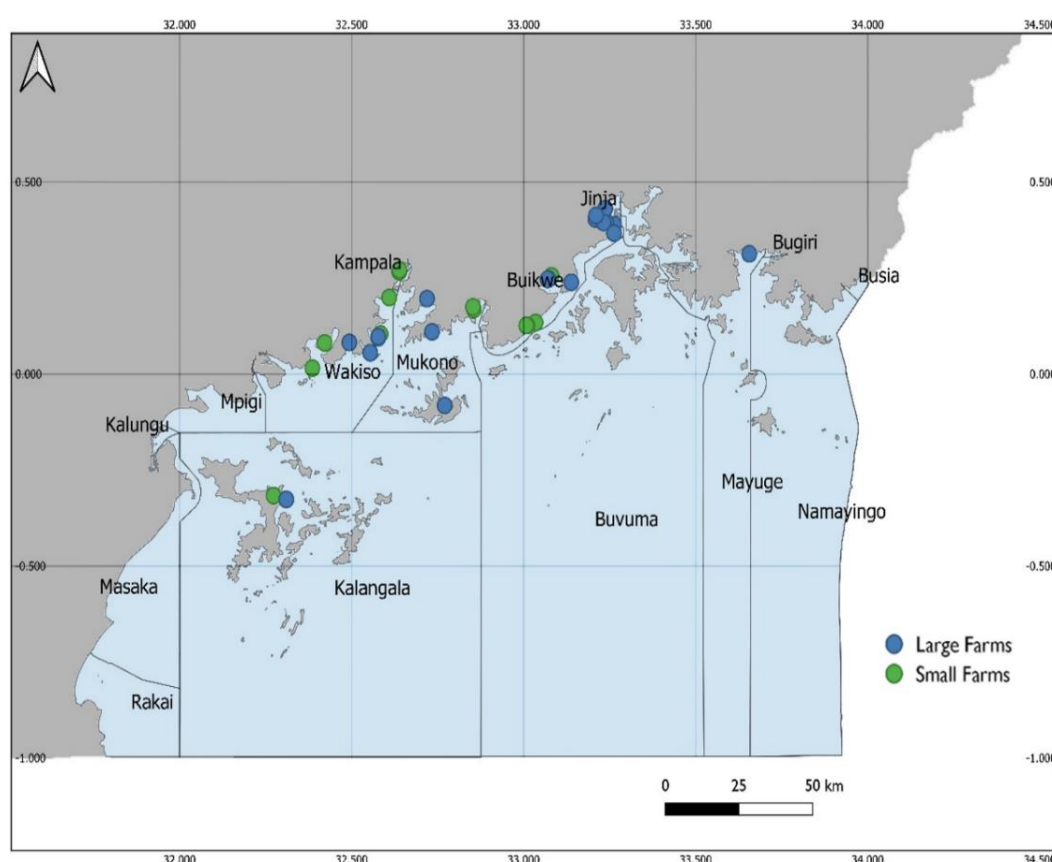


Figure 2.1. Location of cage farms in districts around the lake, categorised into small and large according to number of cages owned and production.

In total, 23 survey responses were obtained (Table 2.1). All the farms visited reported to have setup between 2008-2018 with only one setup earlier in 2006. Information about farming practices and other management aspects was collected (Table 2.1) and detailed are provided in section 2.3.2.1 to 2.3.2.6. Farm ownership ranged from individual to multinational and only one farm was owned under an association. Some large farms

used only small metallic cages, but their volume of production was above the 5 tons/cycle threshold and hence the classification.

Table 2.1: Summary of the general characteristics of farms (information obtained from surveys)

Farm aspect	Category	Number of farms in group	Specifics
Farm size and characteristics of cages dimensions	Small farms	15	Square metallic framed (1.5x1.5x3) m to (2x2x2.2) m
	Large farms	8	Varying between (2.5x2.5x3) m to (6x6x3) m for square cages and 10-20m diameter for circular cage
Year of establishment	From 2006-2018	NA	The earliest farm was set up in 2006, and since then, there has been steady increase in number of cage farms in different districts
Number of staff	Small farms		Average 2-4
	Large farms		10-120 (largest farm reported 110 staff)
Ownership	Sole by nationals	16	All small farms were owned by nationals
	Multi-national	6	All large farms were in this category
	Group/co-operatives	1	Farm had small square cages individually owned but within same area
Average production per cycle	< 5 tons/cycle	15	These were mostly small farms
	Between 5 to 10tons	2	These were categorised as large farms. Most reported several harvests in a calendar year
	> 10 tons/cycle	6	
Type of feed	Pelleted and Extruded	5	All large farms reported only pelleted feed
	Pelleted and Trash feed	18	Small farms reported using trash feed together with pellets
Source of fry	Produced their own fry	2	Most large farms owned hatcheries, and sold their surplus to other farmers
	Locally sourced	21	There are farmers that have capitalised in only fry production (in ponds) and the same were mentioned in many of the farms as the suppliers. Small farms too bought from re-known fry producers
Use of drugs and medicines	Yes	5	Some farms mentioned the use of sodium bicarbonate for prophylactic purposes
	No	18	
Feed administration	By hand	23	Feed was administered manually by hand in all farms

2.3.2.1 Types of cages used

Cage types ranged from smaller square metallic framed (1.5x1.5x3) m cages to large circular 20m diameter High-Density Polyethylene (HDPE) floating cages. Some farms had the Low-Density High Volume (LDHV) metallic cages with dimensions ranging between (2.2x2.2x2.5) to (2.5x2.5x3) m and larger ones that were (5x5x5) m while PVC ones with dimensions (6x6x3) m were common as well. Most farms especially large ones owned more than one or two types of cages for different purposes; conditioning fingerlings; production of juveniles or holding different batches of grow-out. The metallic cages (Figure 2.2 B) were mainly found in shallower waters compared to where the rectangular or circular HDPE cages (Figure 2.2 A, C and D) were. The frame of the metallic cages was made of galvanised pipes welded to a length that fitted several cages to be installed. The frame was fitted with empty plastic drums to act as floaters with the help of ropes. This conjoining created a structure to easily walk along once the cages are deployed. The nets were tightly attached to the frame and released into the water column, held in position by sinkers made of locally available materials. In one farm, the floating structure was created with flexible PVC pipes, rolled to form circular frames (Figure 2.2D). These circular cages had a diameter of 6m. In both cases, double netting was used, with an inner net that holds the fish and an outer net that prevented escape in case of a breakage on the inner one. The HDPE circular or square cages had a higher stocking density and were placed in deeper water. These types of cages were becoming more popular as seen at almost all large farm sites visited. The type of cages in Figure 2.2.A, B and C was common and less for the type in D. Small farm had only the small rectangular type as seen in Figure 2.2 B.

2.3.2.2 Stocking and Harvest

Fry was mostly sourced from local suppliers. Some large farms carried out their own fry production with brood stock harvested from the wild. Selective breeding was done, and hormones were used for all-male production. All the farms stocked mono-sex Tilapia with a production cycle that lasted an average of 7-8 months. Stocking was done when the fish weighed between 20 to 35gms. At one large farm using several rectangular cages of (1.5x1.5x3) m, fingerling stocking density was reported as 400 fish/m² translating into 133 fish/m³. In contrast, another farmer at a medium-sized farm using HDPE rectangular (6x6x6) m cages stocked 117 fish/m³ at an average weight of

20gm/fish. Almost all the farms raised fish throughout the year, i.e., they kept fish in juvenile cages, harvested adult cages during the same period, and restocked again.



Figure 2.2. Assembly point for square floating PVC pipe cages at a farm (A); Co-joined 2.5x2.5x3m metallic cages forming a frame to hold them in position (B); Circular 20m diameter floating cage (C); Circular LDHV cages made with PVC pipes (D).

Stocking was dependent on the availability of inputs, mainly seed and feed, making products available on the market all year round. Throughout the production cycle, on most farms, nets were often checked for breakages, to avoid loss of stock. Large farms had dedicated staff trained for the role. The average harvest size was 350-500gms,

however, timing for harvest varied, with the main drivers being availability of market or a lack of sufficient feed to prolong production. The latter was common among smallholders. Following price negotiations, buyers brought their motor vehicles to the farm, where the fish was harvested, weighed, and loaded into iced trucks. A few large farms owned outlets in cities and towns as well as explored regional markets at the Kenyan and Congolese borders.

2.3.2.3 Feed and Feeding

2.3.2.3.1 Feed sizes, storage, and administration

Farms reported that they imported feed from Mauritius, China, Israel, Vietnam, or Zambia. The feed came in pellets, powder, and crumbles (Figure 2.3A & B). Only one farm locally manufactured its feed. At another farm, under group ownership, trash feed and sinking pellets were reportedly used. The feed sizes administered was dependent on the size of the fish.



Figure 2.3. Fish feed samples from the farms (A).; Pellets of different sizes (B).; Powdered and Crumbles used on fish cages on the Lake. Pellets were sinking or the floating type.

Feed storage (Figure 2.4A) facilities were comparable among farms, however, there was a marked difference with what was seen at the farm owned by members in an association (Figure 2.5B). At this farm, it was reported that every farmer had their own feed store and used a feed they found affordable. The same farm reported the use of trash feed. Schedules of feeding and production data at this farm could neither be established nor confirmed due to lack of consolidated record keeping mechanisms.



Figure 2.4: Feed storage facilities commonly found in large farms (A.); individual storage at a farm owned by an association (B).

Feed administration at all farms was observed to be by-hand with none reporting automation. Some farmers used the experience they had accrued over time to manage feeding schedules. A farmer reported that during heavy waves and mixing on a lake, the waters became turbid. The fish would not respond well to feeding, so they reduced feeding schedules to once a day.

2.3.2.3.2 Fed Proximate Values

Most of the processed feeds seen (pellets, crumbles) came with details of the content and proximate ranges on the back of the feed bags (Figure 2.6). The variables included crude protein, phosphorus (%), crude fat, moisture, ash, calcium, crude fibre, lysine, total energy, and moisture content. Values were indicated as minimum or maximum content. Compared to results from analysis carried out at the institute of Aquaculture, there was no significant difference in the proximate ranges for fibre, protein, ash, moisture, and phosphorus content (Table 2.2).

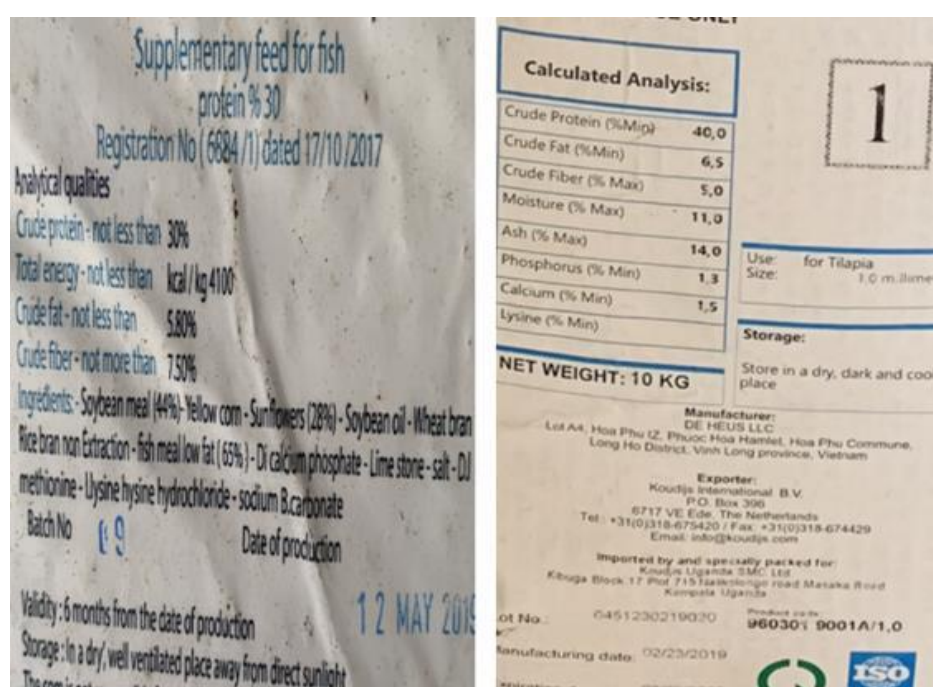


Figure 2.5: Proximate values of ingredients indicated on the feed bags.

Comparison was done for the feed sizes 0.3mm, 3mm, 4mm, and 5mm of fibre, protein, ash, moisture, and phosphorus content. Using the chi-square statistic to determine homogeneity between details of feed proximate values included on the bags and those from lab analysis (Table 2.2), no significant difference existed between the proximate values (significant difference occurs at $p < 0.05$). Other feed sizes were not compared due to missing information on the bags.

Table 2.2: Average proximate ranges for feed sizes analysed at the IoA compared to ranges included on bags in the field. No significant difference in the means for protein, ash, moisture, phosphorus, and fibre.

Feed size	% Protein		% Mean Lipid		Mean Energy Value (kj/g)		Mean %Ash		% Mean Fibre		%Phosphorus Content		Mean % Moisture	
	Lab	On bag	Lab	On bag	Lab	On bag	Lab	On bag	Lab	On bag	Lab	On bag	Lab	On bag
Powder	48.80	-	5.24	-	19.04	-	11.00	-	2.53	-	0.20	-	7.00	-
0.3mm	38.80	50.0	5.27	-	18.84	-	7.05	7.00	3.31	5.5	0.21	1.3	6.73	11.0
1.6mm	38.66	40.0	3.91	-	18.56	-	7.12	14.00	3.46	5.0	0.18	1.3	6.84	11.0
3mm	36.50	35.0	3.78	-	17.21	-	11.37	9.0	5.45	2.0	0.17	1.3	7.59	-
4mm	36.54	34.0	13.72	-	20.78	-	6.74	7.3	3.61	5.0	0.21	1.2	6.54	9.0
5mm	29.99	30.0	0.43	-	13.36	-	13.10	7.0	8.77	8.0	0.10	-	7.32	11.0
Comparison with ranges on labels from field (Pearson Chi-square);														
(2-sided														
significance														
test)														
	<i>p=0.333</i>		-		-		<i>p=0.238</i>		<i>p=0.333</i>		<i>p=0.092</i>		<i>p=0.156</i>	

Note: Some variables tested for in the laboratory were not presented on the feed bags i.e., energy and lipid content. Similarly, for some, feed sizes, the amounts were missing on the bag such as phosphorus in the 5mm feed.

2.3.2.4 Available value addition technologies and facilities

In some large farms, technologies applied to improve efficiency and save time spent during fish harvesting and grading were observed. It was reported that equipment mounted on a jetty made it possible to move between cages in a more efficient manner, thus reducing harvesting time and manpower (Figure 2.7 A). The same machine had compartments for grading fish according to size. From harvesting, the fish was ready for loading into the refrigerated transport trucks heading to the markets. This translated into reduced post-harvest losses. Other post-harvest handling facilities observed in proximity to one cage farm were ice plants and handling slabs (Figure 2.7B). Initially built in a joint project between the government and the African Development Bank (ADB) in 2002 (P-UG-AAF-002), over 7 similar facilities (Figure 2.7 C and D) were equipped with handling slabs, stand-by generators, ice plants, waste incinerators, toilets, offices, meeting, and dining rooms. These facilities were developed at fish landing sites around Lake Victoria to handle fish from the wild. Despite this, they were abandoned due to a decline in catches and lay unused, despite being maintained on the devolved governments' budgets or leased to private operators. These offer an opportunity for handling cage aquaculture products. This indicates a possibility that both small and large cage farms, existing and planned, could leverage quality assurance structures already available for post-harvest handling of fish from the wild. This includes roads leading to fish landing sites and ice plants (Figure 2.7 B).

2.3.2.5 Use of chemicals and medicines

During the farm visits, none of the farmers reported using any antifouling chemicals for net cleaning or chemicals for disease control or prophylactic uses. Some farmers reported prophylactic treatment using sodium bicarbonate as they transferred the fish from the hatchery that was located a long distance from the farm. At one farm, the farmer reported incidences of fish mortalities, even among bigger fish (Figure 2.7A). As a means of prophylactic treatment, the farmer infused sodium bicarbonate into the water by placing it in a sac and attaching it on the cage frame (Figure 2.7B.). This practice was common, reported in four other farms.

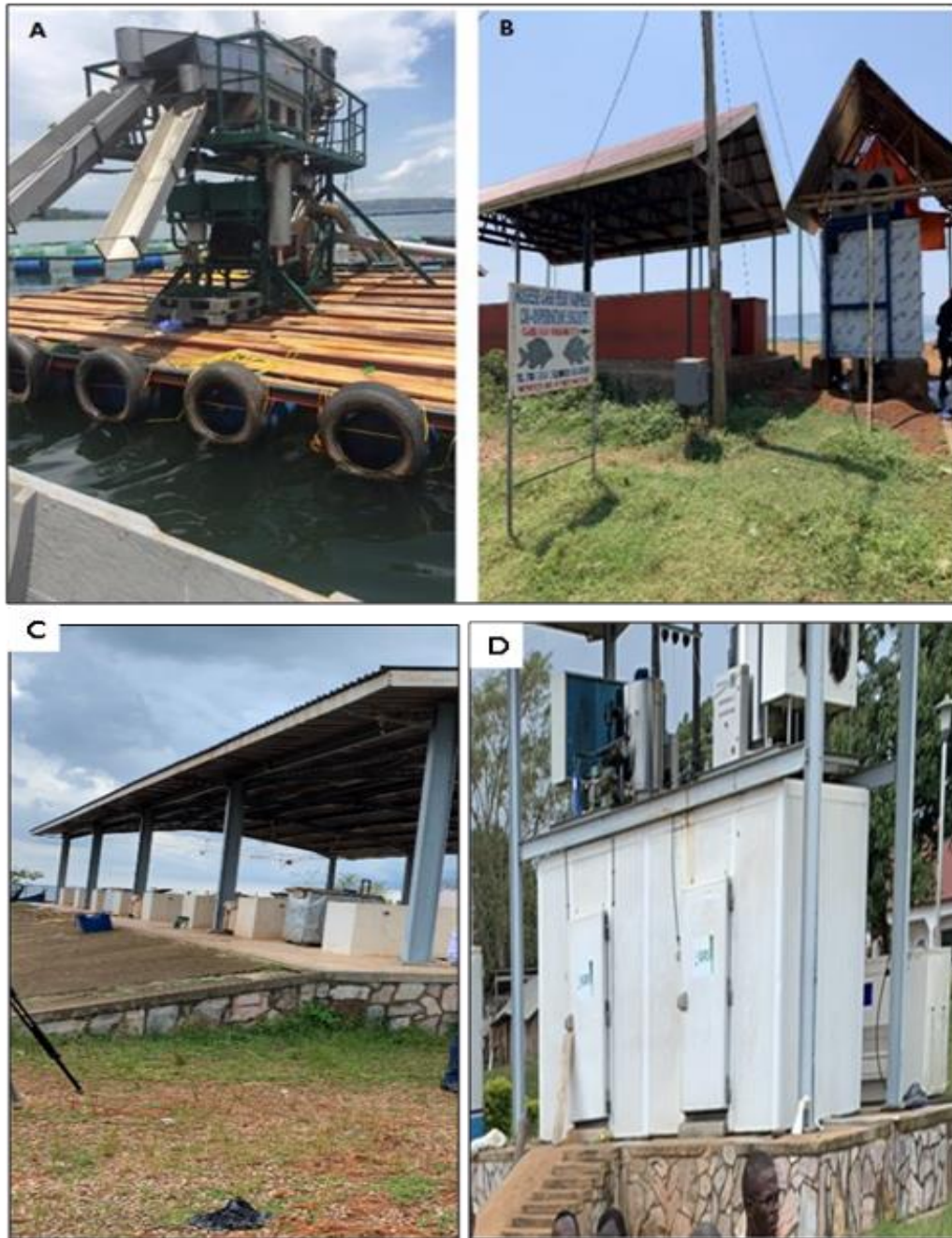


Figure 2.6: Value addition equipment observed on farm and in proximity to other sites. Automated grading machine at one of the fish farms (A); An Ice plant and a market stall, available for post-harvest handling near another cage farm (B); A slab and ice plant at ADB constructed facility earmarked for use om fish harvested for the aquapark site in Kalangala district (C & D).



Figure 2.7: Cases of disease and mortalities reported in some farms, lesions could be seen around the gills and the dorsal fins (red arrows) (A); Sodium bicarbonate put in a sac (red arrow) and let to permeate into the cage. Dead juvenile fish can be seen on water surface (B).

2.3.2.6 Employment

From the field visits, it was appreciated that cage fish farming could indeed contribute to employment within the riparian communities and along the value chain. At one of the sites, youth were involved in activities like transportation of fish feeds (Figure 2.8 A.) and construction of cages (Figure 2.8 B). Some traditional fishermen informed us that during their spare time, they offered paid services to the farms by operating boats during feeding excursions and harvesting or fish loading and feed offloading in trucks. One women's group reported to have received funding worth USD \$720,000 from Standard Bank South Africa, United Nations-Women and the Swedish government for a three-year cage culture project running 2019-2021. Owned and managed by women (Figure 2.8C), the project aimed at empowerment and emancipation of over 1,400 women.



Figure 2.8: Job creation along the value chain. Youth involved in feed transportation (A); Cage fabrication (B); and Women receiving a fish transportation truck as part of donor initiative towards cage farming (C) (Photo by UN-Women on Twitter).

2.3.3 Review of available legal and institutional framework

Management of cage aquaculture in Uganda is governed by strategies, policies, regulations, and voluntary guidelines developed at continental, regional and national levels (Figure 2.9). The devolved structures (district local governments) are legally permitted to develop and implement by-laws for management and utilisation of resources within their jurisdictions. Thus, adding another layer of framework within which cage farming is governed. Management institutions at the continental and regional level work hand in hand with those at national level to implement their mandates. On the continent, playing the key role of driving the African development agenda are the African Union (AU) and Intergovernmental Authority on Development (IGAD). At the regional level, harmonised management of Lake Victoria resources is under the East African Community (EAC). The regional and continental bodies provide guidance, with the mandate of adoption, domestication, implementation, legislating, and monitoring left to national governments. Guidelines developed at regional level in the EAC are debated upon by representatives from the partner states.

2.3.3.1 Regional Framework; strategies and guidelines

Institutional and regulatory framework for management of aquaculture resources in Africa focuses on ensuring that growth is sustainable, improves livelihoods, contributes to regional and global food and nutrition security without compromising other uses (AU-IBAR, 2019). The African Union is a continental bloc mandated to achieve growth and prosperity. Agenda 2063 (African Union Commission, 2015), the continent's strategic framework for inclusive and sustainable growth puts modernisation of agriculture and blue growth for accelerated economic development as major goals.

With processes overseen by the Inter-African Bureau for Animal Resources (AU-IBAR) at the AU level, aquaculture is predicted to achieve desired growth through blue growth transformation. Guidance is provided under the Continental Aquaculture Development Action Plan 2016 – 2025 (AU-IBAR, 2016) that was developed to guide implementation of the Policy Framework and Reform Strategy for Fisheries and Aquaculture in Africa (AUC-NEPAD, 2014) with the aim of supporting the Comprehensive Africa Agriculture Development Program (CAADP).

AU-IBAR developed a guideline to support implementation of the Regional Environmental Management Framework for Sustainable Aquaculture Development (AU-IBAR, 2019), which focuses on the transboundary nature of most African inland water systems, and the fact that such systems have a large population dependent on them with serious socio-political importance. In this framework, suggestions are made to ensure all users of ecosystems continue to benefit from their resources. A key driver behind the framework is that environmental integrity is not to be compromised, and regional management of transboundary resources is emphasized (AU-IBAR, 2019). Development of aquaculture zones at regional and national level is recommended as is the use of effective tools for monitoring the impacts of different aquaculture systems.

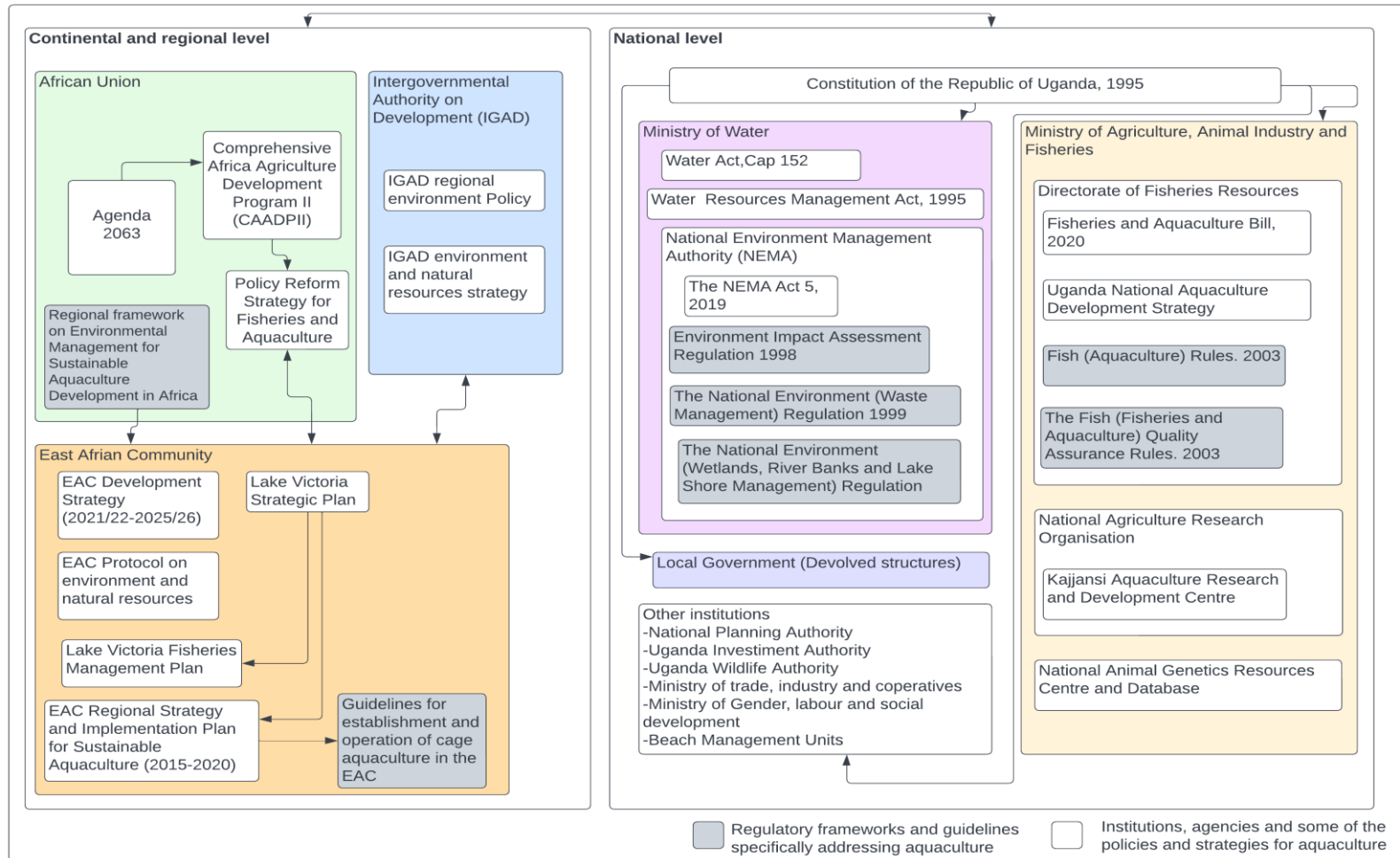


Figure 2.9: Institutional framework utilised in the planning and management of cage culture on Lake Victoria. Arrows indicate which framework stipulates the other or where there are feedbacks within institutions.

Within the East African Community (EAC), the Lake Victoria Fisheries Organisation (LVFO) oversees implementation of the EAC mandate on Lake Victoria. One of the guiding documents on aquaculture development is the EAC Regional Strategy and Implementation Plan (EAC-RSIP) for Sustainable Aquaculture (Michel de San, 2013) which recommends activities to be carried out at national level. These include the establishment of a functional Environmental Monitoring Unit (EMU), and Environmental Impact Assessment guidelines that are specific to aquaculture. Also, aquaculture water quality standards must be established, conflicts need to be resolved, an enabling environment must be created for investment, and feed industry taxes must be waived. To implement the EAC-RSIP, the Aquaculture Regional Working Group of the EAC developed the Guidelines for Establishment and Operation of Cage Aquaculture in the EAC (LVFO, 2018) in which they recognise that if not well planned, cage aquaculture is a potential source of conflict for the multi-stakeholders on the lake, in addition to the potential of environmental degradation. In addition to recommending criteria for site selection, the guideline emphasizes the importance of considering different parameters, including socioeconomic, environmental, and biophysical factors. In addition, recommendations are made that an Environmental and Social Impact Assessment (ESIA) is carried out for clearance from the national environmental management agencies, followed by an environmental management plan and annual audits done to keep cage culture production in check.

LVFO, (2018) recommends variables for monitoring of changes in the ecosystem and the frequency at which this should be done however the basis of these could not be verified. These include changes in fish assemblages and water quality physico-chemical parameters. A knowledge gap still exists regarding modern techniques that can be applied to continuous monitoring or how models can be used to assess the impacts of aquaculture production. This necessitates the isolation of impacts created by cage farming from those emanating from other sources.

2.3.3.2 National regulatory and Institutional framework

In Uganda, according to LVFO, (2018), regulatory authorities involved in aquaculture development include the Ministry of Agriculture, Animal Industry and Fisheries (Directorate of Fisheries Resources); National Agriculture Research Institute (NARO) that handles research; National Environmental Management Authority (NEMA); and the

Ministry of Water and Environment (Directorate of Water Development). Others like the Uganda Investment Authority; National Planning Authority; Ministry of Gender, Labour, and social development; Fisheries training institutions; National Animal Genetics Resources Centre and Database (NAGRI&DB); National Drug Authority for quality of feed and use of medicines; can be involved as of when their mandate relates to the project being implemented or monitored. Considering that most aquaculture production occurs in rural areas, local governments, i.e., decentralised entities, play an instrumental role in aquaculture production.

According to the Uganda National Water Policy of 1999 (MoWE, 1999), allocation of water for commercial aquaculture will be based on economic, social, and environmental values. However, emphasis is put on water abstraction for pond culture, where abstraction rates exceeding 400m³/day necessitate obtaining a water permit. For the case of cage farming, other principles of this act that might apply include Principle (V) on preservation and protection of the environment through the EIA process; Principle (VI) on private sector involvement in planning, development, operating and maintaining infrastructure for aquaculture among other uses; Principle (VIII) on the holistic and integrated approach towards development of water (including that for aquaculture) taking into consideration equitable use and protection of water resources. Finally, Principle (IX) is particular for shared water bodies, emphasizing respect of international obligations. In addition, water resources assessment recommendations include proper siting of aquaculture to optimise the resource and protect it. Assessment of the impacts of cage farming on the quality of the receiving water body, is stipulated as a condition for the issuance of permits.

The National Environmental Authority (NEMA) enacted under the NEMA Act 5, of 2019 (NEMA, 2019), has the mandate for regulating, monitoring, supervising, and coordinating all activities related to the environment. Environmental and Social Impact Assessment is requisite for clearance to be given to projects under this Act. Environmental Impact Assessment Regulations (1998) provide criteria and guidelines for conducting, evaluating, and monitoring environmental impact assessments. In the NEMA Act 5, 2019, under its mandate and functions, Article 9 (1) section (d), the authority has a function to support the mainstreaming and integration of environmental concerns in national and sectoral plans through coordination and collaboration with lead agencies. Section (d) of the NEMA Act 5 emphasizes that, where aquaculture related matters are concerned, NEMA works in collaboration with the relevant authorities to coordinate activities and ensure that social and environmental issues are effectively

dealt with. In addition, NEMA monitors and assesses activities that may degrade the environment. The Directorate of Fisheries Resources through the Department of Aquaculture Development and Management, on the other hand, has a mandate to develop regulations, enforce and foresee activities towards growth in the aquaculture sector. Environmental audits in Uganda are detailed in the National Environment (Audit) Regulations, 2006 (NEMA, 2006). The National Fisheries Resources Research Institute through the Aquaculture Research and Development Centre, Kajjansi, carries out research and provides scientific guidance to the Directorate of Fisheries Resources. However, no specifics are provided, like the cumulative organic loading or how to monitor it.

2.4 Discussion

This chapter focused on establishing the extent of Tilapia cage farming, practices utilised and regulatory framework available at regional and national level. The past decade has seen a rapid increase in the number of cage farms setup on Lake Victoria, Uganda. Information about the extent of production was obtained in surveys done on cage farms. A review of the regulatory framework available nationally and regionally was done to establish the context in which developments are happening. This discussion is divided into two sections. The first focuses on the farm findings, followed by the available regulatory framework.

This study confirmed growth in both large and small cage farms. Farm groupings were based on the type and number of cages used and the production volume per cycle. Farms were mostly located in the districts of Mukono, Jinja, Buikwe, Wakiso, Kampala and Kalangala. These districts are located near large cities and have a high volume of traffic and easy access to inputs. Among the cages used at large farms, HDPE circular cages were common. This could be due to the simplicity they provide when performing cage operations, and a limited investment cost (Cardia and Lovatelli, 2015). The ability for this type of cage to withstand strong winds and being plastic increased their durability. Small farmers operated small rectangular metallic cages that were manufactured from locally available materials, designed for operation in shallow waters. Harder to classify were farms that operated multiple small rectangular cages exceeding the 5tons/cycle threshold. Future efforts towards defining the criteria for farm scale i.e., small, medium, or large farm will be necessary to devise management plans tailored to the level of farm production. Stocking densities among large farms were within ranges of 117-133 fish/m³

which is slightly higher than the recommended ranges of 50-100 fish/m³ (Schmittou *et al.*, 1998). Among the smaller holders, at the farm owned under an association, stocking density estimates were unclear, but the farmer reported it to be according to availability of inputs (Majjidu, pers comm.). In Kenyan waters, stocking densities among smallholders were reported as high as 350-359 fingerlings/m³ (Njiru, Aura and Okechi, 2018).

Among the challenges reported by small cage farmers was the high cost of quality feed. Even as feed production on the African continent is improving as local investment and access to international suppliers improve, feeds are still exorbitantly expensive for small operators to obtain. It was established that this forced some small cage operators to use trash feeds; locally available household food waste (Mo, Man and Wong 2018), as an alternative. However, the use of trash feed has been linked to high organic loading from farms compared to those where floating pellets were used (Cai *et al.*, 2016) and needs to be strictly discouraged. Investment into upscaling feed formulation using locally available ingredients such as worms and insects will benefit farmers (Ragasa *et al.*, 2022). On the contrary, as it was established, large farms mostly owned by multinationals had an opportunity to benefit from economies of scale. Most reported to source their feed from international markets, leveraging on the government policy of subsidizing importation of agricultural inputs (Owani, 2013).

Having to compete for ingredients between humans and other animal feed manufacturers, aquaculture feed producers in sub-Saharan Africa are always faced with high costs of production, inadequate or poor-quality ingredients (most importantly proteins), forcing them to compromise on quality or quantity produced (Hasimuna *et al.*, 2019, Hinrichsen *et al.*, 2022). Hence, it was relevant that an assessment was done to establish the quality of feed used in cage farming on Lake Victoria. Results from the analysis of fish feed samples showed no significant differences between values obtained following laboratory analysis and the values provided on feed bags. For instance, average protein content for feed between 0.3-5mms were above 36% as recommended by Lim, (1989). Dietary energy: protein ratios were between 4.1-4.5kcal/g; all within moderate ranges for tilapia growth and survival (El-Sayed and Teshima, 1992). Phosphorus ranges for all feed sizes were between 0.15-0.2% which was less than 0.9% required for normal bone mineralisation (Lim, 1989). With high quality diets, phosphorus retention increases, lowering the amount excreted into the environment as waste (Bueno *et al.*, 2016). However, it is pertinent to note that as demand grows with more farmers utilising imported feeds, it's not uncommon that the quality of feeds gets compromised.

Controls and surveillance should be developed. Most importantly, interventions should be focused on equipping laboratories for monitoring the quality of feed sold to farmers. On feed administration, farmers reported not feeding on some days because of poor weather conditions but this could affect growth. Lim, (1989) recommended daily feeding of 3-10% of fish weight between 3-6 times a day for fish of sizes 1-5g to <100g at a temperature of 28°C, with less food and fewer feeding times for the larger fish. Feeding rate was reported to have a direct impact on the welfare of the fish (Flores-García *et al.*, 2022).

Development of aquaculture parks, as embedded in the National Development Plan of Government (NPA, 2020) may address challenges like access to inputs and markets (Adeleke *et al.*, 2021). In a lake with high nutrient loading from catchment runoff, atmospheric deposition, and sediment internal loading, concentrating intensive aquaculture production could have serious consequences. As such, proper site selection, monitoring, and proper biosecurity plans need to be in place if aquaculture parks are further implemented. From the field visits, it was evident that jobs were created along the value chain especially for youth and women. A detailed account of the specifics was not done. However, it is hypothesized that this will be a major contributor to the income and employment that is needed in the region. Taking advantage of existing value addition equipment like fish processing plants, built landing sites near cage farms, is a way to reduce post-harvest losses as well as ensure a high-quality product to effectively compete in the market. Improvements in rural infrastructure such as roads (OECD, 2016) will bring farmers closer to agricultural inputs and extension services, making their businesses more profitable while attracting more investors. During the fieldwork, limited accessibility due to poor seasonal roads and distant locations hindered the ability to physically visit most of the small farms.

Field observations indicated the presence of diseases at some of the fish farms. Mugimba *et al.*, (2018) reported the presence of Tilapia Lake Virus in farmed fish on Lake Victoria. The capacity for aquaculture pathogen detection, isolation and disease diagnosis needs strengthening, as does harmonised monitoring and comprehensive disease management (Akoll and Mwanja, 2012). A more robust plan with a concept of Farm Management Areas (FMA) and Disease Management Areas (DMA) as seen in the salmon industry (Murray and Gubbins, 2017) could be explored.

Uganda has formulated several policies, regulations, guidelines, and is a member of several regional and continental frameworks for managing aquatic resources. However, Rwakakamba, (2009) found that the availability of laws and policies did not translate into

effective protection of natural resources including water basin catchments. Ntiba *et al.*, (2001) reported a lack of harmonised laws for management of effluents from point sources in the lake watershed as a major challenge, and this is still persistent. Management institutions such as AUIBAR and LVFO play an advisory role. Beyond this, national governments must commit, establish, and develop legislative measures for monitoring and enforcement within their jurisdictions. Having various institutions with mandates around the issuance of licences and permits for aquaculture development, as was established in the Ugandan context might be a cause of delays in the clearance processes. Goulding and Kamel, (2013) found that having to obtain a long list of licenses and permits from multiple authorities resulted into over 60% of fish farms in Egypt being unlicensed. The reduction of time spent doing business by creating a more centralised system, where roles are delegated from among line ministries to one or two authorities, could instead reduce bureaucracies and attract foreign investment. Coordinated efforts towards creating an integrated framework for multi-stakeholder application and approval like the Aquaculture Toolbox developed for supporting licensing and decision making for European aquaculture (TAPAS, 2020) will be needed, especially for shared water bodies like Lake Victoria.

Environmental Impact assessment was found as one of the common tools recommended by regional and national authorities for management of cage farming (LVFO, 2018; AU-IBAR, 2019, NEMA, 2019). The NEMA Act 5, 2019 in Schedule 5 section 6 (f, h and i) provides for cage farms in any lake in the country, necessitating all commercial production to implement an EIA (NEMA, 2019) as a licensing condition. However, EIA has been criticized on how impacts are monitored and how actions are implemented. Traditionally, EIA targets mitigation or minimisation of significant impacts (Christopher, 2007), however, cage culture impacts may not be significant but cumulative, worse under slow hydrographic conditions (Beveridge, 2004). Tools like predictive modelling were not mentioned anywhere in the EIA guidelines neither were they used in any of the EIA reports seen yet they are crucial in predicting impacts (Ortolano and Shepherd, 1995). Robust protocols for impact assessment are required to avoid over dependence on assessments that end at “business as usual”. Possible irreversible impacts on flora and fauna in the vicinity of the cages need evaluation and decisions made based on empirical evidence. Failure to effectively include stakeholders in the EIA process (Betey and Godfred, 2013; Kaku, Zhu and Fangninou, 2022) would fail for reasons of transparency and inclusiveness. In addition to suggesting mitigation measures, EIAs can influence fundamental decision changes, such as adapting alternative actions or changing the location of projects (Ortolano and Shepherd, 1995).

However, in the global South, when projects are government priority, such decisions are mostly political and making changes to utilize alternatives, or completely abandoning projects seldom happens.

2.5 Conclusion

This chapter establishes the status of cage culture on the Ugandan part of the lake, the existing regulatory framework, and the gaps therein. It is established that Tilapia cage production on the lake is increasing. Small cage farmers are scattered in different parts of the lake and highly impacted by the inability to benefit from economies of scale. Interventions in quality feed affordability will benefit both the farmers and the environment in which production is done. An enabling environment that promotes local investment is needed to avoid developing an unbalanced sector where smallholders are alienated from development. Undoubtedly, cage farming could boost fish supply, nutrition security and employment in the region. With the promotion of large aquaculture farms that can run on their own capital and handle shocks easily, the country is on track to contribute substantially to regional (and international) fish supply chains. However, growth in production needs to be planned to avoid impacting other sectors benefiting from the lake. To prevent the escalation of externalities that negatively affect farm production and the environment, diseases and organic enrichment from cages need to be acknowledged. Incentivizing aquaculture to attract both local and international investment will ensure profits from fish farming benefit various stakeholder groups. AU-IBAR, EAC and several FAO guidelines recommend measures and procedures, whose adoption and further implementation, monitoring and enforcement is done by individual states. In other words, these recommendations are not legally binding and states sharing the lake must adopt and domesticate these guidelines and disseminate them to end-users in common languages in addition to devising tools to monitor the implementation of the regulations.

CHAPTER 3

STAKEHOLDER PERSPECTIVES ON CAGE AQUACULTURE GROWTH IN LAKE VICTORIA, UGANDA

3.1 Introduction

Utilisation of common pool resources shared among multiple users for aquaculture development calls for equity and social inclusion in planning and management (Aguilar-Manjarrez *et al.*, 2017). Lake Victoria is a shared water body and up until recently, the major socioeconomic activity has been fishing. Fisheries directly and indirectly support over three million people in the East African region (El-Noshokaty, 2017). Management of Lake Victoria's fisheries has evolved over time, with marked periods of self-governance (pre-colonial times); top-down management; co-management structures and most recently military involvement on the Ugandan side of the lake (Mpomwenda *et al.*, 2022).

Top-down approaches in natural resources governance are common in traditional centralised authoritarian control (Hishamunda, *et al.*, 2014) and in Lake Victoria these have been criticized for alienating key stakeholders from decision making processes (Ogwang, *et al.*, 2009). Military involvement has been a temporary political move by the state to have control over the fisheries resources (Kantel, 2019). Lake Victoria fisheries co-management included stakeholders' opinions, though it had its own failures such as better results in social realms than core management goals (Obiero *et al.*, 2015). In such low-income economies where livelihoods are highly dependent on natural resource exploitation, it is seen to allow for optimal utilisation of local knowledge and a sense of ownership (Pyakurel and Marasini, 2021).

To set a better foundation for the aquaculture sector in Lake Victoria, stakeholders' beliefs, views, or opinions need to be captured in paving way for cage aquaculture development on Lake Victoria. Inclusion of the interests of the stakeholders is important (Aguilar-Manjarrez *et al.*, 2017). Recent reports have been made about cage farming being an infringement to fishers' access on the lake (Nunan, 2018). Amidst the economic and contentious political tensions, extending such thinking may impact on aquaculture

businesses. For a sustainable aquaculture sector growth, management failures made in fisheries like disregarding scientific recommendations made prior to Nile Perch introduction and implementation of closed seasons to manage declining wild stock yields need to be avoided (Balirwa, 2007). In that case, legitimised stakeholder input is desirable compared to political, administrative, or economic rationalism (Armatas, *et al.*, 2014) to ensure equity and social inclusion.

The perception of aquaculture is affected by people's socio-economic interactions (Krause *et al.*, 2015), or perceived environmental impacts (Whitmarsh and Palmieri, 2009). Given, (2008) states that "Individual perception influences opinion, judgment, understanding of a situation or person, meaning of an experience, and how one responds to a situation". Relating to aquaculture development in Lake Victoria, the stakeholders working on the lake are hypothesized to have perceptions of the growth in fish farming. To establish these subjective views, the Q-methodology which focuses on scientifically observing the inner self of others (Ramlo and Newman, 2011) is applied.

This study applies a mixed method approach known as the Q Methodology (Watts and Stenner, 2005). Q methodology is defined as the most elaborate form of statistically understanding human subjectivity (Given, 2008). Originally developed in 1953, the Q methodology aims to provide a way of objectively understanding subjectivity (Stephenson and Stephenson, 1994). The method unearths subtle differences in perspectives on a topic as well as highlighting major ones (Coogan and Herrington, 2011). In this methodology no hypothesis is tested, but rather, the focus is on generating and evaluating possible explanatory theories (Watts and Stenner, 2012), views or opinions that target group members may hold. Q methodology does not seek to create correlations between the study subjects and potential objective factors (Cairns, 2012), but to identify what the shared or common meanings among participants are (Watts and Stenner, 2012).

The aim of this study is to establish stakeholder's perspectives towards cage culture growth on Lake Victoria. Since lake users are diverse, there is a likelihood of obtaining a wide range of views. Q methodology is applied because it is a quantitative measure of subjective data collected from individuals with knowledge on the topic of interest (Webler, *et. al* 2009), and it fits socially debatable topics (Cairns, 2012). Focus here is on establishing what the views held by participants are, rather than how frequent these views exist. This research highlights key issues or concerns that stakeholders might have, and possible intervention areas for decision makers.

3.2 Methodology

3.2.1 Study design

The steps followed in executing the Q methodology in the current study (Figure 3.1) involved, first, Key Informant (KI) interviews conducted to generate literature specific to the Lake Victoria context. Second, was reducing the responses from the interviews (the concourse) into statements that formed the Q-set (a list of statements summarised from the interviews but covering all aspects presented). Third was recruitment of participants who worked in the fisheries or aquaculture on Lake Victoria to conduct the sorting of statements according to how much they agreed or disagreed with them. Also called the Q sorting. These participants are called the P-set. Due to COVID 19 restrictions, the sorting, analysis of the results and the post and pre-sorting surveys was done using the same online sorting tool as detailed in the sections 3.2.4.

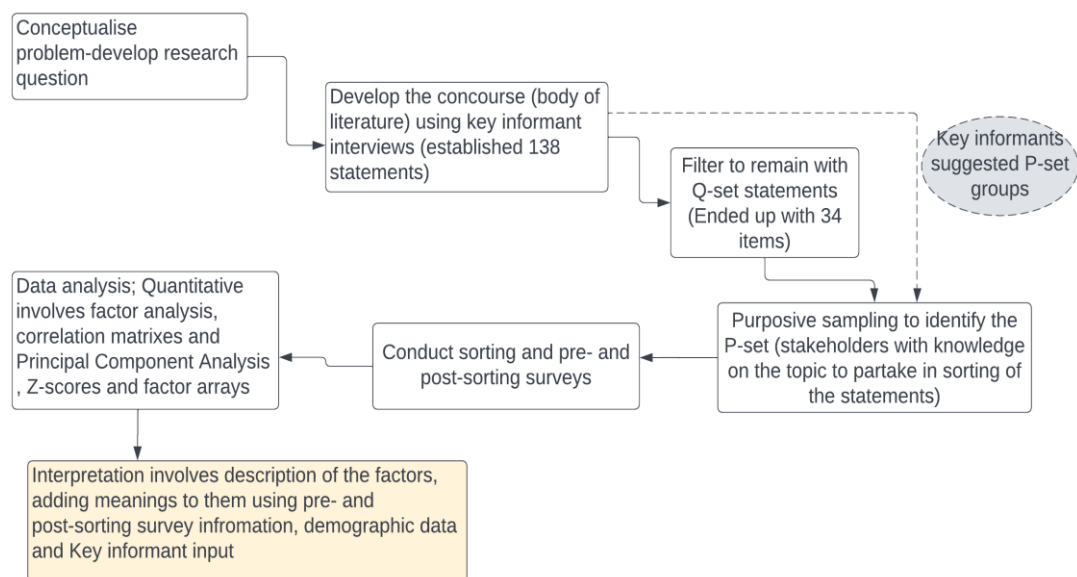


Figure 3.1: Summary of steps followed in the study. These included generating a body of literature from KI interviews; reducing them to statements which were sorted to generate patterns of similarity that formed shared views.

3.2.2 Collection and selection of the statements

Key Informants (KIs) from 10 groups Table 3.1. with experience working on Lake Victoria were recruited for semi-structured interviews. The interviews were held between March and April 2021 and were conducted online. Our guiding questions were: (i) how aquaculture can benefit the environment, society, locally, and regionally; (ii) Are there any concerns regarding cage aquaculture growth on the lake and (iii) how cage farming would affect the environment, and society. During these open discussions, participants were encouraged to speak freely about the topic, avoiding political correctness and all responses were considered equally.

Responses of the KIs were transcribed. 138 statements were drawn from the submissions of KIs. These statements represented the pros, cons, and neutral dimensions of the topic. This formed the wider body of literature termed the concourse in Q methodology studies. These were summarised, while ensuring both negative and positive aspects of the topic were covered to develop the Q-set; a list of 34 statements. Randomly, each was assigned a number from 1 to 34. To check for clarity and conciseness, the list of statements was shared with five experts in aquaculture. These experts were two from the Institute of Aquaculture, University of Stirling, one from an aquaculture training institute and two from regulatory authorities in Uganda. The five participants were not involved in either the interview or sorting phases of the study.

Table 3.1: Categories and numbers of participants that took part in the interviews and sorting the statements.

Stakeholder category /Type	No of KI interviewed (n=10)	Q sorting (P-set) (n=14)
Academics	2	2
Cage farm owners	1	1
Cage operative	-	2
Fisherman	1	1
Co-management structure representative (member of Beach Management Unit)	1	1
Aquaculture consultant	1	-
Central government staff	2	3
Local government staff (devolved systems)	2	3
Fisherman; also, a leader in fishing community	-	1

3.2.3 Participant recruitment for the sorting

In the KI interviews, respondents were asked to make recommendations of the stakeholder groups to take part in sorting part of the data collection. From the suggested groups, 14 participants from 9 groups were selected to make the P-set (Table 3.1). These stakeholders were not necessarily located in the proximity of the cage farms, but were a group thought to have knowledge on the topic. The strategy was that these views were as divergent as possible hence the use of purposive recruitment which is an integral part of using the Q methodology (Webler et al 2009). The knowledge participants had would determine the shape of the bell-shaped sorting tool (Figure 3.2) used in data collection. Environmental NGOs were recommended by the KIs as one of the groups, however, when contacted, no response was received back.

3.2.4 Q sorting

Q sorting was the third step in the data collection conducted online between May and July 2021 using the online tool Q-Method Software™ (Lutfallah and Buchanan, 2019). This is a web-based application in which a pre-and post-sorting survey was incorporated together with step-by-step guide.

Prior to- and after the sorting, participants filled a pre-and post-sorting survey to obtain details on their demographics and brief explanation about the structure of their final sort. Participants were required to rank-order the 34 statements into a forced-distribution quasi-normal 9-point, +4 to -4 distribution bell-shaped grid (Figure 3.2) depending on how they agreed or disagreed with the statement. For instance, in terms of agreement with, $Z > Y > X$ while $W = Z$ and in position 0 were the neutral statements. Participants were given an opportunity to do a systematic comparison of all the statements and use relative judgement, making the output most representative of their thoughts. The aim was to promote a holistic thinking. Forced distribution meant that each one of the boxes could take only one of the 34 statements and until all statements were positioned, the process would not proceed.

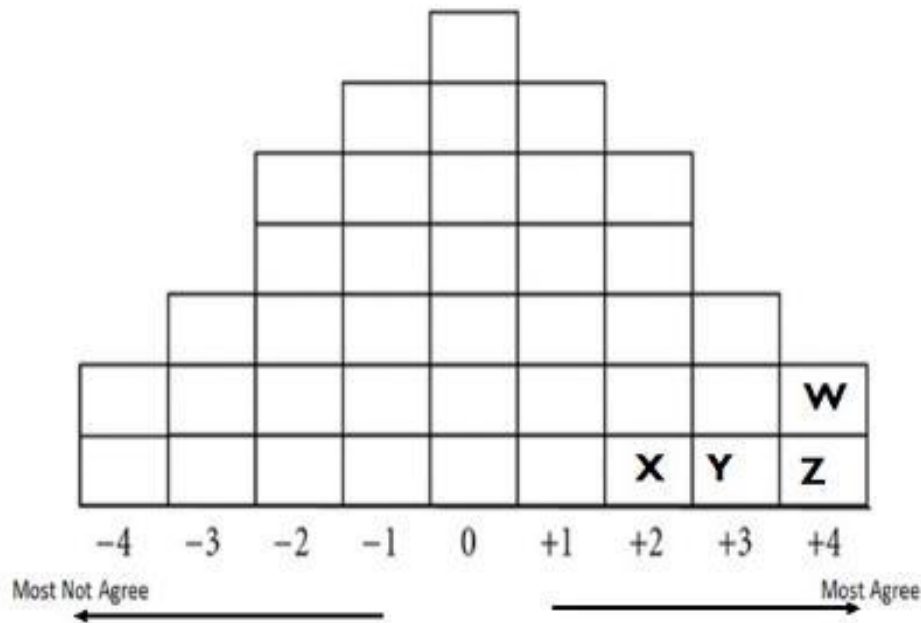


Figure 3.2: Quasi-normal grid on which Q set are sorted. Horizontal rather than vertical placement is of interest in the methodology.

3.2.5 Statistical Analysis

3.2.5.1 Factor analysis and Principal Component Analysis

When all the sorting was done, the 14 Q sorts were analysed using the inbuilt modules of the Q Method software™. The methodology utilised factor analysis and Principal Component Analysis to cluster the responses based on similarity in how Q sorting was conducted. Factor analysis was used to identify these patterns of similarity that represented shared meanings. First a correlation matrix that showed how participants' Q sorts correlated with each other was generated (Watts and Stenner, 2012). Some participants significantly loaded on a factor, and these formed the exemplary Q sort against which other loaders sorts were benchmarked. A measure of how a participant's Q sort was atypical of (or mimicked) that factor was given by their factor loading. Item configuration (arrangement of the statements in a Q sort) was similar within a factor but different from one factor to the next (Watts and Stenner, 2005). In instances where participants sorted the statements in very similar ways, that is, highly correlated Q sorts, the placement of the items in their sorts was closely related. Such patterns of similarity in the item configuration formed the unrotated factors.

To reduce the dimensions of the data and capture the variability in the item configuration among the Q sorts, Principal Component Analysis (PCA), a data reduction technique

was used. Factors were rotated using varimax rotation with an auto-flagging set at $p < 0.01$ (99% confident the results did not occur by chance), and a common variance required. The common variance represented the full range of variability in meanings present in the study. Varimax rotation provided maximum study variance and a statistically significant result (Watts and Stenner, 2012). The PCA produced eight (8) principal components, from which four (4) factors were selected for rotation. For every factor rotation, the eigen value i.e., the strength of the shared meaning, decreased as the factor loadings became weaker. The eigen value was given by the sum of square factor loadings for all the Q sorts on that factor and was an indication of the explanatory potential of that factor (Watts and Stenner, 2012). Each rotation produced different factor loadings for each factor. However, the overall aim was that the Q sort as placed as close as possible to the Q sort presented by that of the significant loaders. Several rotations were done, changing the number of rotated factors between 4-6, until no difference was seen in the results. The retained factors gave the most accurate estimate of the Q sort (most exemplary) and were a weighted average of the individual Q sorts that significantly loaded on that factor. Generation of a factor array representing the most exemplary Q sort of that factor is as explained in section 3.2.5.2.

3.2.5.2 Factor estimates.

The factor estimates that forms the factor array were a weighted average of the Q sorts that significantly loaded on that factor. This meant that those with higher loading, had a higher weight and hence influence on the item configuration in the extracted factor array. To calculate the factor array, first, the weights for all the Q sorts significantly loading on the factor were generated. The study used a 9-point +4 to -4 distribution in the design of the inverted bell-shaped quasi-normal grid (Figure 3.2). For calculation of each factor ranking corresponding to a particular Q sort, the ranking distribution was scored from 9 to 1 i.e., a Q sort ranking of +4 corresponded to 9, +3 corresponded to 8, +2 corresponded to 7 till -4 that corresponded to 1. To obtain the weighted score, for each statement on a given factor, the rank given was interpreted on the 1-9 scale, the value multiplied by the factor weight to obtain the new weighted score. The Z-score were generated as an output and interpreted as how those who select a factor prioritised each statement against all other statements (Carr, 2019).

3.2.5.3 Factor retention

To retain a factor, the criteria followed was an eigenvalue greater than 1 at $p < 0.01$ i.e., Kaiser-Guttman criterion (Watts and Stenner, 2012); and presence of distinguishing statements to differentiate among the extracted factors were used. Furthermore, two or more significant loaders automatically flagged on analysis (at $p < 0.01$) (Humphrey's rule) were required. Each of the retained factors were interpreted first by the determining statements at the extreme end (+4, +3; -4, -3) until those in the neutral or indifference zone of 0. These meanings were built on by the pre- and post-survey responses, differentiating statements and from support statements obtained from the KI interviews.

3.3 Results

Overall, four factors were extracted. Table 3.2 shows how stakeholders who significantly loaded on one factor performed on other factors. Factor 1 accounted for the largest study variance of 37%. This factor was significantly loaded on by six Q-sorters; 4 males and 2 females, who on average had 14 years working on Lake Victoria. Factor 2 accounted for 12% of the study variance, loaded on significantly by 4 Q sorters: 3 males and 1 female with an average of 25 years work experience on the lake. Factor 3 explained 11% of the study variance but was significantly loaded on by one loader, failed to reach the minimum requirement of two and hence, was not included further. Finally Factor 4, explained 8% of the study variance and had two significant loaders, hence, was retained. This factor was loaded on by 2 females with an average of 15 years work experience. One confounding sort was generated when the sorter; failed to significantly load on any of the factors (Table 3.2). Of the four extracted factors, only three were further considered for interpretation into perspectives.

The factor array (Table 3.3) was automatically generated in the data analysis and was used to ensure that the derived meanings were from analysis of how each statement featured in each of the factors retained. For each factor to culminate into a perspective, a name, and a summarised account of the determining statements (those with scores from ± 4 , ± 3 , ± 2) and the distinguishing statements were internalised and used in explaining the three perspectives; the institutionalists (Factor1), Economists (Factor 2) and the Environmentalists (Factor 3) as detailed in section 3.3.2

Table 3.2: Extracted factors, stakeholders that significantly loaded on them, how they loaded on other factors, defining factor loadings (in bold), and stakeholder years of experience of working on the lake.

	Factor 1	Factor 2	Factor 3	Factor 4	Years of experience
Factor 1					
Regulator (Central Government)	0.680	0.300	-0.318	0.295	9
Cage Farm Owner	0.826	0.149	-0.112	0.023	14
Regulator (Local Government)	0.704	0.064	0.185	0.069	20
Co-Management Representative.	0.591	0.263	0.458	-0.051	20
Regulator (Local Government)	0.772	0.240	0.311	0.065	15
Regulator (Local Government)	0.554	-0.117	0.317	-0.364	5
Factor 2					
Cage farm operative	0.238	0.829	0.062	0.069	6
Academic - Aquaculture technical Institute	0.049	0.835	-0.012	0.010	20
Fisherman	0.140	0.814	0.353	-0.020	45
Fisherman-Village head	0.446	0.532	0.419	0.174	27
Factor 3					
Regulator (Central Government) *	0.056	0.140	0.912	0.088	11
Factor 4					
Academic	-0.069	-0.162	-0.001	0.815	20
Regulator (Central Government)	0.235	0.313	0.099	0.669	10
Confounding sort					
Cage farm operative	0.393	0.277	0.381	-0.129	6
Eigenvalues	5.13	1.67	1.51	1.14	
% Explained Variance	37	12	11	8	
Total defining Q sorts (Significant loaders)	6	4	1	2	

Table 3.3: Factor Array of all statements across factors 1, 2, and 3

Statement Number	Statement	Factor 1 Rank	Factor 2 Rank	Factor 3 Rank
1	Cage aquaculture contributes to food security.	1	2	-1
2	In cages, we know how much we stock and can estimate our harvest, unlike in fisheries.	2	2	-1
3	Sections of the lake should not be leased to investors but rather, encourage farmers to engage with community to create rapport and ensure social acceptability	0	0	-4
4	Wasted feed from the cages is source of food for wild fish hence cages act as refuge and contribute to replenishment of wild stock.	0	1	1
5	Faecal matter and waste feed from cage farms pollute the lake.	-2	-2	2
6	There is potential increase in competition for shoreline space among lake users due to cage farming.	-3	-1	-2
7	Cage farming is a potential livelihood alternative for fishermen	0	4	1
8	Shift from command and control to co-management and shared responsibility will encourage private sector investment especially among small scale farmers	2	-2	1
9	Cage aquaculture reduces available space for other activities in the lake	-4	0	1
10	Open access fisheries have been abused by their beneficiaries; sustainable production is easier to manage with cage farms.	1	-3	-2
11	Cage farms risk fish escape and hence a potential for ecological destabilization.	-3	-2	2
12	There is potential for job creation along the cage farming value chain.	-1	4	-2
13	There is a possibility of genetic contamination if genetically improved species are introduced in Lake Victoria.	-2	-1	3
14	Sufficient research, monitoring, and surveillance should be done to check what is being farmed and potential interactions with wild species.	3	-1	1
15	Cage farming is growing faster than the laws, which is not most ideal.	-2	1	0
16	We risk social imbalances where cage farming interferes with fisheries, because the fisheries support a larger population.	-3	-3	-2

Table 3.3: Factor array-cont'd

17	Aquaculture policy should adopt research findings to guide the sector using a precautionary approach and avoid detrimental effects on ecosystems.	2	-1	3
18	Cage farms bring infrastructure closer to lake communities such as roads, and electricity.	0	1	-1
19	Smallholder farmers do not have enough capacity to import feed, technology, and expertise.	-1	2	4
20	Availing tools and skills for water quality management is key.	2	3	2
21	Having a map that shows what is allowed in different parts of the lake will guide effective licensing and avoid conflict among users.	3	3	-3
22	Proper structuring of production to avoid exceeding carrying capacity is important to protect the lake from degradation.	-1	1	-1
23	Regional and international cooperation is needed in research, trade, and utilisation of technology to promote cage farming.	1	0	-3
24	If local breeds of Nile Tilapia perform better, farmers have no reason to opt for genetically modified species.	-2	-4	-4
25	Baseline data is important to collect before further expansion of cage farming on the Lake.	4	0	0
26	Laxity in monitoring, surveillance, and reporting can compromise ecological integrity.	1	0	0
27	Inadequate research and lack of farmer skills hinders disease surveillance, and reporting.	0	0	4
28	Poor implementation of social safeguards might lead to workers exploitation on cage farms	0	-3	2
29	Stakeholder engagement, and access to information is one of the key factors for sustainable cage growth on the lake	3	2	0
30	Aquaculture is one of the key sources of introduction of fish diseases in water bodies in case of occurrence in cage farms	-1	-1	-1
31	Lake Victoria is being sold to private investors	-4	-4	0
32	Foreign exchange from export of cage fish products contributes to the GDP of the country	-1	3	0
33	Farmers need training and guidance to develop sustainable aquaculture practices on the lake	4	1	3
34	Officers need to be facilitated and covered by law to carry out their mandate.	1	-2	-3

3.3.1 Factor distinguishing and consensus statements.

Distinguishing statements (Table 3.4) were autogenerated in the analysis and were those that differentiated one perspective from others. Specifically, sorters who loaded on one factor placed a statement in a position significantly different to where participants who loaded on other factors placed it (Coogan and Herrington, 2011). Consensus statements (Table 3.5) on the other hand were those which the sorters for the different factors had similar or closely related thoughts on. Together with the factor arrays, the distinguishing and consensus statements were used in compounding the factor titles for each of the three perspectives and in clearly differentiating among them.

Table 3.4: Distinguishing statements for the factors and their corresponding factor score in brackets.

Distinguishing Statements for Factor 1	
14	Sufficient research, monitoring, and surveillance should be done to check what is being farmed and potential interactions with wild species. (+3)
34	Officers need to be facilitated and covered by law to carry out their mandate) (+1)
19	Smallholder farmers do not have enough capacity to import feed, technology, and expertise (-1).
6	There is potential increase in competition for shoreline space among lake users due to cage farming (-3).
Distinguishing Statements for Factor 2	
12	There is potential for job creation along the cage farming value chain (+4).
32	Foreign exchange from export of cage fish products contributes to the GDP of the country (-1)
33	Farmers need training and guidance to develop sustainable aquaculture practices on the lake (+1).
9	Cage aquaculture reduces available space for other activities in the lake (0).
Distinguishing Statements for Factor 3	
11	Cage farms risk fish escape and hence a potential for ecological destabilization (+2).
9	Cage aquaculture reduces available space for other activities in the lake (+1).
14	Sufficient research, monitoring, and surveillance should be done to check what is being farmed and potential interactions with wild species (+1).
31	Lake Victoria is being sold to private investors (0).
3	Sections of the lake should not be leased to investors but rather, encourage farmers to engage with community to create rapport and ensure social acceptability (-4)

Table 3.5: Consensus statements for factors 1, 2 and 3 and corresponding factor scores

Consensus statements for Factor 1	
26	Laxity in monitoring, surveillance, and reporting can compromise ecological integrity (1).
4	Wasted feed from the cages is source of food for wild fish hence cages act as refuge and contribute to replenishment of wild stock (0).
18	Cage farms bring infrastructure closer to lake communities such as roads, and electricity (0).
22	Proper structuring of production to avoid exceeding carrying capacity is important to protect the lake from degradation (-1).
30	Aquaculture is one of the key sources of introduction of fish diseases in water bodies in case of occurrence in cage farms (-1).
16	We risk social imbalances where cage farming interferes with fisheries, because the fisheries support a larger population (-3).
Consensus statements for Factor 2	
2	In cages, we know how much we stock and can estimate our harvest, unlike in fisheries (2).
22	Proper structuring of production to avoid exceeding carrying capacity is important to protect the lake from degradation (1).
4	Wasted feed from the cages is source of food for wild fish hence cages act as refuge and contribute to replenishment of wild stock. (1)
26	Laxity in monitoring, surveillance, and reporting can compromise ecological integrity (0).
30	Aquaculture is one of the key sources of introduction of fish diseases in water bodies in case of occurrence in cage farms (-1).
16	We risk social imbalances where cage farming interferes with fisheries, because the fisheries support a larger population (-2).
Consensus statements for Factor 3	
4	Wasted feed from the cages is source of food for wild fish hence cages act as refuge and contribute to replenishment of wild stock. (+1)
26	Laxity in monitoring, surveillance, and reporting can compromise ecological integrity (0).
15	Cage farming is growing faster than the laws, which is not most ideal (0).
30	Aquaculture is one of the key sources of introduction of fish diseases in water bodies in case of occurrence in cage farms (-1).
22	Proper structuring of production to avoid exceeding carrying capacity is important to protect the lake from degradation (-1).
16	We risk social imbalances where cage farming interferes with fisheries, because the fisheries support a larger population (-2).

On the consensus statements, focus is drawn to the statement that are common among all three factors. Statement 4: *wasted feed from the cages is source of food for wild fish hence cages act as refuge and contribute to replenishment of wild stock* that ranked (0, +1, +1) , statement 26: *laxity in monitoring, surveillance, and reporting can compromise ecological integrity* ranked (+1,0,0), statement 22: *proper structuring of production to avoid exceeding carrying capacity is important to protect the lake from degradation*

ranked (-1,1,-1) and statement 30: *aquaculture is one of the key sources of introduction of fish diseases in water bodies in case of occurrence in cage farms* ranked (-1,-1,-1) for factors 1,2, and 3 respectively. These statements were interpreted as an area of indifference due to their placement in the array among all stakeholders. This highlighted potential need for further research on specific issues like aquaculture diseases, carrying capacity and monitoring cage production; to establish a case specific to Lake Victoria.

3.3.2 Detailed account of the perspectives

For interpretation, three perspectives were generated: factor 1 was built into perspective 1, factor 2 into perspective 2 and factor 3 into perspective 3. Together with the factor arrays, distinguishing statements, pre-and post -sorting survey feedback and KI interview detailed accounts, detailed meanings of the perspectives were generated. Specifically, the naming, was constructed from the factor arrays and distinguishing statements. Details are presented with the statement number and score in brackets in section 3.3.2.1 to 3.3.2.3.

3.3.2.1 Perspective 1: “The Institutionalists”

These stakeholders viewed the roles of government and research institutions as core to facilitation of further growth of cage farming on the Lake. The need for baseline data (25; +4) and farmer training in sustainable aquaculture practices (33; +4) was highlighted. These stakeholders agreed that having a map that showed what was allowed in different parts of the lake would guide effective licensing and avoid conflicts among users (21; +3). The role of research institutions was emphasized with the ranking of statements like collection of baseline data (25; +4) and sufficient monitoring and surveillance to check what is being farmed and potential interactions with the wild (14; +3). The shift from command and control to co-management and shared responsibility (8; +2) was further supported by this statement given by a stakeholder in the KI interview: *“Uganda needs farmed fish, but law should be developed to guide than focus on penalisation of abusers. Current laws are not private-sector support in nature. Farmers need to make profit, so naturally, they will stop at nothing, hence, statutory instrument should be guiding than the use of command-and-control approaches.”* The same stakeholder further added, *“Private investor attraction rather than government providing stimuli to encourage people*

into cage farming is important. More farmer-led than government facilitation in terms of feed, and other inputs, is what should be done”.

Stakeholder engagement and access to information being key to sustainable cage aquaculture growth on the lake (29; +3) is a key role of regulatory institutions. One stakeholder had this to say: *“It is important that authorities provide the public and potential investors with the right information on how they make money using Best Aquaculture Practices, in simple, clear understandable languages, and encourage self-regulation. All players should know their roles, including what is expected of each party to ensure sustainable production and environmental integrity”.* Another stakeholder said, *“for sustainable growth, there is need for coordinated management to ensure all stakeholders are involved in developing legislation”.* Stakeholders for this perspective did not think that cage aquaculture reduced available space for other activities in the lake (9; -4), and that cage aquaculture posed a risk of ecological destabilisation if fish escaped (11; -3). Neither did they think Lake Victoria was being sold to private investors (31; -4). These stakeholders were uncertain about cage aquaculture being a potential livelihood alternative for fishermen (7;0). In an interview, one stakeholder, a regulator said; *“Changing the lifestyle of fishermen to fish farming is not something that can happen so easily. Fishermen have been sensitized on numerous issues including better post-harvest handling, but till today, similar practices are happening”.* This group of stakeholders presented as those advocating for cage growth on Lake Victoria, following proper implementation of institutional functions, research, and farmer capacity building.

3.3.2.2 Perspective 2: The “Economists”

Stakeholders with this perspective were mostly for the economic benefits that could be accrued from cage farming; however, they also appreciated the environmental externalities involved. They strongly believed that cage aquaculture had potential for job creation along the value chain (12; +4) and that cage farming was a potential alternative livelihood for fishermen (7; +4). One of the stakeholders in an interview said, *“I own over 50 fishing boats, I pay for licenses, wages, nets, and fuel for the boats. In the past, I was always assured of recovering my costs, and making profits but today, my expenses are the same, but the boats return with no fish. If I knew how to grow my own fish in cages, I would invest in it.”* They believed that cage farming could be a source of foreign exchange from exports (32; +3) and that cage farming could contribute to food security (1; +2). These stakeholders didn’t agree that Lake Victoria was being sold (31; -4) but

agreed that having a map that showed what was allowed in different parts of the lake would guide effective licensing and avoid conflicts among users (21; +3) and concurred with the need for availing tools and skills for water quality management (20; +3). One stakeholder in this group said, *“Without proper knowledge on technologies and skill for water quality management, farms will also be impacted by waste realised over time. Training and availing tools is key for sustainable production”*. They agreed that proper structuring of production to avoid exceeding carrying capacity was important to protect the lake from degradation (22; +1). The stakeholders in this group did not agree that open access fisheries had been abused by their beneficiaries (10; -3) or that poor implementation of social safeguards could lead to workers exploitation on cage farms (28; -3).

3.3.2.3 Perspective 3: The “Environmentalists”

These stakeholders agreed that a capacity gap in research and farmer training would affect the Lake ecosystem. They agreed that smallholder farmers did not have enough capacity to import feed, technology, and expertise (19; +4), with one in the post-sorting survey stating that, *“Smallholder farmers do not have enough capacity and without clear guidance, they end up destroying the environment with rudimentary practices.”* The stakeholders in this group believed faecal matter and waste feed from cage farms polluted the lake (5; +2) and that there was a possibility of genetic contamination if genetically improved species were introduced in Lake Victoria (13; +3). In the post-survey questionnaire, one stakeholder added *“Pollution from cage farming is a concern yet current legislation does not effectively cover management of cage farm wastes”*. They agreed that cage farms were at risk of fish escape and hence a potential for ecological destabilization (11; +2) and that aquaculture policy needed to adopt research findings to guide the sector and avoid detrimental effects on ecosystems (17; +3). These stakeholders did not agree that where local Nile Tilapia breeds performed better, farmers had reason to opt for modified species (24; -4). One stakeholder added in the post-sort of survey that, *“deeper research was needed into introduction of genetically improved species prior to adopting the same on Lake Victoria.”* This group disagreed that there was potential for job creation along the aquaculture value chain (12; -2). One stakeholder in an interview said, *“Fisheries create a larger value chain that grows into communities, if cage farming is to interfere with fisheries production, this will result into imbalances in the society”*. The same stakeholders further added that *“Average number of lake users to benefit from cage farming is limited compared to those to be potentially*

displaced by the farming, and regardless of the fact that space is wide, some areas are not accessible for certain fisheries e.g., Tilapia.” This group further disagreed that a map which showed what was allowed in different parts of the lake would guide effective licensing and avoid conflict among users (21; -3) and that officers needed to be facilitated and covered by law to carry out their mandate (34; -3). In the interview, one of the stakeholders said, *“The lake is indeed a public good and restricting areas from access for cage farming will be a form of privatisation of public spaces”*. These stakeholders did not agree that open access fisheries have been abused by their beneficiaries; sustainable production is easier to manage with cage farms (10; -2) or that regional and international cooperation is needed in research, trade, and utilisation of technology to promote cage farming (23; -3). This group of stakeholders were indifferent about baseline data being important to collect before further expansion of cage farming on the lake (25;0) and if stakeholder engagement, and access to information being key factors for sustainable cage growth on the lake (29;0).

3.4 Discussion

This study used Q-methodology to explore what fisheries and aquaculture stakeholders on Lake Victoria thought about the growth in cage fish farming. The aim was to generate patterns of similarity among the stakeholders' views. This was done using a specialised methodology that provided a statistically acceptable outcome even with a small dataset. Indeed, a small number of participants is recommended to take part in Q sorting, as quality patterns and consistency can still be observed (Watts and Stenner, 2005). Webler et al (2009) recommended a ratio of 3:1 between the number of statements to be sorted to the number of participants. This was achieved when the 34 statements in the Q-set were sorted by 14 participants. Instead of studying a sample of the population, the aim of this study was to gain a deeper understanding of the perspectives held by stakeholder groups on the growth of cage culture.

Statements pertaining to five themes; social, environmental, research, governance, and economics of fish farms were compiled from key informant interviews and provided to a group of stakeholders for sorting into what they relatively agreed or disagreed with and what they were neutral or indifferent about. This abductive approach provided possibilities (Lee, 2017) from which the most adequate explanation of the situation was derived at the end of the study. From analysis of the data collected in the pre- and post-

sorting exercises, more insight into the results of the sorting exercise was generated, thereby taking advantage of the unique hybrid nature of qualitative and quantitative research methods embedded in the Q-methodology (Ramlo and Newman, 2011). In generating the concourse, a similar approach, of combining literature review, and using expert interviews was used by Bacher et al, (2014); MacDonald, et al (2015); Chikudza et al., (2020). This provided a rich body of context specific literature that was not obtained from literature review alone. Purposive respondent recruitment (Webler, et al, 2009) was used to avoid erroneously biasing results and misrepresenting opinions (Alexander, et al 2018).

Three perspectives emerged from the analysis; the first was concerned with institutions and the role they played, constituting the “Institutionalist” view. Second, the growth in cage farming was found significant for the economy and livelihoods of the people forming the “Economists” perspective; and lastly, the protection of the Lake Victoria environment was found of the utmost importance and cage culture externalities needed thorough consideration forming the “Environmentalists” perspective. These perspectives examined potential benefits, challenges, and areas for improvement for more sustainable growth. The “Institutionalists” and “Economists” presented their views in support of cage culture on the lake. However, the “Environmentalists” seemed more focused on the possible externalities of fish farming and less inclined to the potential benefits the sector could add.

Social views were not heavily loaded to generate a shared perspective. As can be seen in Ross et al., 2013, social carrying capacity is not given high priority in Africa. Comparatively, it ranks lower than physical, production, or ecological carrying capacities. A lack of involvement from players within the Environmental NGO sector might also explain this, or the infancy of the lake's cage culture development. These players normally advocate for social issues in similar regional contexts. All the consensus statements (4, 22, 26, 30) on which participants showed indifference among participants highlighted a possible knowledge gap because of limited research on the interaction between cage farming and the ecology of Lake Victoria. For instance, statement 22: *Proper structuring of production to avoid exceeding carrying capacity is important to protect the lake from degradation* was ranked (-1,1and -1) and statement 16: *We risk social imbalances where cage farming interferes with fisheries, because the fisheries support a larger population* was ranked (-3, -3 and -2) across significant loaders of factors 1,2, and 3 respectively. The lack of agreement on social implications could be linked to the fact that there has not been much pressure from development of the sector,

or an indication that more engagement needs to be done with the communities living near cage farms, to establish the social licence (Billing, 2018). Similarly, the neutrality on the carrying capacity aspect is an indication of the knowledge gap among stakeholders.

Cowx & Ogutu-Owhayo, (2019) mention the urgent need for good practices to govern cage culture on Africa's Great Lakes, as suggested by the "institutionalists". Orina *et al.*, (2021) highlights the role of government in ensuring availability of quality feed and seed, and the need to address issues like species introduction, and diseases as the sector grows while Munguti *et al.*, (2014) proposes that governments create an enabling environment for private sector investment. Furthermore, Aura *et al.*, (2018) suggests the development of policy and regulations to guide investment, protect local economies and ensure ecological and economic sustainability for cage farming on Lake Victoria. Investment in research, innovation in feed production and hatchery management is recommended for the growth of aquaculture within the East African region (AU-IBAR, 2016). As agreed by the "Institutionalists", Musinguzi *et al.*, (2019) recommends promotion of Best Aquaculture Practices and capacity building for institutions if cage farming is not to add on the pressure most African Great Lakes are already experiencing.

Lake Victoria offers a source of livelihood to a large population, amidst immense anthropogenic degradation (Musinguzi *et al.*, 2019). Results indicated agreement among stakeholders on the need for capacity building among farmers, to ensure they have basic skills, for instance, on water quality management. To sustain production at a given site, over time, environmental integrity should be maintained (Hishamunda, *et al* 2014). The third stakeholder perspective that was revealed in this study was that of the "Environmentalists" who were concerned about the potential environmental impacts of cage aquaculture on the lake. Similar concerns led Kashindye *et al.*, (2015) to recommend caution on authorisation of cage culture in an already eutrophic lake, with emphasis on proper monitoring. This study revealed that some stakeholders believe cage culture could be a potential source of pollution. However, further research is needed to determine how it compares to loading from the catchment and the lake sediment. These stakeholders agreed a gap in monitoring efforts. Effective monitoring to ensure pollution from cage aquaculture is maintained within acceptable limits is critical for farm management and the ecosystem at large. Having the requisite legal framework and technical capacity guides licensing and monitoring for environmental regulation (Njiru *et al* 2018). Water Quality standards must be developed reflective of local conditions and needs (Ross *et al.*, 2013) if ecological integrity was to be preserved.

Mwebaza-Ndawula *et al.*, (2013) suggested the promotion of cage culture with caution especially regarding site selection and the cumulative effects of several cages in an area. It is imperative to understand how species interact with endemic ones prior to such introductions (Soto *et al.*, 2006). Farmers' lack of capacity to import technologies and expertise was supported by Opiyo *et al.*, (2018) who suggested that capacity building of various stakeholders with skills for disease identification, and training of farmers on simple biosecurity measures was crucial for Lake Victoria.

3.5 Conclusion

An imperative element of the second principle of the Ecosystem Approach to Aquaculture is the improvement of human well-being and equity. In this regard, the perspectives of stakeholders towards the growth of cage farming on Lake Victoria were explored. Perspectives generated by patterns of similarity in stakeholder views focused on concerns about institutional gaps, environmental externalities of aquaculture, and economic benefits the development might bring to the region. Social concerns did not appear to be a significant perspective among respondents. This could be due to the level of aquaculture production at present having limited interference with other uses. Soliciting these views generated an understanding of the issues that stakeholders thought needed addressing. Based on these findings, context-specific interventions are developed. Further analysis revealed issues that all stakeholders were indifferent to. The placement of the consensus statements was interpreted as having neutral feelings or a lack of knowledge of the topic due to the way the ranking was done. These included statements pertaining to monitoring aquaculture production, the carrying capacity of the lake, the fate of waste in the water column, aquaculture disease and the social implications of aquaculture development. These findings contributed to framing the subsequent research questions.

Intrinsically, the methodology used, the Q methodology, seeks to explore the breadth of views that participants might have on a topic. As a result, the views expressed gave an understanding of what existed among the participants. The results presented what stakeholders with experience working on Lake Victoria thought about cage culture growth. The results represent patterns created from personal opinions hence high reliability, but they are not generalisable to the entire population. There can only be a finite number of significant shared meanings or perspectives that people with a level of

knowledge of a topic can hold. Using the Q methodology aided the process when access to respondents was limited by the COVID pandemic. Future studies could adopt an in-person approach compared to online sorting. This would aid the benefit of personal interaction and observations during the data collection phase.

CHAPTER 4

ASSESSMENT OF SUITABILITY FOR CAGE AQUACULTURE IN THE UGANDAN SECTION OF LAKE VICTORIA

4.1 Introduction

Lake Victoria's large surface area of 68,800km² makes it difficult to determine where the most appropriate locations for aquaculture production are. Zoning is done at regional or water-body scale, to identify and allocate areas that are biophysically and socio-economically suitable to fish farming (Aguilar-Manjarrez et al 2017). The other objectives relate to mitigating social conflicts, biosecurity, and disaster risk reduction, and to controlling environmental degradation by farm-level focused controls (Sanchez-Jerez *et al.*, 2016; Aguilar-Manjarrez *et al.*, 2017).

Several factors and constraints determine the profitability of aquaculture enterprises. This calls for consideration of these spatial differences in mapping potential for aquaculture development. GIS offers a suitable platform for integration of spatial datasets for assessment and modelling in aquaculture and ultimately supports decision making (Nath *et al.*, 2000; Falconer *et al.*, 2020). GIS-based analytical tools and decision support frameworks are recommended in spatial planning due to the ability to combine multiple criteria and the ability to work at different scales; local, regional, or global (Stelzenmüller *et al.*, 2017). Spatial models could be updated to include new features as suitable data becomes available; hence, they are not static but represent conditions based on the spatial and temporal resolution of the data.

The use of GIS for planning, monitoring, and modelling aquaculture production varies among geographies and this difference could be linked to different levels of aquaculture development, differences in adoption of scientific research, data availability and accessibility of tools to end users (Falconer *et al.*, 2020). Application of spatial modelling to facilitate cage aquaculture in lake systems has lagged compared to ponds and marine systems (Falconer *et al.*, 2020). Earlier studies applying GIS to assess the potential for aquaculture on the African continent included Kapetsky (1994); and Aguilar-Manjarrez and Nath (1998). In both cases, lakes were included as constraints, and eliminated in

the assessment. Examples of recent studies focusing on the application of GIS for aquaculture development in the African Great Lakes include Asmah *et al.*, (2021); Aura *et al.*, (2021); Hamilton *et al.*, (2020).

Cage culture on the Ugandan section of Lake Victoria is done on a commercial basis (Netherlands Enterprise Agency, 2022). A survey done in 2019 (presented in chapter 2 of this thesis) found that large, medium, and small-sized production cages were common. Small cages were mostly of the rectangular floating type, with dimensions ranging between (1.5x1.5x3) m to (3x3x3) m, mostly made locally using metallic or wooden frames. Larger cages were circular High Density Polyethene (HDPE) with diameter ranging between 20-30m, and a net depth of 6 metres. The medium-sized ones were square (6x6x6) m, commonly found in larger farms. Large farms especially those that conditioned their fingerlings or held parent stock had both small and large or medium cages while other smaller farms had mostly small rectangular cages.

The aim of this study is to assess the suitability for commercial tilapia cage culture in the Ugandan section of Lake Victoria using small and medium to large cages. The results of this assessment contribute to the identification of areas where detailed assessment could be done for locating farms. Ultimately, designating areas for cage culture would ease monitoring, licensing, and disease management, in addition to mitigating conflicts among lake users.

4.2 Methodology

4.2.1 Model development

The site suitability assessment was executed using a GIS software TerrSet (Clark Labs)(Eastman, 2016) and QGIS 2.6.1 (Brighton) and QGIS 3.12.0 (Bucureşti)(QGIS Development Team, 2020) for data processing. Selection of the layers to include as factors or constraints followed deliberations with experts but most importantly, availability of good quality data, in a usable format and of appropriate resolution. A conceptual framework showing the layers included and the steps followed in developing the model is shown in Figure 4.1. The steps taken in model development were (i) Data sourcing and pre-processing , (ii) Reclassification of the layers onto a uniform scale, (iii) Multi-Criteria Evaluation that involved weighting and combining layers according to their relative importance and (iv) Final model output; as subsequently detailed.

4.2.2 Model Outline

4.2.2.1 Structure

The first step in model development was identification of input data layers and sourcing for the data. Data layers used in decision making were either factors or constraints. Factors increased or diminished the alternative under consideration while constraints limited the alternative (Eastman, 2016). Lake Victoria Fisheries Organisation (LVFO) under the East African Community (EAC) provides guidance to the riparian states for utilisation of the resources of the lake. This study utilized some of the criteria for cage aquaculture on the lake suggested by LVFO (2018). Availability of data in a usable format, quality and resolution were determinants of inclusion. Spatial factor layers included distance from water hyacinth hotspots, distance from existing fish farms, access to markets, bathymetry, and visual impact. Constraints were protected areas, buffer zones around the lake, fish breeding grounds and buffer zones for navigation ways. All the layers were projected to Universal Transverse Mercator (UTM) Zone 36S specifications in Table 4.1. The model was developed at 90m resolution to the resolution in which the available spatial data layers were presented.

Table 4.1: Specification of the model spatial extent

Parameter	Dimension
Number of columns	3011
Number of rows	2013
Minimum X (UTM Coordinate)	340643.9953280
Maximum X (UTM Coordinate)	611633.9953280
Minimum Y (UTM Coordinate)	9879620.8011400
Maximum Y (UTM Coordinate)	10060790.8011400
X Resolution	90m
Y Resolution	90m

4.2.2.2 Data sources and pre-processing

Primary and secondary sources that included government reports, online databases, reports from natural resources management agencies (Table 4.2) were used. Researcher's knowledge about the study area was a valuable aspect too. Locations of the fish farms were the only primary dataset obtained during a fieldwork done in October-December 2019 (Chapter 2 of this thesis). Google Earth was used to map locations of recreation services, and other tourists' residences, and to validate the navigation lines.

The data was inspected, cleaned, and validated where necessary. Layers were then projected to same reference system and clipped to cover the extent of the study focusing in on only the Uganda section of Lake Victoria. Some datasets like water quality proxies (Chlorophyll a) were obtained from online repositories and pre-processed but due to gaps therein, eliminated from subsequent steps. Instead, proxy for water quality; water hyacinth hotspots that were available in literature were used.

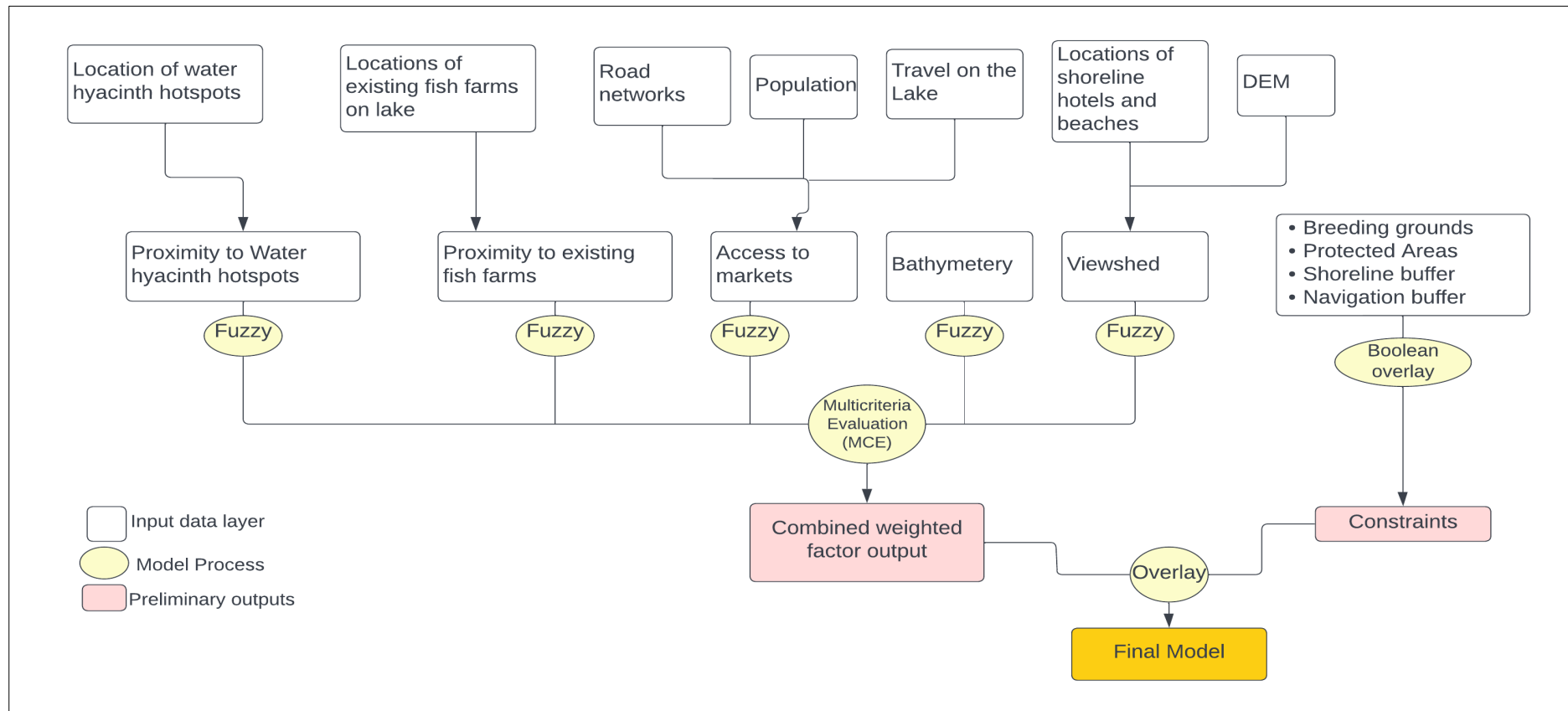


Figure 4.1: Schematic presentation of the model structure. Data for the input layers (top row) was sourced from online repositories or other secondary data sources. In some cases, manual digitization was done following fieldwork or researcher's knowledge of the study area.

Table 4.2: Data sources used for the developing the database.

Data layer	Type	Source Projection	Resolution	Source / custodian institution
Lake polygon	Vector (polygon)	Lat-Lon	n/a	Online data (Greg, 2015)
Lake shoreline	Vector (line)	Lat-Lon	n/a	Online data (Greg, 2015)
Bathymetry (m)	Raster	Africa Lambert Conformal Conic ESRI:102024 (https://epsg.io/102024) Geographic Datum: D_WGS_1984	100m	(Hamilton <i>et al.</i> , 2016)
Water hyacinth hotspots	Vector (Point)	Lat/Lon	n/a	Lake Victoria Basin Commission Secretariate, 2015
Road network	Vector (Line)	Lat/Lon	n/a	World Food Program GeoNode dataset (Crevoisier, 2017)
Recreation services and shoreline residences with swimming pools	Vector (Point)	Lat/Lon	n/a	Uganda Tourists Board Additional sites manually digitised on Google Earth
Digital Elevation Model	Raster	Lat/Lon Degrees Datum_Name: D_WGS_1984	90m	Jarvis <i>et al.</i> , 2008
LandScan 2018 global population database (number of people per cell)	Raster	Lat/Lon Degrees Datum_Name: D_WGS_1984	3arc sec	Oak Ridge National Laboratory (Rose <i>et al.</i> , 2018)
Location of cage fish farms	Vector (Point)	Lat/Lon	n/a	Primary data collected between in September 2019 using GPS
Gazetted fish breeding grounds	Vector (Point)	Lat/Lon	n/a	Uganda Statutory Instrument, The Fish (Fish Breeding Areas) Order, 2017.
Protected areas	Point (Vector)	Lat/Lon	n/a	Key Biodiversity Areas (KBA) (http://www.keybiodiversityareas.org/) Additional sites manually digitised on Google Earth using researcher's knowledge
Navigation Lines	Vector (Point)	Lat/Lon	n/a	Uganda National Roads Authority

4.2.3 Reclassifications of data layers

The original data layers had different units of measurement, so to create uniformity and enable combination in the modelling process, each data layer was reclassified to a numerical scale of 0 (unsuitable) to 1 (most suitable) (Eastman, 2016). There are different forms of reclassification the simplest being the Boolean method (Falconer *et al.*, 2018) with a score of 0 (no) or 1 (yes). This form of classification was used for the constraint layers, determining where aquaculture could take place or not. Other forms of classification include the use of user defined crisp boundaries where hard classes or ranges are provided from which a suitability score can be drawn. Limitations with using a user defined scheme is that distinct thresholds might not be available for the parameter in consideration (Falconer *et al.*, 2018). In the present assessment, factor layers were reclassified using another method called fuzzy reclassification (Eastman, 2016).

Fuzzy logic was selected because it gave a possibility (likelihood) rather than a probability (yes or no) of suitability for a given factor. This allowed for a degree of membership between classes (Eastman, 1999; Ssegane, Tollner and Veverica, 2012). A gradual transition from unsuitable to suitable scores was achieved, allowing a smooth transition from one level of suitability to the next. For most of the factors in this assessment, thresholds for the parameters were not accessible in the literature. An example was the distance from water hyacinth hotspots. Fuzzy reclassification has four membership categories: Sigmoid, Linear, J-shaped, and User-defined (Eastman, 2016). In this assessment the sigmoid (S-shaped) membership type was used. Specifics for reclassification were based on cage sizes with small cages ranging from (1.5x1.5x3) m to (3x3x3) m) and medium to large dimensions of (6x6x6) m to (20-30m diameter, 6m depth). To ease interpretation of the results (Falconer *et al.*, 2018), for the final model, fuzzy outputs were reclassified into a five-point score scale as later explained.

4.2.4 Model Inputs

4.2.4.1 Market Access

Accessibility in rural areas eases marketing and overall improves agricultural productivity and employment in the sector (Valdés *et al.*, 2009). Transportation of inputs such a feed, seed, and taking fish to markets requires infrastructure such as roads. Within the lake region, access to remote areas is characterised by poor connectivity to all-season roads. This translates into increased costs of production, inhibition of product and information flow and an increase in risks to the farmers (Bradbury *et al.*, 2017). Due to perishability of fish and most other

agricultural products, in some cases farmers are forced to sell their produce at farm-gate, attracting less profits (Hine, 2014). Hence, an accessibility layer was included in the current assessment that took into consideration the effect of road type and location of markets from the lake.

Access from any point along the lake shoreline to the markets via the different road types and access over the lake were considered. Access to the lake was assumed to be from any point along the shoreline due to lack of a dataset on access points. Markets (Figure 4.2) were defined as locations with a population greater than 25,000 people (Balk et al, 2010) , taken from the LandScan population distribution raster (Rose *et al.*, 2018).

The roads network dataset was obtained from the World Food Program GeoNode (Crevoisier, 2017) as a GIS vector file. The original data set had 12 classes (Table 4.3.) that were reclassified into three categories: primary, secondary, and tertiary roads (Figure 4.2). This was based on the number of lanes and surface type. Unclassified roads were eliminated in the subsequent assessment.

Table 4.3: Reclassification of the Uganda Roads Network Dataset

Number	Class	Number of lanes	Surface Type	Reclassified to...
1	Track	NA	Unpaved	Unclassified
2	Motorway	2-3	Asphalt	Primary
3	Primary	2-4	Asphalt	Primary
4	Primary link	1-2	Asphalt, Paved	Primary
5	Secondary	NA	Unpaved, Gravel	secondary
6	Residential	1	Ground	Unclassified
7	Tertiary	NA	Unpaved, Ground	Tertiary
8	Tertiary-link	NA	Ground or Gravel	Tertiary
9	Truck	NA	Unpaved	secondary
10	Trunk link	1-2	Asphalt	secondary
11	Tertiary link	NA	Unpaved	Tertiary
12	Unclassified	NA	Ground	Unclassified

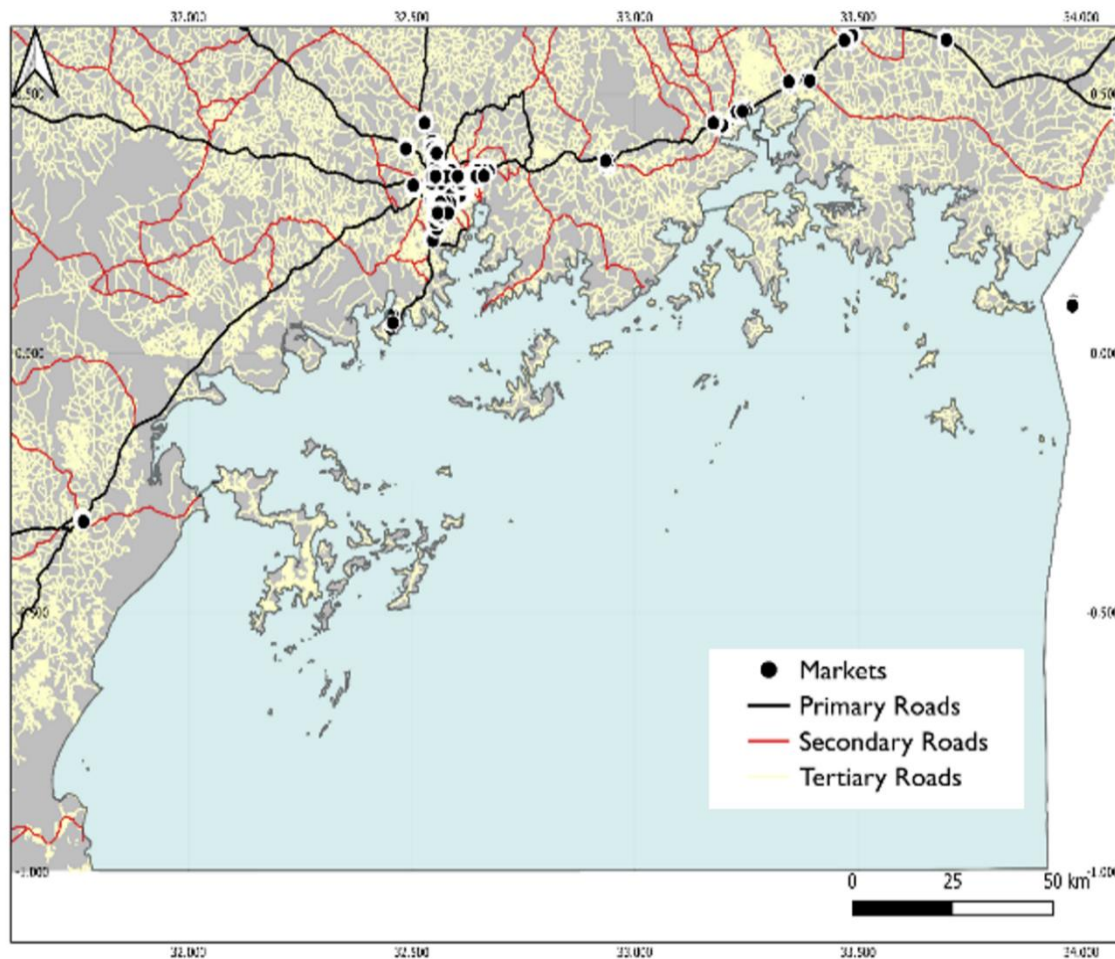


Figure 4.2: Location of markets and roads in the study area. The largest portion is covered in a dense network of seasonal tertiary roads.

4.2.4.2 Reclassification of the market access layer

Different road types were allocated different speeds adapted from Pozzi *et al.*, (2009). A friction surface layer where areas with no roads were taken as an impediment to travel, allocated a value of -1, and speeds on primary roads (60km/hr), secondary roads (30km/hr) and tertiary roads (10km/hr) was created. For interpretation purposes, the friction layer was reclassified into time taken to access each cell (minutes/cell) using a given road type. At 60Km/hr, it would take 0.09 minutes to cross a cell of 90m (spatial resolution). Similarly, at 30km/hr it would take 0.18 minutes/cell and at 10km/hr it would be 0.54 minutes/cell using secondary and tertiary roads respectively. For the ferry lines, a speed of 9 knots (Otieno Douglas, 1986) and equivalent of 16km/hr reclassified as 0.34 minutes/cell. The resulting costs layer was reclassified for small and medium to large cages (Table 4.4).

Table 4.4: Reclassification of the market access layer

Cage size	Membership Function	Membership Function Shape	Control points adapted from Ross <i>et al.</i> , (2011) Travel cost as time- (mins)			
			a	b	c	d
Small	Sigmoidal	Symmetric	5	10	30	60
Medium to Large	Sigmoidal	Symmetric	5	15	40	90

4.2.4.3 Proximity to Water hyacinth hotspots

Water Hyacinth (*Eichhornia crassipes*) weed has been linked to ecological and socio-economic impacts (Villamagna and Murphy, 2010). The water hyacinth was introduced in the lake in the 1980's (Williams *et al.*, 2005). The weed was linked to eutrophication (Albright *et al.*, 2004; Kabenge, Wang and Li, 2016) of the bays and gulfs of lake. By 1995, the weed covered 80% of the Ugandan waters (Matagi, 2002). On Lake Victoria, the floating weed was known to cause changes in the ecology and food-web, interfere with fish breeding, impeding access to fishing grounds, blocking navigation, secluding sections of the littoral zone, clogging hydropower turbines (Masifwa *et al* 2001; Otieno *et al.*, 2022) , increased incidences of waterborne diseases, and loss of lives among fisher communities (Lake Victoria Basin Commission Secretariate, 2015). Due to its fast growth rate, proliferation of the weed has always hard to control which over the years has caused high financial cost to riparian governments in managing it. Unclear circumstances linked to its rapid decline in 1997(Adrian *et al* 2007), with some attributing it to introduction of the weevils (*Neochetina spp.*), a biological control measure hastened with changes in weather conditions(Wilson *et al.*, 2007).

Data on the extent of the water hyacinth spread has always been based on rough estimates, field observations, and anecdotal sources. Lake Victoria Basin Commission Secretariate (2015) provided locations of hotspots for water hyacinth resurgence on the lake for the years 2012, 2013 and 2014. Based on these, a spatial layer for water hyacinth hotspots was created (Figure 4.3).

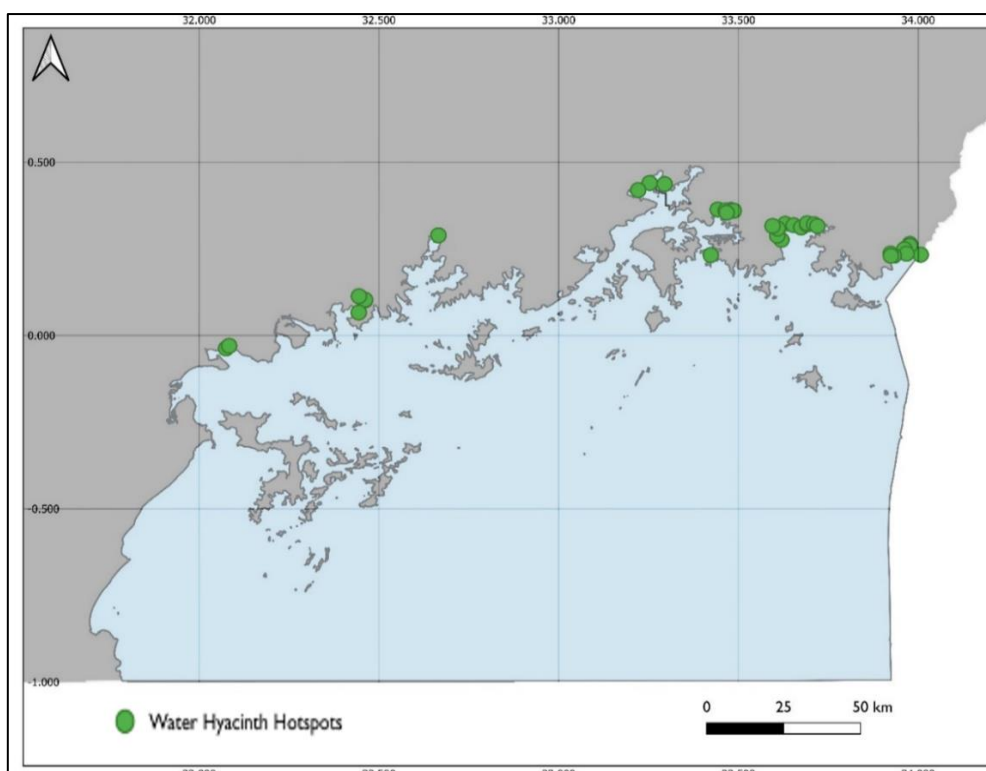


Figure 4.3: Mapped Water hyacinth hotspots on Lake Victoria (Modified from Lake Victoria Basin Commission Secretariate, (2015)

4.2.4.4 Reclassification of the Water Hyacinth Hotspots layer

The largest coverage recorded over the three years of monitoring was 80.7 hectares at a location on the western part of the lake. The weed is not static, and it has an exponential growth rate that depends on prevailing conditions (mostly winds and nutrient levels in the water). This makes estimation of its exact size challenging. In this assessment, following expert deliberations, a safety zone was created by setting a buffer of 1km around any of the hotspots. The further away fish cages are located from the hotspot buffers, the better are the chances of averting impact.

Smaller farms were assumed most liable to destruction in case of weed infestation. They were less likely to have manual equipment to clear it off and were more likely to be in nearer shore waters due to the cage sizes they use. Hence, reclassification was done using estimates (Table 4.5) with the estimated safety zone based on the potential risk and vulnerability due to cage size and financial feasibility of farm scale to employ mechanical weed removal.

Table 4.5: Reclassification of the Proximity to water hyacinth layer

Cage size	Membership Function	Membership Function Shape	Control points (Estimated distance from water hyacinth hotspots-km)	
Small	Sigmoidal	Monotonically increasing	a	b
			1	2
Medium to Large	Sigmoidal	Monotonically increasing	0.5	1

4.2.4.5 Visual Impact

The lake shoreline is characterised with swamp, sand, or rocky boulders in some parts. One of the key features that attracts residences or recreation services to the lake shoreline is the scenic seascape. Visual impact is one of the immediate consequences of cages on the nearby land or seascape (Pérez et al 2005). It can be considered as an interference with tourism or leisure activities which are social values (Falconer *et al.*, 2018) derived from the same space. Hence an important aspect to take into consideration in planning aquaculture on the lake. The features considered in developing the visual impact layer were locations of recreation services, beaches, marinas, residential places, and resorts within 500m of the lake shoreline which were digitised from Google Earth image dated 10/7/2018 (Google Earth, 2018). This involved identifying features relating to leisure centres or resorts such as sand along beaches, residences, or recreation services with swimming pools, (and) or golf courses. In total, 96 such places were digitised (Figure 4.4).

4.2.4.6 Reclassification of the Visual Impact layer

A viewer height of 2m, and a maximum search distance of 5km based on Falconer *et al.*, (2013) were used. Suitability for cage farming increased further from within the viewshed on the seascape. Larger cage farms were assumed to be immediately seen, have significant visual impact compared to smaller cages with minor components. Reclassification was done for the two cage types (Table 4.6.). In both cases, mere presence of the cages was seen as an impact on the view. The control points were set too close, almost creating a Boolean viewshed output for both types of cages.

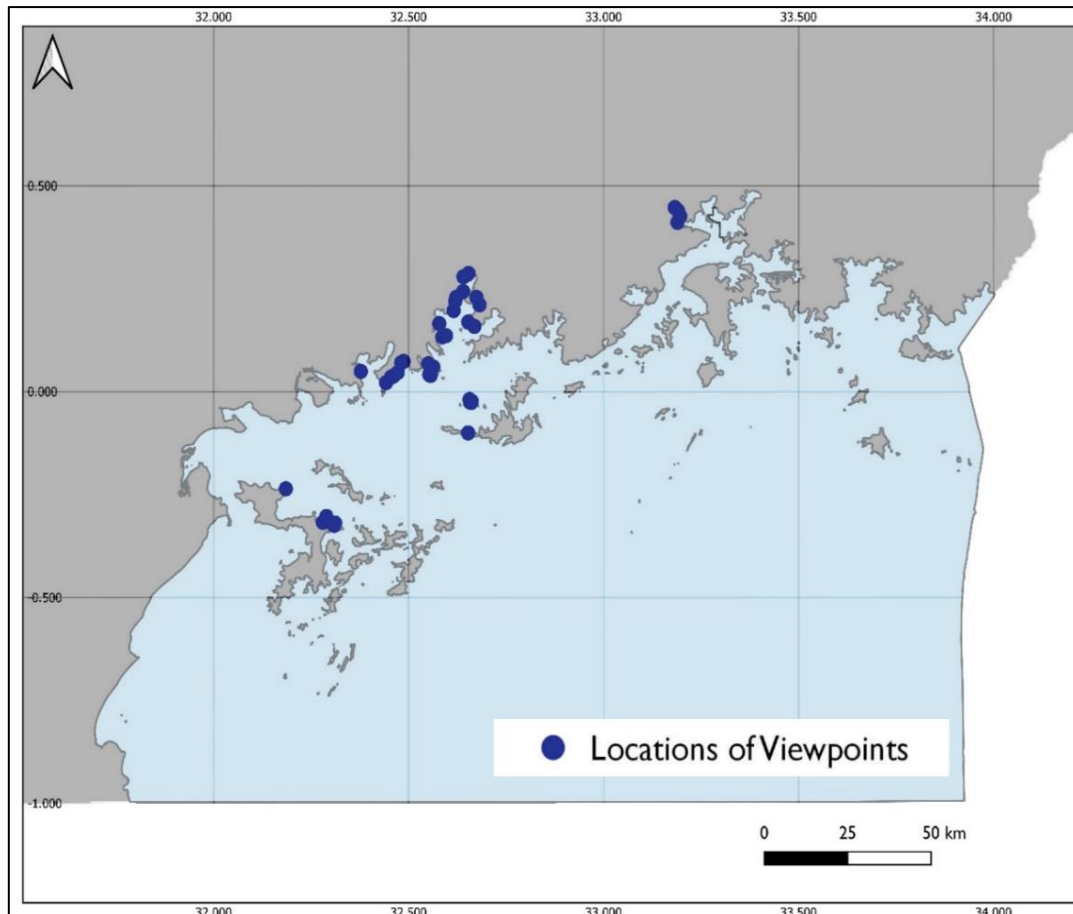


Figure 4.4: Location of viewpoints (recreation services, beaches) in the study area.

Table 4.6: Reclassification of the visual impact layer

Cage size	Membership Function	Membership Function Shape	Control points adapted from Falconer <i>et al.</i> , (2013) Proportional viewshed scale	
Small cages	Sigmoidal	Monotonically decreases	c	d
			0.2	0
Medium to Large	Sigmoidal	Monotonically decreases	1	0

4.2.4.7 Proximity to Existing fish farms

The lake has several small and medium to large cage farms that are in operation. These farms can apply for expansion of production sites if they need to. Pertaining to potential biosecurity (Pruder, 2004; Rusekwa *et al.*, 2020) and the cumulative effect of concentrating farms in an area, it was important to include this aspect as a factor in planning production zones. With a relatively low flushing rate in the lake (Hecky *et al.*, 2006), presence of other cage farms

contributes to reducing space suitable for locating new ones. Hence, it is important that consideration is given to location that are occupied, to reduce on the impact of cumulative organic loading or risks associated with disease outbreaks.

In a survey carried out in October to December 2019 (Chapter 2 of this thesis), coordinates for locations of the farms were collected (Figure 4.5). Using the distance module in Terrset, the distance from existing farms was established. In this assessment, within 2km from existing cage farms suggested by LVFO (2018), suitability for cages was set to increase beyond the 2km from existing farms.

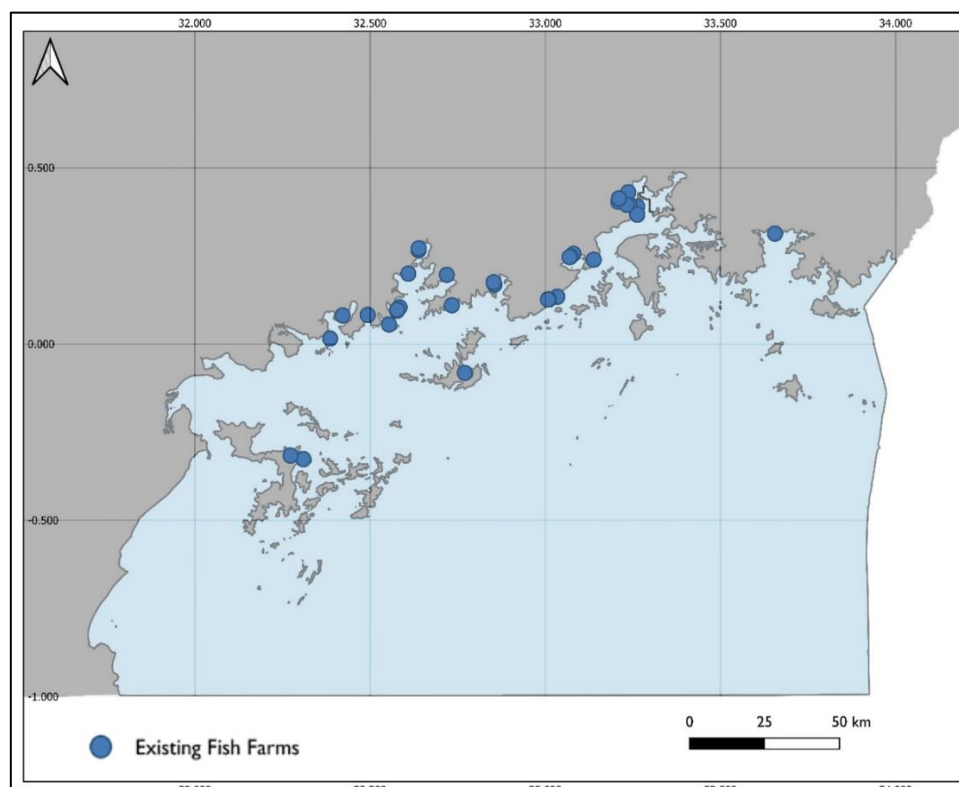


Figure 4.5: Location of existing fish farms in the study area. Several large farms are located along the channel leading towards to mouth of the river Nile.

4.2.4.8 Reclassification of the proximity to existing cage farms layer

Reclassification was done based on suitability distances suggested by LVFO (2018) (Table 4.7). This varied for small and medium to large farms due to the assumption that the larger farms were more likely to be impacted by proximity to other farms than smaller ones. The higher production volumes in larger farms and a higher possibility of expanding compared to smaller cages were assumed as determining factors of the importance of being close to

already existing fish cages. Further on, increase in biomass was assumed to increase on organic loading into the water column, posing an impact on the production carrying capacity at the site, most important at low flushing rates.

Table 4.7: Reclassification of the Proximity to existing fish farms layer

Cage size	Membership Function	Membership Function Shape	Control points adapted from LVFO, (2018) distance from other fish farms (km)	
Small	Sigmoidal	Monotonically increasing	a	b
			1	2
Medium to Large	Sigmoidal	Monotonically increasing	0.5	1

4.2.4.9 Bathymetry

Bathymetry is one of the key physical factors that must be assessed for cage installation. It has an implication on the type of cages to use, the height at which the nets are dispensed to allow for waste dispersion, and the mooring techniques (Falconer *et al.*, 2013; Aguilar-Manjarrez *et al* 2017; Francesco *et al.*, 2017). As fish swim within the cages during feeding and other activities, there is potential to agitate the bottom substrate. To allow for free water exchange, and mitigate against localised oxygen depletion, Beveridge, (2008) recommends holding the fish cages at least 4-5m above the sediments. Mooring done to hold the cages against forces due to wind and currents is another determining factor for the maximum bathymetry suitable for cage placement. This is a site-specific aspect depending on hydrodynamic conditions although Kumar and Karnatak, (2014) recommend the total length of the mooring line to be at least 3 times the maximum bathymetry at the site. Data for creation of the bathymetry layer was sourced from Hamilton *et al.*, (2016). (Figure 4.6).

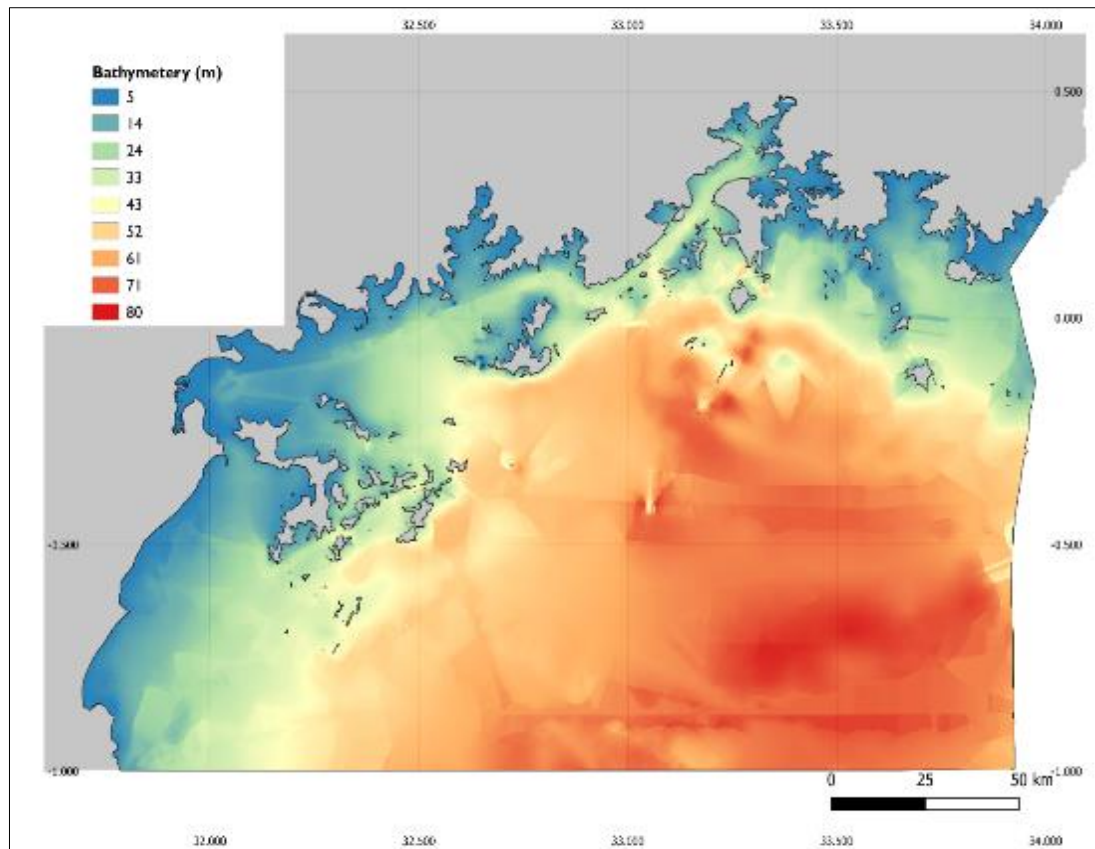


Figure 4.6: Bathymetry map of the study area. Data sourced from Hamilton et al. (2016)

4.2.4.10 Reclassification of the Bathymetry layer

Reclassification of the layer (Table 4.8) was done for small and medium-large cages based on suitability ranges provided in literature and from farmers' experience. Given the size of cages presently used, at least 3m below the cage hanging was required (Andrew pers.comm.). It was also dependent on cage mooring technologies employed. The depth at which small cages were set in Lake Victoria made them susceptible to impacts of bottom sediment agitation, especially when high densities of fish are held in the net. The reclassification (Table 4.8) was based on the most suitable ranges provided in literature rather than current practices observed in farms.

Table 4.8: Reclassification of the Bathymetry layer

Cage size	Membership Function	Membership Function Shape	Control Points based on Beveridge, (1987); Ross <i>et al.</i> , (2011); Asmah <i>et al.</i> , (2021)			
			a	b	c	d
Small	Sigmoidal	Symmetric	3	6	8	12
Medium to Large	Sigmoidal	Symmetric	9	16	24	30

4.2.5 Weighting, Analytical Hierarchy Process and Multi-Criteria Evaluation

Multi-Criteria Evaluation (MCE) (Eastman 1999) is one of the methods used in decision making where a range of factors that trade-off with each other is involved. MCE involved an assessment of various factors, evaluating attributes against each other, and aggregating them following awarded weights, to obtain a ranked output with respect to the problem or target objective. Analytical Hierarchy Process (AHP) produced weight values for each alternative based on the judged importance of one alternative with respect to a common criterion; an expression of how one factor traded off with others in MCE (Eastman, 1999). This process is based on stakeholders' priorities (Falconer *et al.*, 2018). Hence, stakeholder engagement is recommended in the weighting process, however, due to their varying needs, the results might be subjective (Nath *et al.*, 2000). In the present study, relative importance of one factor over the other was achieved through pairwise comparison facilitated in the Analytical Hierarchy Process (AHP) where alternatives were analysed and prioritised (Saaty, 1990).

4.2.6 Layer Weighting

Weighting of the layers according to relative importance in the models for small and medium to large cages was done in TerrSet software guided by expert opinion. Several iterations of pairwise comparison and resultant consistency ratios was done. The consistency ratio is a metric that indicates the consistency or reliability in a pairwise comparison. The higher the consistency ratio, the more inconsistent the output was. Saaty, (1990) recommends a consistency ratio of ≤ 0.1 . Guided by the researcher's knowledge on the study area, and an input of experts on aquaculture in the region; a researcher, a cage farm operative, and a sector regulator about which factors ranked most important for the two types of cages, several

pairwise comparison were made. The weights that gave the least consistency ratios of 0.04 and 0.07 for small and medium-large cages respectively (Table 4.9) were selected.

Table 4.9: Layer weights used in the final models for medium-large and small cages respectively.

Model Layer	Small cages	Medium to large cages
Market access	0.3844	0.1484
Proximity of Existing cage farms	0.1878	0.4269
Proximity of water hyacinth	0.0763	0.0722
Bathymetry	0.3121	0.2910
Visual Impact	0.0394	0.0615
Consistency ratios	0.04	0.07

For farms using smaller cages, access to markets was considered the most important factor while for medium to large scale farms, proximity to other farms was the most important factor. Larger farms are more likely to expand, hence, would require more space. The proximity to other farms was considered a major factor, weighting it highest in the model for farms that used medium to large cages. Accessibility challenges could force smaller farmers to sell their produce on farm, which lowered enterprise profitability (Jagger and Pender, 2001) hence, a higher weight was allocated to the market access factor in the model. Their vulnerability to post-harvest losses and an inability to hold fish in the cages longer due to feed and management expenses also contributed to the weighting.

4.2.7 Constraints

For the constraint layers, protected areas, fish breeding grounds and the lake shoreline are provided for in different national legislation or international protocols. Fish breeding grounds are provided for in the Uganda Statutory Instrument, The Fish (Fish Breeding Areas) Order, 2017, Section 35 of Fish Act, cap 197. Fish breeding zones on Lake Victoria are restricted from any fishing activities, for stock replenishment purposes. Uganda ratifies to the Ramsar convention, with 12 Ramsar sites, 5 of which are wetlands buffering Lake Victoria. In the dataset, some of these were wetlands that touched the peripheral of the lake shoreline, or bays mapped for fish nurseries. Similarly, protected wetlands have their hydrological

processes dependent on exchanges with the lake, hence, a buffer zone of 1000m was put around them to create a Boolean layer avoiding cage farming in these spaces. For navigation lines used by ferries, a 1km buffer was created over the mapped ferry lines. Vector data in form of lines shapefiles for the ferry lines was obtained from the National Roads Authority, Uganda and these were the Kiyindi-ferry, Nakiwogo and Bukakata-Lutoboka ferry.

Finally, a buffer zone of 200m was created along the lake shoreline to cater for provisions of the law. The National Environment (wetlands, riverbanks, and lake shores management) Regulations, No.3/2000 of the National Environment and Management Act of Uganda provides for a protected zone of up to 200 meters from the low water mark of lake shoreline. In the model, constraints were expressed in form of Boolean maps (Figure 4.8), with 0 coded for exclusion and 1 for inclusion.

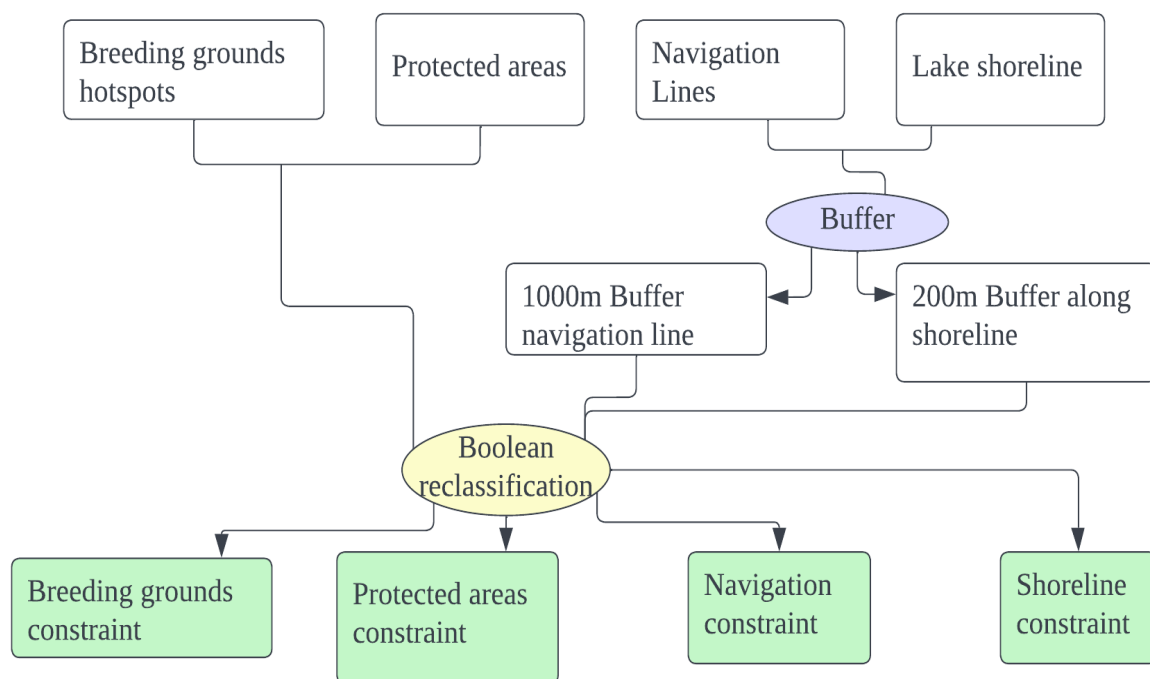


Figure 4.7: Schematic presentation of processing of data into the constraint layers

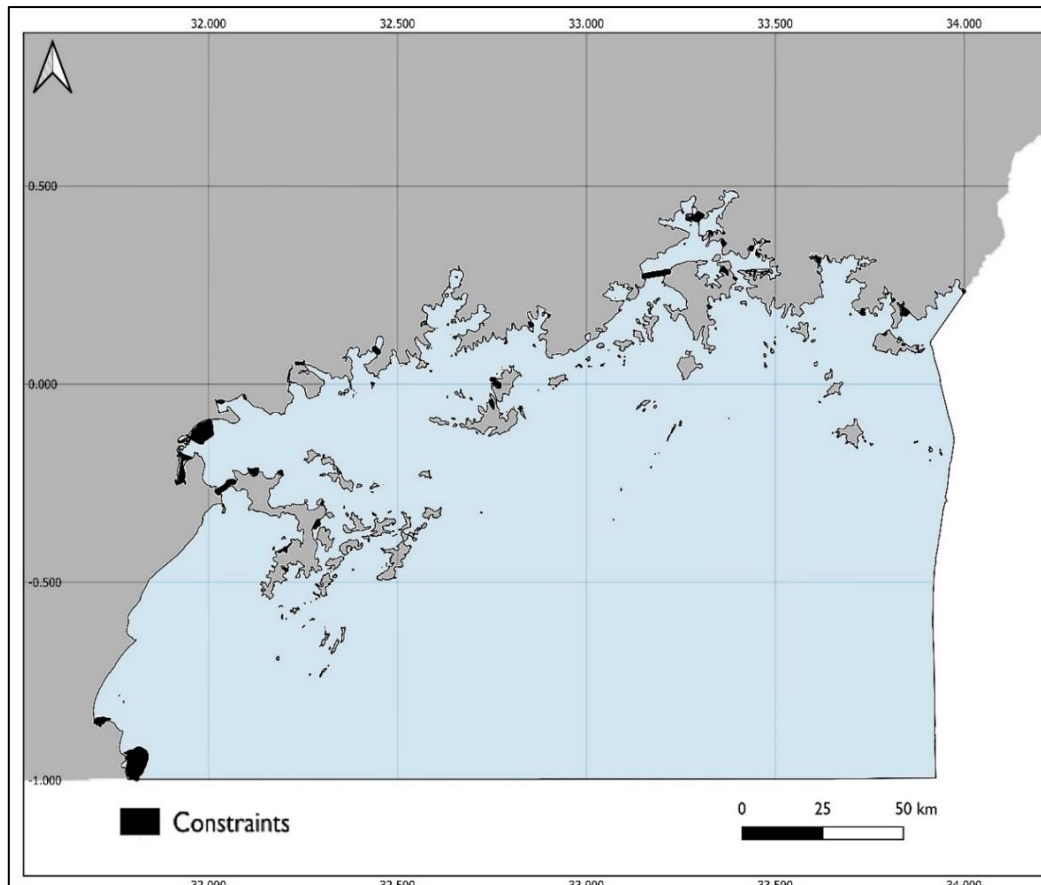


Figure 4.8: Constraints including fish breeding grounds, navigation buffers, protected areas and a shoreline buffer that is less visible due to the scale.

4.2.8 Final Model

The reclassified input factor layers and the constraint layer were combined using the Weighted Linear Combination; a simple additive technique (Al-Abadi *et al.*, 2017) , and the AHP-derived weights in MCE in Terrset. The final suitability model for small and medium-large cages produced with fuzzy classification was then reclassified into 5 equal interval classes (Table 4.10). This reclassification of the fuzzy output was meant to ease interpretation of the final models (Falconer *et al.*, 2018).

Table 4.10: Interpretation of fuzzy classes using 5 intervals and corresponding meanings.

Interval	Category	Interpretation
0-0.01	Constraint	No aquaculture can happen here
0.01-0.25	Very low suitability	Area mostly unsuitable for cage farming
0.25-0.50	Low suitability	The areas have low suitability for cage farming
0.50-0.75	Moderate suitability	These are suitable for cage culture
0.75-1	High suitability	Area is most suitable scoring relatively high on all factors in the assessment

4.3 Results

4.3.1 Bathymetry

The bathymetry reclassified on a scale of 0 to 1 (Figure 4.9) indicate that only 1.88% and 5.55% lie between 0.25-0.5 (low) suitability for small and medium-large cages respectively. Similarly, only 5.6% and 16.6% are above 0.5 (moderate to high) suitability score for small and medium to large cages. The largest portion of the study area i.e., 92.5% and 77.9% are within depth limits not suitable for small and medium-large cages. However, as technology improves such as the use of better mooring technologies, cages that could be used in deeper waters would provide opportunity for expansion.

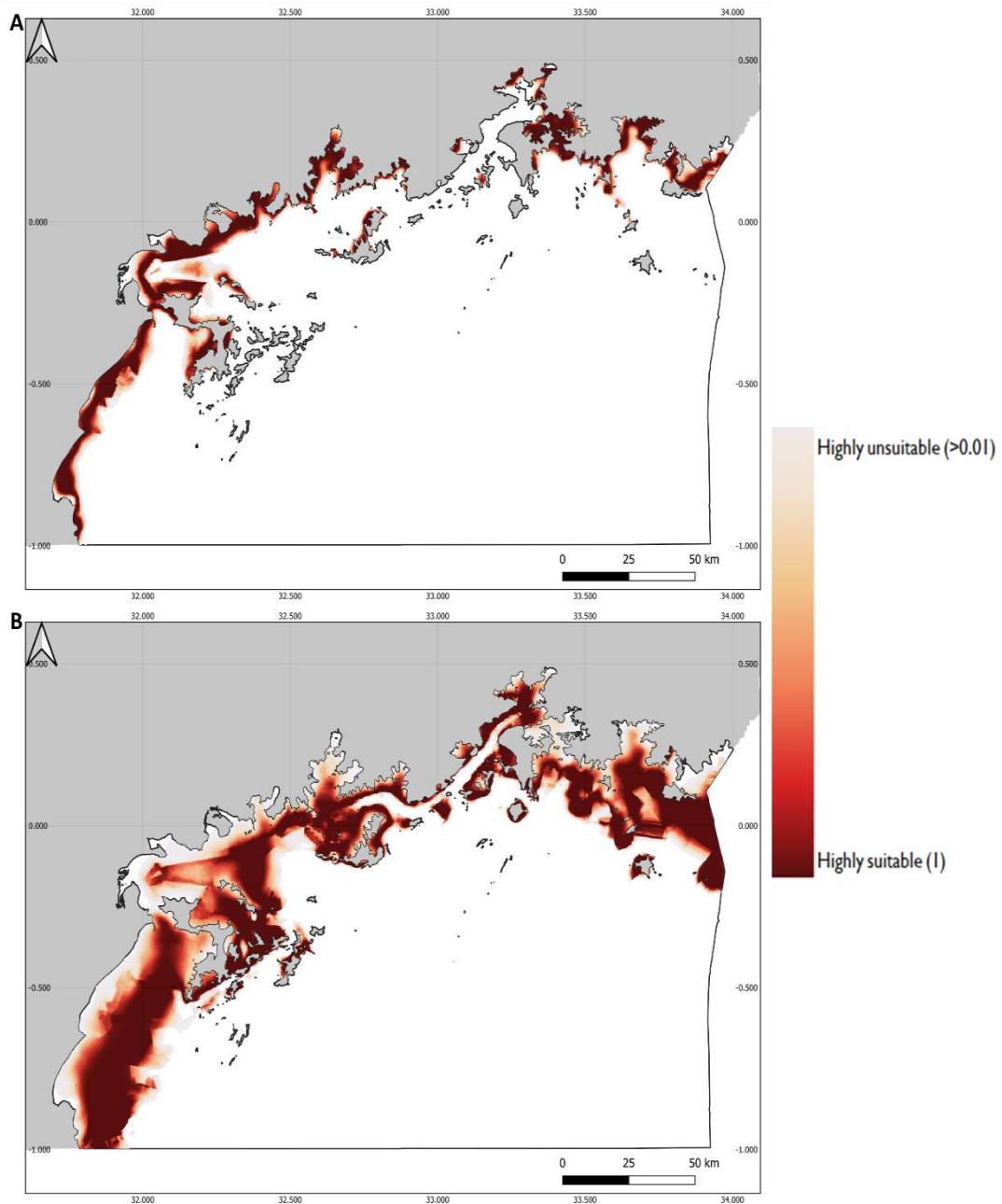


Figure 4.9: Bathymetry fuzzy classification (A-Small cages, B-Medium-large cages)

4.3.2 Water Hyacinth

Proximity to the water hyacinth weed (Figure 4.10.) made the study area less suitable by 0.5% and 0.3% for medium-large and small cages respectively. In both cases, the impact was minimal, higher for the smaller cages than the medium and large ones. The water hyacinth being a direct indicator for nutrient loading, with the enclosed and semi-enclosed bays closer to cities and towns receiving the bulk of the catchment pollution. Secondly, the water hyacinth

is a light floating weed that would attach onto the lake shore vegetation or form a mat in sheltered areas. In the open waters, it is broken by the wind due to lack of substrate to hold. This meant that areas that would have been suitable for cage farming were reduced with small cages being more susceptible than the medium-large ones which could be deployed in deeper waters.

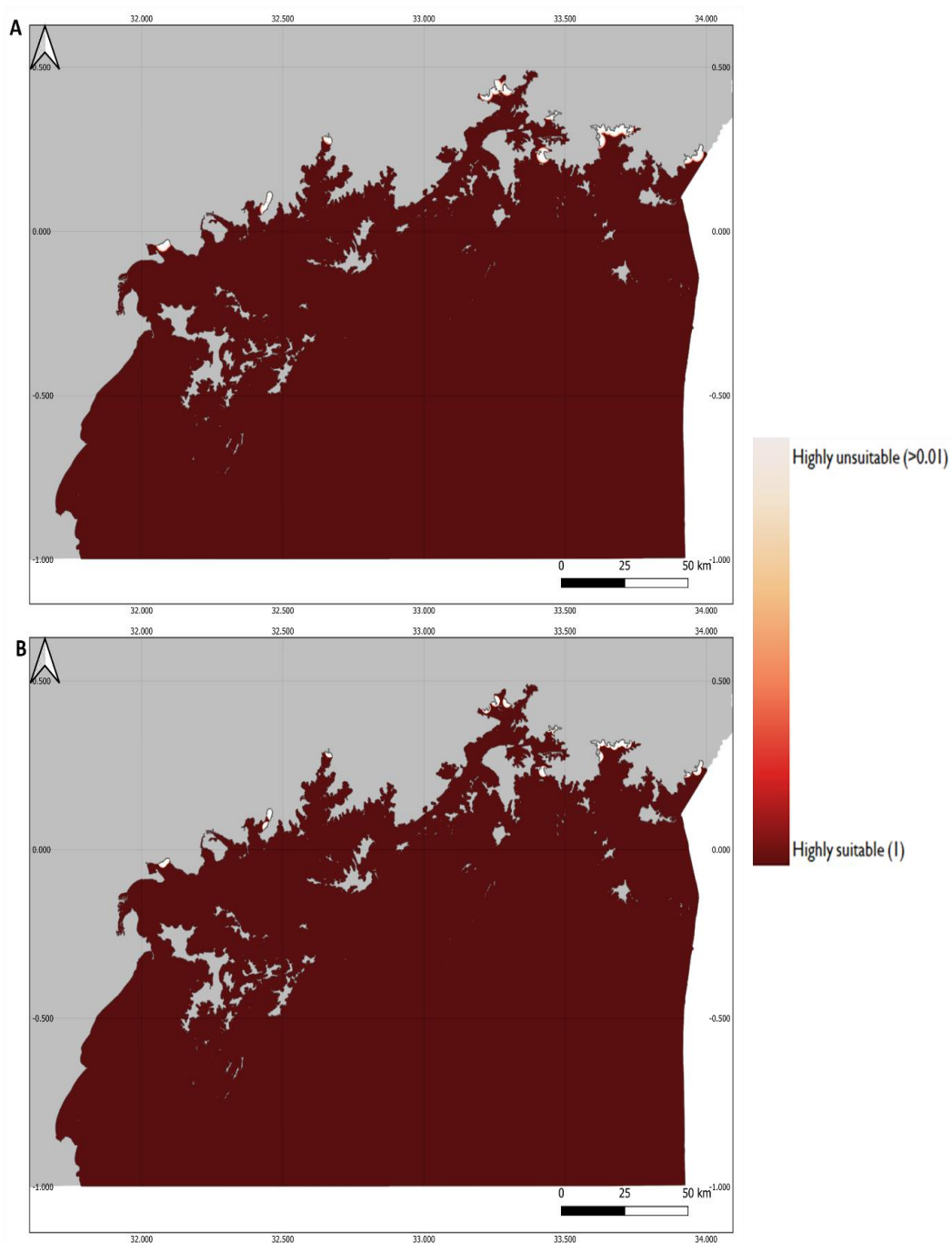


Figure 4.10: Fuzzy classification based on proximity to water hyacinth hotspots (A-small scale cages and B-Medium to large scale)

4.3.3 Market access

For access to markets (Figure 4.11), 0.7% and 1.5% of the study area had suitability over 0.5 for small and medium-large cages respectively. The remaining 99.3% and 98.5% were not accessible for small and medium to large cages respectively. Highest around major towns, large farms have a higher capacity to manoeuvre seasonal roads or utilise better vessels on water which could explain the higher suitability compared to small farms.

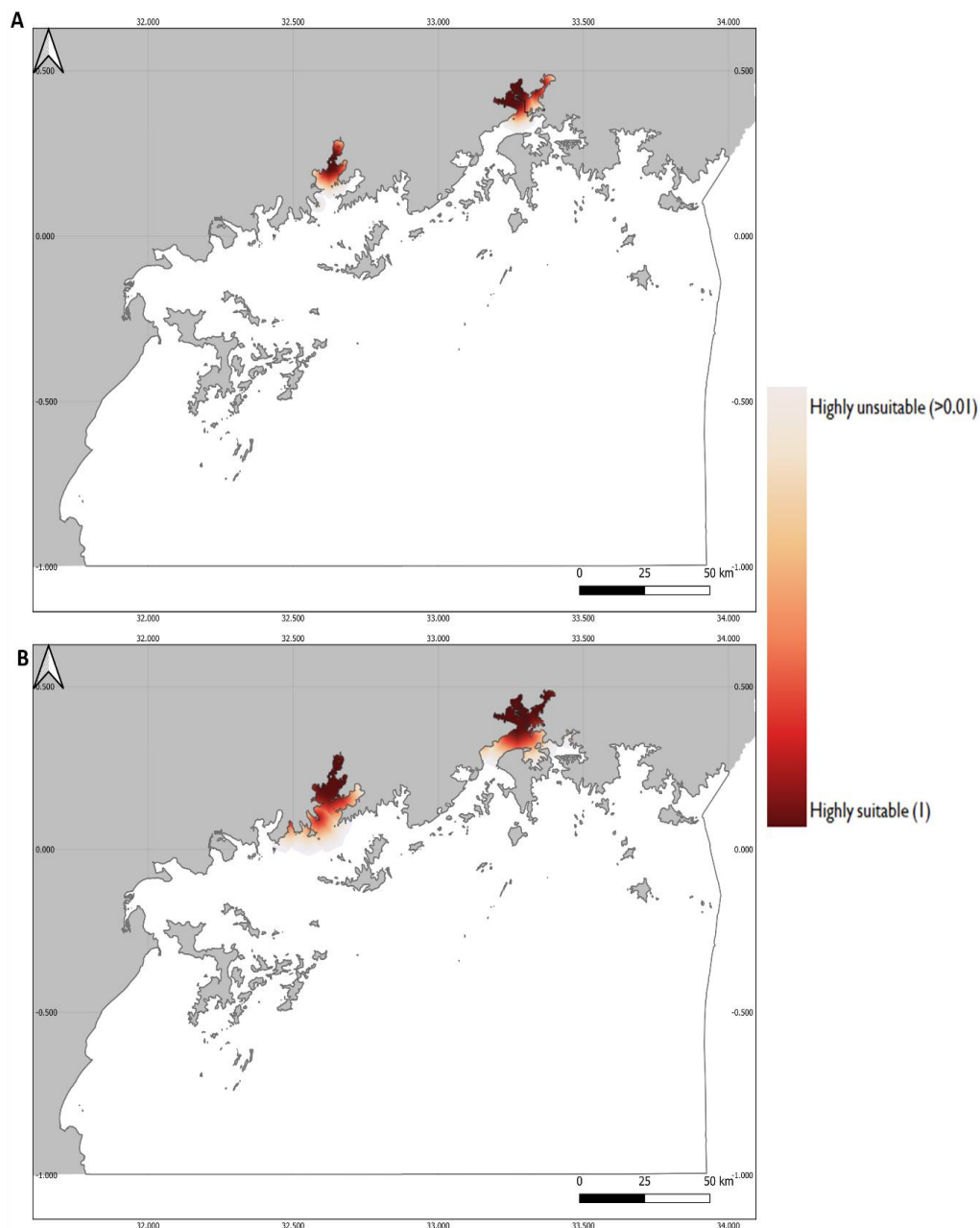


Figure 4.11: Fuzzy classification of the access to markets layer (A-small scale cages and B-Medium to large scale)

4.3.4 Viewshed

The viewshed layer yielded results for small cages that were closely like those for medium to large cages (Figure 4.12.). One percent of the study area would be visually impacted by presence of medium-large cages while 99% would not. For small cages, 0.03% of the area would be visually impacted by presence of small cages while the remaining 99.9% would not have any scenery impact. Presence of the cages, regardless of the size almost had a similar output, with places closer to the capital exhibiting worst suitability scores compared to the rest of the lake. These are also the parts of the catchment occupied by the affluent who care about the scenery and similar aesthetic services.

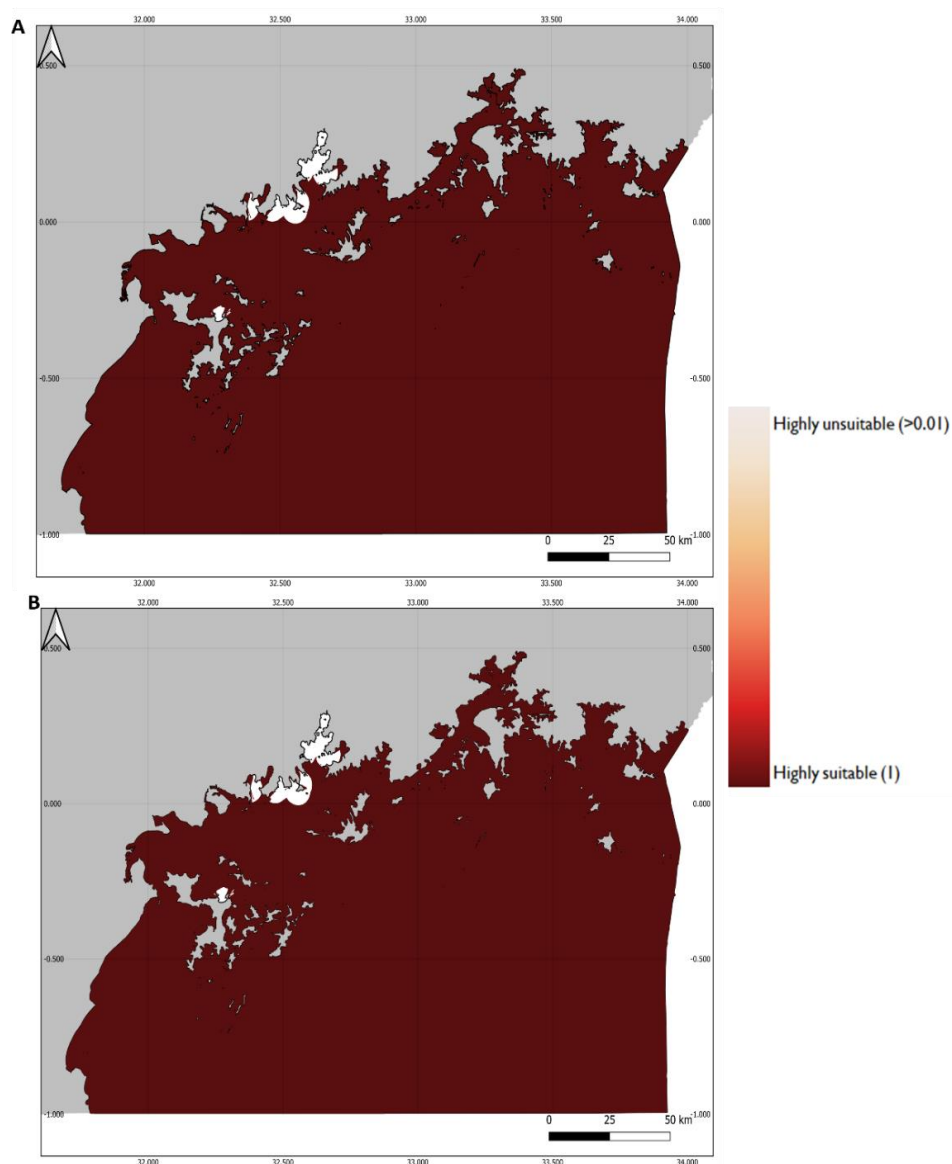


Figure 4.12: Viewshed fuzzy reclassification for A-small and B-medium to large cages

4.3.5 Existing fish farms

The fuzzy reclassification of the factor; proximity to existing fish farms (Figure 4.13) when interpreted on a scale of 0 to 1 showed over 99.7% of the study had a suitability over 0.5 for small cages and 98.9% for medium to large cages. Most impacted for large cage suitability were sites along the channels leading towards the river mouth. On the contrary, when using small cages, the influence from other farms was assumed minimal, hence a slightly higher suitability based on the proximity to existing farms.

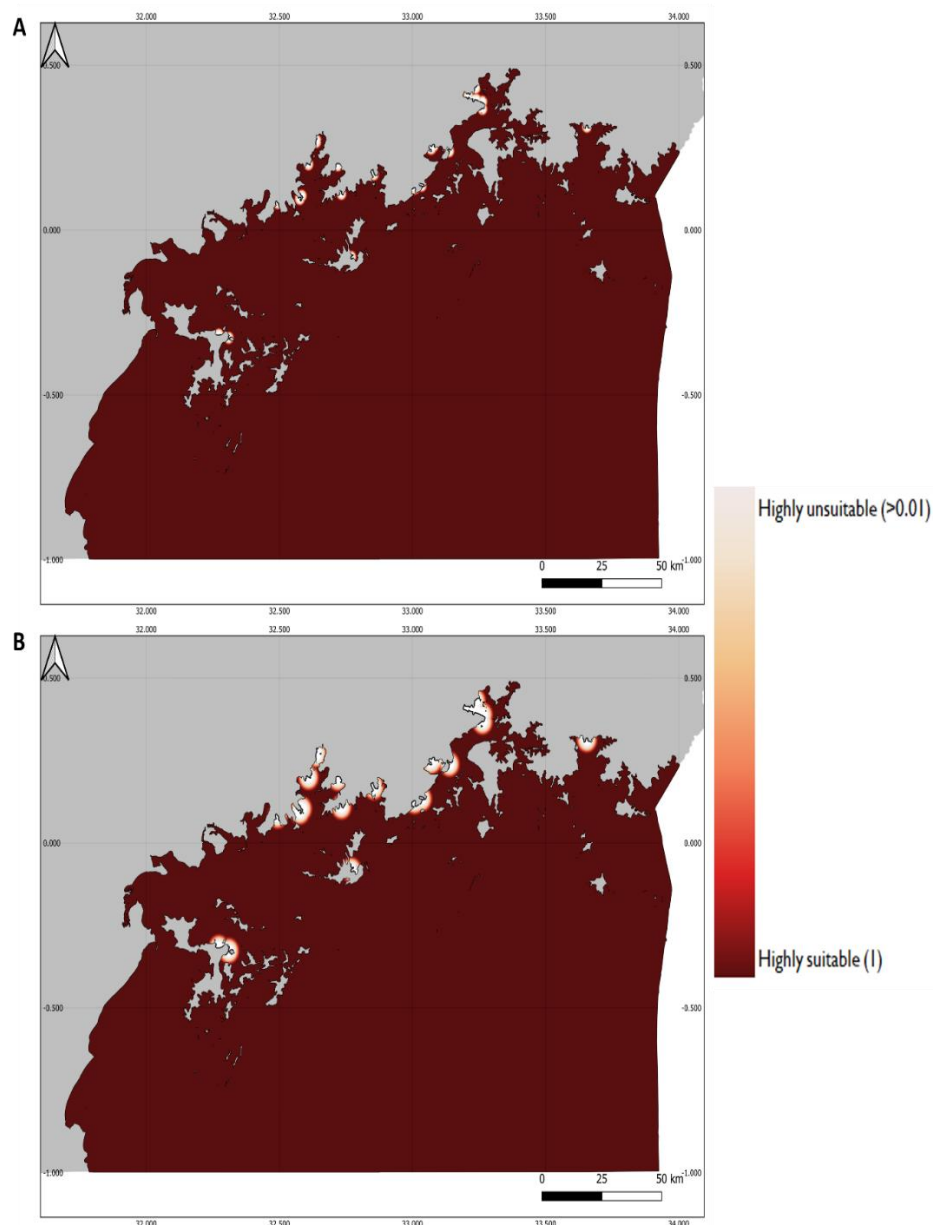


Figure 4.13: Existing Farms fuzzy classification for A-small and B-medium to large cages

Overall, the final model results revealed that the potential for cage farming (Table 4.11) was higher for medium to large cages (Figure 4.14) than for smaller cages (Figure 4.15). The two input layers against which using both small or medium-large cages performed poorest were bathymetry and access to markets. For the remaining input layers, i.e., viewpoint, proximity to existing farms, and proximity to water hyacinth hotspots, suitability was generally high. The final model for each type of cage category is dependent on the weights assigned to the layers. The inshore of the far western part of the lake shows a strip of seascape with moderate suitability (Figure 4.14), stretching towards the capital city where small cage suitability is highest. High suitability is also seen in parts of the gulf close to the source of the river Nile, the outlet draining the lake. Patches of moderate to low suitability are seen in the east.

For the large cages (Figure 4.15), very low suitability is observed around the capital city and in areas that are close to other farms. From the far west toward the central part of the lake, up to the far east is an area with high and moderately high suitability. Patches of low suitability can be seen in some sections, more to the west. On the offshore, suitability tends to be low for medium to large cages.

Table 4.11: Overall potential for aquaculture in the Uganda section of the lake

Potential for cage culture in final model output	Medium-large cages (km ²)	% of the study area	Small cages (km ²)	% of the study area
Very low	119,215,800	0.4	25,509,216	90.3
Low	24,279,759,500	85.9	797,939	2.8
Moderate	2,399,584,500	8.5	800,264	2.8
High	346,451,800	1.2	57,915	0.2

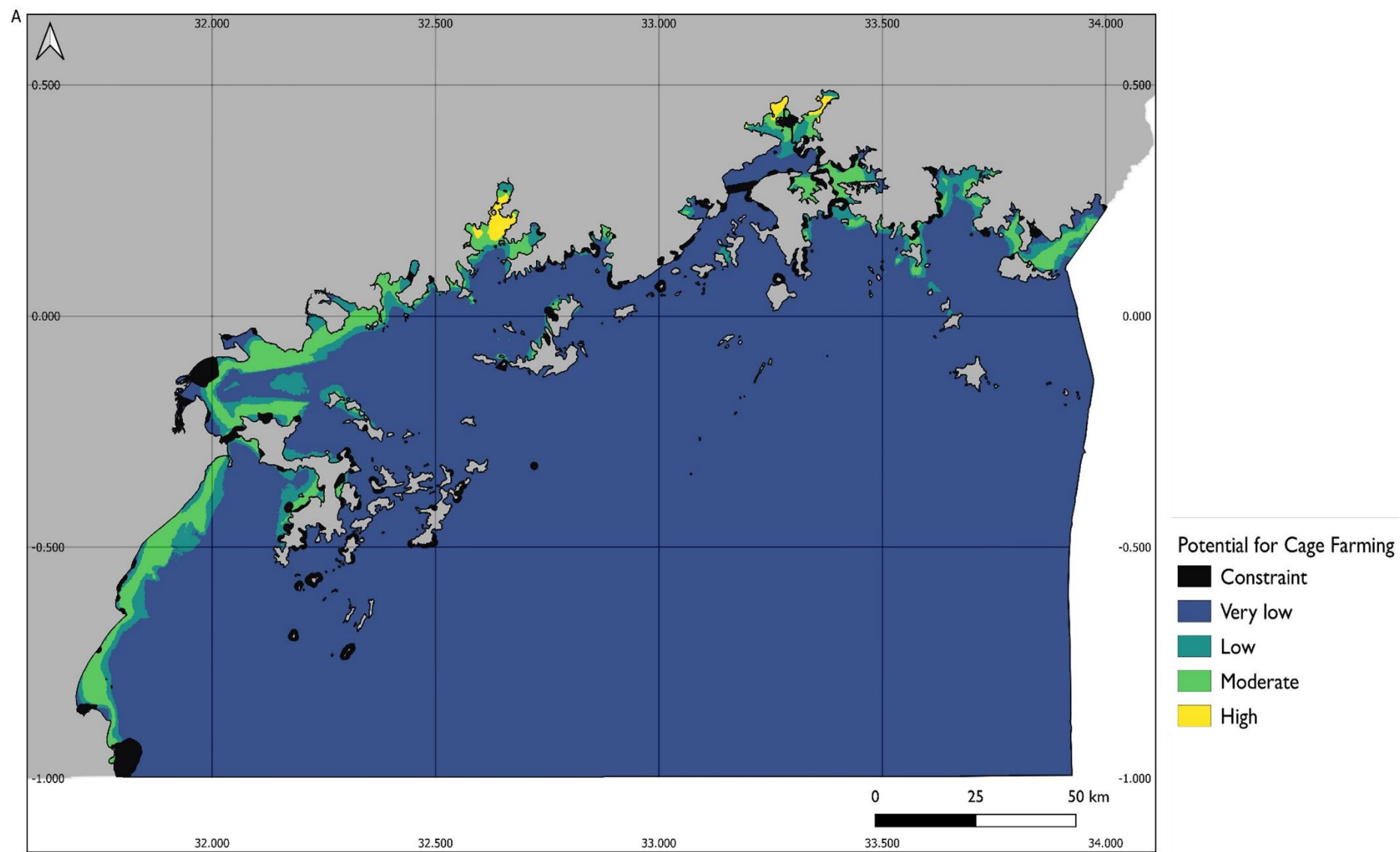


Figure 4.14: Potential for cage culture using small-sized cages between (1.5x1.5x3) m to (3x3x3) m) common in small cage farms.

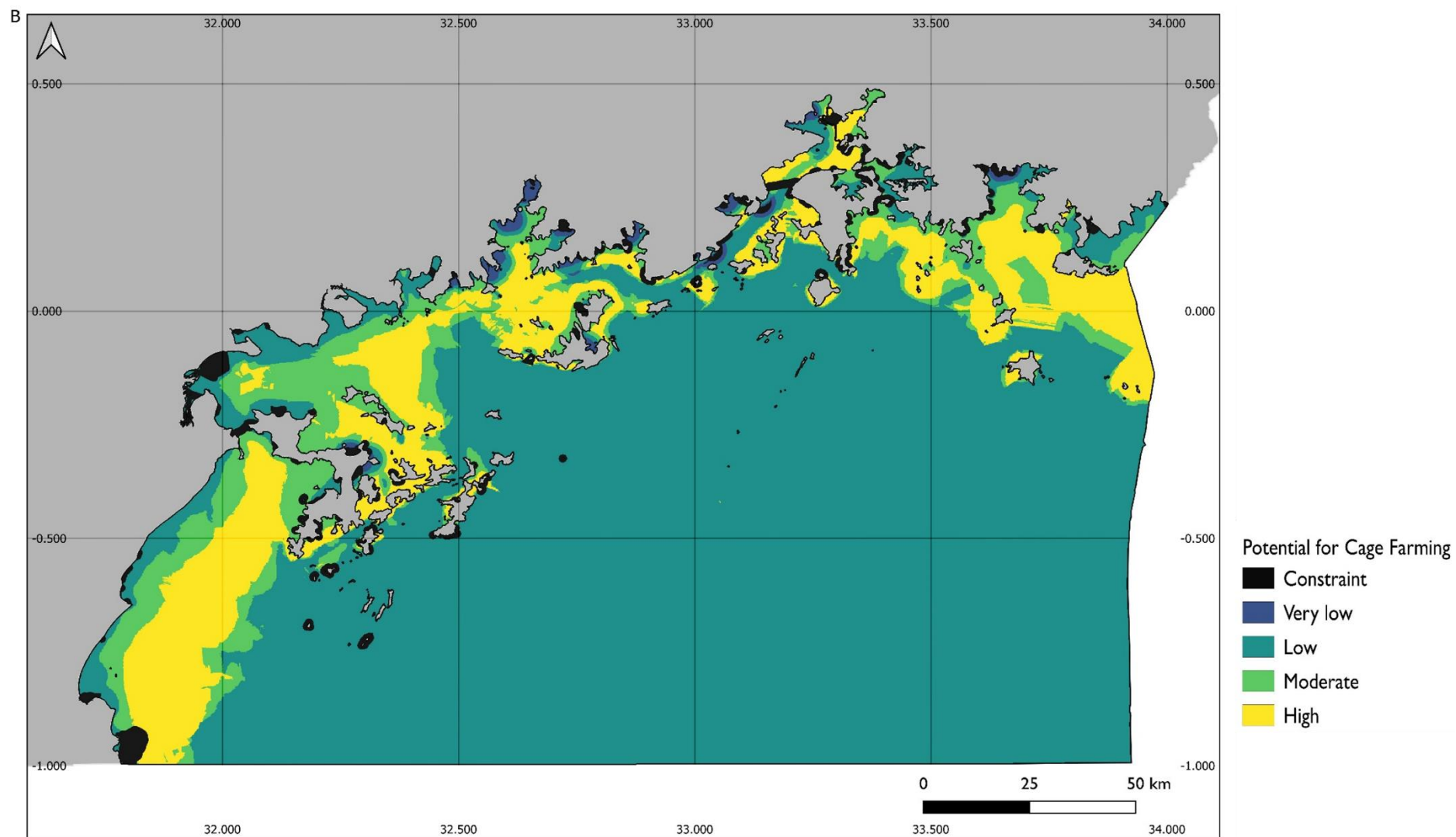


Figure 4.15: Potential for medium-large cage farms with cage size ranges between (6x6x6) m to (20-30m diameter x 6m depth)

4.4 Discussion

In the present assessment, GIS software and spatial modelling was used to identify parts of the lake suitable for Tilapia cage farming. Cage farming on Lake Victoria is a recent development done commercially with a range of cage sizes, depending on the activity. Hence, this assessment was done for both small and medium-large cages. Consideration was made of the location of farms already in existence, proximity to markets via different road types, water depth, viewshed and water hyacinth hotspots. With the available data, these factors were combined in a multicriteria evaluation that provided for a pairwise comparison accounting for the relative importance of each factor. This is an incredibly significant aspect, where multiple criteria, different interests and perspectives are part of the decision-making process (Liang *et al.*, 2017). The factors included in the assessment were based on the availability of sufficiently high-quality data, which was a major constraint common in similar spatial assessments (Falconer *et al.*, 2020). Fuzzy set theory (Eastman, 2016) was used for layer reclassification to address the lack of values on factor thresholds, uncertainty, and vagueness.

The study revealed that only 9.7% of the Ugandan part of the lake had between moderate to high suitability for cage farming using medium-large cages. A larger section, 86% was found to have low potential suitability, for Tilapia fish farming. This section lies mostly in the exposed parts of the lake, where winds and access to shoreline activities would play a key role in determining overall suitability for sites. One advantage is that the exposed areas are further from the impacts of catchment pollution common in bays and gulfs (Kabenge, Wang and Li, 2016; Akurut *et al.* 2017; Olokotum *et al.*, 2021) and could be explored as technologies for offshore cage farming become more available. Similarly, only 3% of the study area was found to have moderate to high potential for Tilapia cage farming using smaller cages. However, this level of potential was attained closer to the large cities, a challenge that might further affect suitability when water quality is factored into the model. Littoral zones of Lake Victoria are heavily laden with nutrients due to intensive catchment degradation, poorly managed point-sources of pollution like abattoirs and sewerage works and the natural erosion of the carbonatite rocks that form the lakebed (Egessa *et al.*, 2020; Guya, 2020; Onyango and Opiyo, 2021) reducing suitable space.

Environmental factors like wind speed and water quality are crucial to be included in suitability assessments for fish farming. However, challenges with obtaining data of suitable resolution limited inclusion of these factors in the model. A water quality proxy for proximity to water hyacinth hotspots was included. With its proliferation linked to poor water quality due to high nutrient concentrations, the water hyacinth has been reported to return in some bays and gulfs

(Otieno *et al.*, 2022) on Lake Victoria. This poses a major threat to cage aquaculture on the lake. The weed floats, forming dense colonies that block light penetration into the water column. The hyacinth is a mobile weed, whose growth rate is dependent on air temperature, plant density and nutrient content (N and P) (Imaoka and Teranishi, 1988), doubling in population in 5-15 days (Mitchell, 1978). High nutrient levels, a major precursor to weed multiplication, have been reported in most parts of the lake (Njagi *et al.*, 2022). Even when the present assessment showed that the widest part of the lake scored highly against the weed factor, infestation is mostly along lake shores due to relatively high nutrient levels and the presence of attachment facilities. This would significantly reduce the space available for small cages likely to be in shallow waters. Monitoring of the weed needs to be more effective. To protect the environment and aquaculture enterprises themselves, hotspots that are already challenged by high nutrient levels should be avoided.

Accessibility to rural areas improves marketability and overall agricultural productivity and employment in the sector (Valdés *et al.*, 2009). Transportation of inputs such as feed, seed, or fish to markets requires proper infrastructure. Within the region, access to remote areas is characterised by poor connectivity to all-season roads, and mostly feeder roads heavily impacted by rains (Banjo *et al.* 2012; Ngezahayo *et al.* 2019). This translates into increased cost of production. Due to the perishability of fish and most other agricultural products, in some cases, farmers are forced to sell their produce at farm-gate, attracting less profits (Hine, 2014). The present assessment revealed how access to markets was a major challenge when using either small or medium to large cages for farming. Only 0.68% and 1.52% of the study area was between moderately and highly accessible. This highlights the need to improve accessibility to markets through improved road networks, leveraging technology (Thompson and Gyatso, 2020) and quality & reliable extension services (Sebaggala and Matovu, 2020).

Aquaculture needs to grow along with other sectors of the economy. Other aspects of the model, such as viewsheds for recreation services and leisure centres, were included, though with less impact; 99% and 98.9% were highly suitable for small and large cages respectively. However, only a few large recreation areas and lakeside residences were included. In Chapter 3 of this thesis, it has been demonstrated that social issues, such as aesthetics, viewshed may not be of much significance now. However, as more farms are setup on the lake, conflicts caused by stalls that interfere with the view may become more apparent. Hence, assessment models for strategic planning should include factors or constraints that are present or likely to present as farming expands on the lake.

4.5 Conclusion

Using spatial modelling tools, this study examines cage farming suitability on the lake. The tool combined criteria for and against cage farming. Validation of the outputs against current locations of large cage farms showed consistency with our results. The findings of the study give a preliminary indication of where further assessment of site-specific conditions for setting up cage farms could be done. Further assessment needs to incorporate environmental factors such as wind speed and water quality as data becomes more available at a suitable resolution. It is imperative to assess seasonal variations in suitability in the identified sections of the lake. This is because critical variables such as wind speed and water quality change with the seasons. The growth of cage farming and competition for inshore space will necessitate exploration of deeper sites. Decisions will be guided by cost-benefit analyses of increasing site production capacity versus the cost of doing business. Further research into oxygen dynamics in the water column at periods of deep mixing which mainly affects deep water (>20m) will be necessary.

Similar assessments could be done in the other African Great Lake systems using spatial data that is available in open repositories. This will enable us to determine the exact extent to which cage farming is feasible. It is only then that plausible production targets can be set rather than relying on assumptions based on the presence of a water body as the only criterion. Such spatial planning tools will be very crucial for objectively guiding resource allocation and putting aquaculture in areas that would have the least impact on production and business profitability.

CHAPTER 5

TRADEOFF ANALYSIS BETWEEN CAGE FARMING AND OTHER ECOSYSTEM SERVICES

5.1 Introduction

Ecosystem Services (ES) are defined as contributors to anthropogenic value (Atkinson *et al.*, 2012). Matching ES supply and demand is imperative to ensure the sustainability of social and natural capital (Burkhard *et al.*, 2012). Supply of ES is defined by the physical or ecological processes that lead to its existence (Schägnier *et al.*, 2013) whereas the demand is determined by human interactions that are direct or indirect. These human interactions are drivers of change or aggravating factors, and examples include population, social-cultural factors, technological advancements, or economic interaction with the ES (Leemans and de Groot, 2003; Brugere, Lymer and Bartley, 2015). Broadly, ES have been classified as provisioning, regulating (or supporting) or socio-cultural (Millennium ecosystem assessment, 2005). The direct-use value of provisioning ES contributes to socio-economic transformation while regulating ES contributes to ecosystem health. Pan *et al.*, (2021) define ecosystem health as the ability of an ecosystem to sustainably deliver ES. An ever-increasing demand amidst degradation pressures is one of the major challenges facing ecosystems globally. Meeting multiple socio-economic and ecological objectives requires flexible resource planning (Geneletti and van Duren, 2008).

Lake Victoria provides ES reaching millions of people across the region. A few examples of provisioning ES with immediate economic value (direct uses) include fisheries, domestic and agricultural water, cage farming space, hydroelectric power generation, and transport routes across the region (El-Noshokaty, 2017). The fisheries alone were estimated to have a beach value of 840 million USD in 2014 (LVFO, 2016), bringing in foreign exchange and providing employment for over 3 million people (Kimani *et al.*, 2018). Immense pressure from anthropogenic activities and changes in land use have threatened the ecosystem health and sustenance of some of these ES (Olokotum *et al.*, 2020).

The multiple uses of Lake Victoria's resources are spread among stakeholders, and management is fragmented among sectors. In Uganda, each sector runs on its own mandate and has varying objectives and targets while utilizing the same space. Examples of current management arrangements include the need for a license from responsible authorities for

fishing, or fish transportation (Nunan, 2018); clearance for installation of cage facilities (LVFO, 2018) and a license for transport vessels (EAC, 2008). Navigation in the sheltered bays is by small wooden boats that are manually operated, or in larger wooden boats powered with outboard engines if in open waters. Fish breeding areas are designated as no-fishing zones, provided for in the Fish (Fish Permanent Closed Areas) Rules of 2010 and operationalized by the guidelines for management of fish breeding areas in Uganda (MAAIF, 2017a). Recreation services are licensed under the Uganda Tourism Act (2008) (GoU, 2008) done by the Uganda Tourism Board while ports and harbours are provided for under the Transport Act (EAC, 2008). This shows how fragmented the process is.

Cage farming is expected to expand as riparian governments provide incentives for investment (Orinda *et al.*, 2021). Exploitation rates for the fisheries have been reported as 50% higher than annual production of most commercial species (Natugonza *et al.*, 2022) prompting authorities to seek options like mapping and protecting fish breeding and nursery grounds (MAAIF, 2017). To ensure equitable, efficient, and integrated management, and to maintain the ecological integrity of the lake, holistic planning is required. It will require assessing synergies, tradeoffs, and testing different management strategies. Wider ecosystem interactions, economic and societal benefits need to be considered using an Ecosystem-Based Management (EBM) approach.

Within EBM, provision of ES is concurrently considered alongside biodiversity protection and sustaining ecosystem health (Domisch *et al.*, 2019). One limitation to EBM is a lack of universally agreed tools for the implementation of holistic planning. There is a need for exploration of available decision support tools and how these could be adapted to meet multiple needs. One such approach to EBM when multiple uses are involved is the creation of Management Areas (MA) (zones or networks) dedicated to specific management interventions i.e., multi-use schemes (Snäll *et al.*, 2016; Vallecillo *et al.*, 2018; Domisch *et al.*, 2019). The objective is to optimize space and ensure all different uses of the same area are effectively catered for. Construction of multi-use schemes has been done using various software like C-Plan (Pressey *et al.*, 2009), Zonation (Lehtomäki and Moilanen, 2013) and Marxan (Ball, Possingham and Watts, 2009). As spatial allocation tools evolve, the goal of optimisation or cost effectiveness in planning is added, particularly where competing objectives may be involved (Watts *et al.*, 2009; Serra-Sogas *et al.*, 2020). With Marxan with Zones (Watts *et al.*, 2009) (a simulation software extension of Marxan), the planning process can incorporate varying management actions. With an optimisation algorithm, this widely utilized planning software (Janßen *et al.*, 2019) allows for inclusion of costs, with the goal of minimizing them to achieve targets on competing ES objectives, with an aim of providing near optimal solutions

(Watts *et al.*, 2009). Penalties for not meeting the targets are included, as is a definition of how connected the MAs need to be (Ball *et al.*, 2009; Watts *et al.*, 2009). However, to date, limited application of such optimization tools has taken place in freshwater systems (Abell *et al.* 2007; Heilpern 2015; Langhans *et al.* 2019) compared to what is seen in the terrestrial and marine realms.

In this study, aquaculture is assessed in a framework with other ESs. Synergies and trade-offs necessary to meet targets for other uses are evaluated. The ESs included are spatially explicit and have a potential to conflict with cage farming. An indicator of ecosystem health, water quality, is added as a cost to evaluate the level of trade-off that needs to be made if benefits across ESs were to be maximized. This study builds on scholarly work that demonstrates the potential for application of spatial prioritization in large freshwater bodies and its ability to contribute to EBM in aquaculture development. The aim of the study is accomplished by using MarZones software that allows for designing management areas (zones) within the same space and assessing multiple ecosystem services holistically.

5.2 Methodology

5.2.1 Decision support framework

For running the spatial optimisation, MarZones software was used together with QGIS (QGIS Development Team, 2020). MarZones is a stand-alone open access software that utilised files prepared externally such as text files (.txt, .dat) generated with text editors like Windows Notepad while others were prepared in QGIS. Some of the technical terms used in designing or operating MarZones and their meanings (Table 5.1) and subsequent steps followed in implementation are explained in sections 5.2.2 to 5. 2.4

First, the study area was divided into smaller Planning Units (PU) in which the ES were spatially contained. When the algorithm was run, PUs were selected to contribute or not contribute to the solution in each run. PUs were assigned to Management Areas (MAs) types, at the least cost to achieve ES targets with minimal trade-offs at the end of the run. ESs were allowed or restricted from belonging to a specific MA type to manage synergies and trade-offs. Every MA type was assigned a proportion of the ES targeted from it. ES target failure and MA type contribution failure together resulted in the penalty.

Using the objective function *Equation 5.1* , a combination of the cost of selection of an area to contribute towards the ES target, a penalty for not meeting that ES's target and a cost attached

to the configuration of the planning units in any zone was minimised. The aim was to produce an optimal solution in which the PUs were allocated to the cheapest alignment that minimized conflicts and trade-offs in that solution.

Table 5.1: Key terms used in MarZones and definition of some of input files that must be kept in the same folder as the executable and. input file that specific locations of the inputs and outputs from the model.

Planning Unit (PU)	Part of the study extent created by dividing the area into smaller hexagonal grids. Each PU contained the spatial distribution of the ES over which planning occurred
Cost	This was the value that had to be minimised in the final solution. Each PU could have one or more costs. In this study, it was the opportunity cost of replacing an ES with another, the socioeconomic cost of not meeting that target of an ES or a cost attributed to the ecosystem health. The area occupied by an ES was calculated in QGIS and where actual socioeconomic costs were available, these too were used. When multiple costs were used, they were standardised to a uniform scale.
Objective function	This was the mathematical equation that the model used to calculate the solution by minimising costs and generating a compact configuration of the PUs
Zone / management network / management area (MA) type	This was defined by the management framework e.g., by allowing or restricting activities. The zone or management area type was a spatial representation of the management choice.
Ecosystem Service target/ Feature target	This was the amount of the ES aimed to be included into one or more MA type and missing this target came with a cost
Optimal solution	Binary output at the end of the run where each PU was allocated to a specific zone and this allocation changed in each run till the best solution (least costly) was attained
Penalty factor	This was the value added to the objective function for ES target not met
Boundary Length Modifier (BML)	Was the perimeter that the model used to weight the importance of placing MA types in proximity to each other. If a higher BLM was given between one MA type to another, the algorithm attempted to reduce how close the MA types were to each other and vice versa.
Selection frequency	The number of times a PU was assigned to a MA type across runs.

$$\text{Objective function} = PU \text{ cost} + (B \text{ cost} \times BLM) + (ESP \text{ cost} \times ESP \text{ Factor}) \quad \text{Equation 5.1}$$

Where *PU* is the Planning Unit, *B* is Boundary, *BLM* is Boundary Length Modifier, *ESP* is the penalty for not meeting ES while *ESP factor* is a multiplier factor of the penalty.

MarZones used this objective function to compare the solutions of each planning unit, ending up with an optimal output representing the cheapest configuration of PUs (with least trade-off) towards solving the problem. As seen in Figure 5.1, three steps were involved in the assessment i.e., preparation of the spatial layers for the ES and preparation of other input files, running the scenarios with different management priorities (targets); and examining the outputs.

The MarZones output included two files: one showing that the ES targets had been met, and another showing how the PUs were spatially arranged in the final product. In the final evaluation, if the desired result was not achieved i.e., if ES targets were not met, more iterations were done, or changes were made to the targets set for other ES (by increasing or decreasing trade-offs), or the management area type (MA) contribution to achievement of the target was adjusted (co-benefits, synergies) to ensure maximum performance across all ES. The final stage involved visualising results to identify the spatial arrangement of the areas created and where different ES were placed. When the structure of the solution was not satisfactory for the decision support, MA type connectivity values were adjusted to allow for a better spatial arrangement.

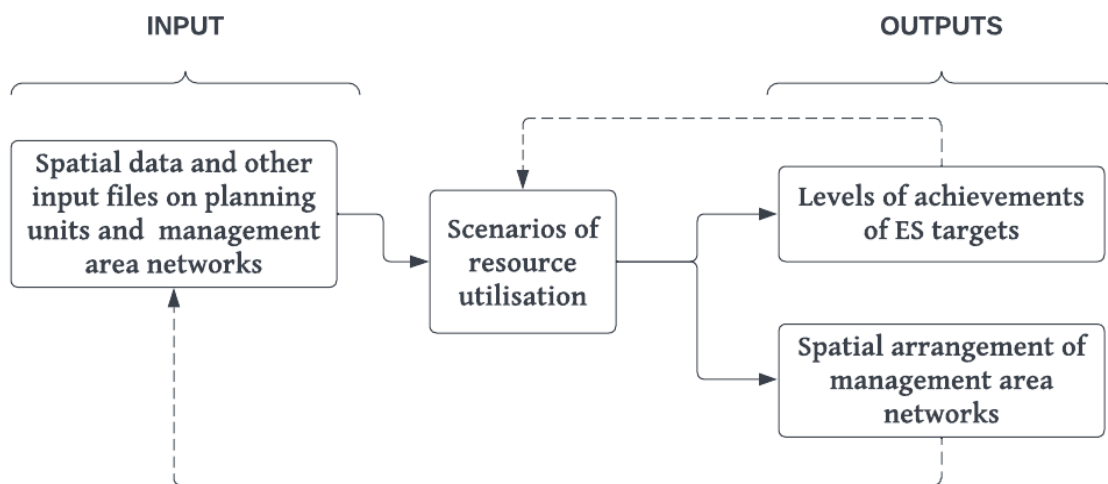


Figure 5.1: Summary of steps adapted for implementation of MarZones (based on Serra-Sogas et al., (2020). Back dotted lines represent actions in case the output was not optimal or acceptable.

5.2.2 Data management

Data management was done in QGIS (QGIS Development Team, 2020) while preparation of input files was done with QMarxan v2.0.1 (Apropos Information Systems Inc, no date) and a text editor. QMarxan v2.0.1 is a QGIS plugin that supports the creation of MarZones input files. Vector layers were created to map the ES as later explained in section 5.2.2.1 and Figure 5.4. Data used was from modelled results, maps by regulatory authorities or manual digitisation done using a recent Google Earth image dated 12/5/2022 (Google Earth, 2018).

For areas suitable for large scale commercial fish farming, model results generated in Chapter 4 were used. Areas used for recreation services were modelled. All spatial layers were presented in vector file format done in QGIS (Figure 5.2). Other files created as GIS vector layers were the locations of the ES (for ports and harbours) and a costs layer. A file defining the target for each ES and how much could be contributed from each MA type, the abundance of each of the ES in a designated area and a file defining connectivity allowed among the MA types were included in the input folder. Additionally, there was a file defining the spatial arrangement of the MA types. There was also a boundary file including distance between PUs, and an input parameter information file containing variables and software instructions generated with the QMarxan plugin in QGIS. After preparation of inputs in QGIS or text editor, the files specifying model parameters (Table 5.1) were stored in the same input folder as the MarZones executable file. Likewise, the output folder was specified, as was the input file that carried calibration values.

5.2.2.1 Data sourcing and processing for the ES mapping

Secondary data sources on locations of the different ES were used (Table 5.2). Spatial map for fish breeding and nursery grounds was created using data of locations of gazetted fish breeding grounds appendix in Uganda Statutory Instrument, The Fish (Fish Breeding Areas) Order, 2017 (Under Section 35 of Fish Act, cap 197). Ports and harbours were manually digitized using Google Earth Engine (Google Earth, 2018) and expert knowledge about locations of the ports and harbours. An estimated buffer area of 4km to provide for future expansion of ports was suggested from discussions with experts working with the ports and harbours authority in Uganda. For recreation services, modelled results generated using the recreation module in the INVEST model (Sharp *et al.*, 2014) were used.

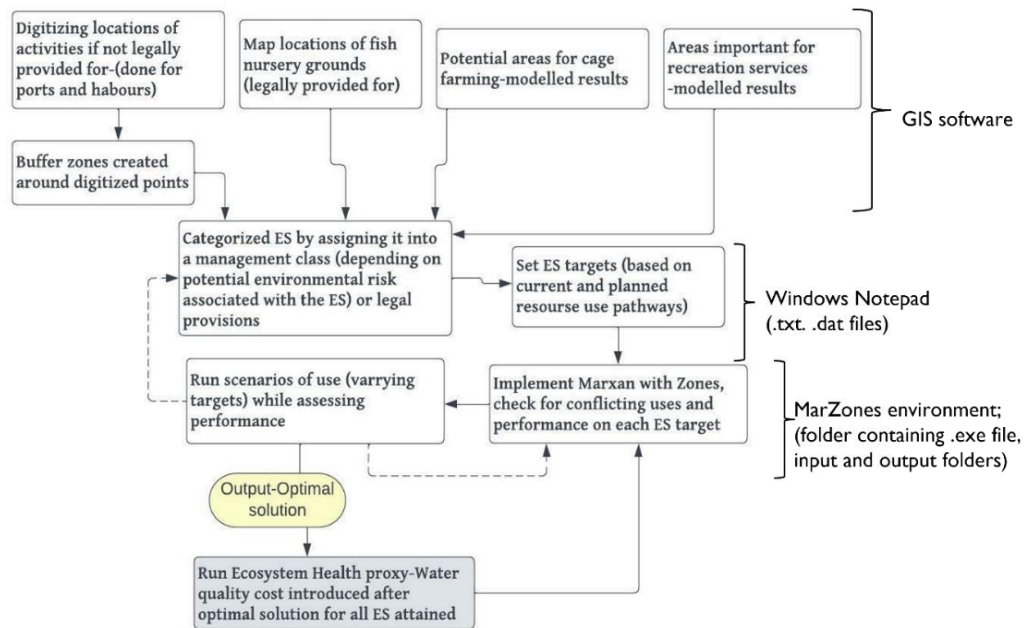


Figure 5.2: Schematic presentation of the files generated for the software, how ES were assigned into management areas; possible actions incase ES target was not achieved (dashed line).

Table 5.2: Details of the sources of the data used in mapping the ecosystem services.

Layer	Data source	Data Type	Reference
Cage fish farming	Modelled using TerrSet	1km Raster	Chapter 4 of this thesis
Ports and harbours	Mapped in Google Earth	Lat-Lon	Google Earth, (2018)
Navigation	Estimated to occur anywhere in planning area within a depth of 50m	Vector (polygon)	Provisions of the law; EAC, (2008)
Fishing	Estimated to occur anywhere in planning area except in breeding grounds	Vector (polygon)	Provisions of the regulations; MAAIF, (2017)
Recreation services	Modelled using INVEST Recreation module	Vector (polygon)	Sharp <i>et al.</i> , (2014)
Fish breeding grounds	Coordinates annexed in regulation	Lat-Lon	MAAIF, (2017)

Water Quality	Copernicus Global Land Services, Lake Water Quality	1km trophic state index averaged for 2017 & 2018	EEA, (2021)
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Similarly, for the areas suitable for large scale cage production, modelled results generated in Chapter 4 of this thesis were used. For fishing and navigation, an assumption was made that the entire extent of the study area was available for fishing activity and for navigation. An exception of no fishing within fish breeding grounds was presented by restricting the activity in the MA type where fish breeding was the core ES. Fish breeding and nursery grounds are regulated in the management of fish breeding grounds under MAAIF, (2017) . Navigation with small boats was the type included. This navigation was assumed to happen anywhere in the section of the lake selected for the study i.e., shallow lake parts with depth less 50m.

For the water quality layer, the lake water quality; trophic state index layer created over long-term assessment of chlorophyll a concentration at 1Km resolution was obtained from Copernicus Global Land Service (EEA, 2021). Averaged over two years; 2017 and 2018, the resulting layer was reprojected to UTM 36S reference system and converted to a GIS vector file. Gap filling was done in QGIS and manually verified for outliers. The output layer was joined to the PUs file by location in QGIS. All the layers were converted into vector format in QGIS projected to Universal Transverse Mercator (UTM) Zone 36S.

5.2.3 Inputs

5.2.3.1 Spatial framework

All the ecosystem services of interest occurred inshore, hence, to reduce on model runtime, an exclusion zone of depth over 50m was created. Using the Lake's bathymetry layer (Stuart *et al.*, 2016), a polygon of 1m-contour lines was generated (Figure 5.3). Areas greater than 50 m depth were excluded and the remaining section saved as the planning extent, an area covering 15,037km².

For each ES layer, small management units; PU were created (Figure 5.4). Planning units could be square, hexagonal, or irregularly shaped (Serra-Sogas *et al.*, 2020). Ardron, Possingham and Klein, (2010) recommend hexagonal PUs because they are multi-edged, have a smoother connection among neighbouring units, reducing on errors in border value configuration compared to when square planning units. Planning units were derived from dividing the planning area into 1km x1km hexagonal grids (Figure 5.4). This was done using

the Vector Research tools module in QGIS, resulting into a total of 18,603 planning units (PUs) over the planning extent. Each PU was given a unique ID number ranging from 1 to 18,603. A PUs layer containing the distribution of each of the ES was generated for each ES. The area covered within the PU was calculated using the field calculator option in the Attribute table in QGIS. Hence each PU had a unique ID and a corresponding area column for the ES contained in that PU. This area was later used as a cost in case the PU dedicated for an ES was displaced for another ES.

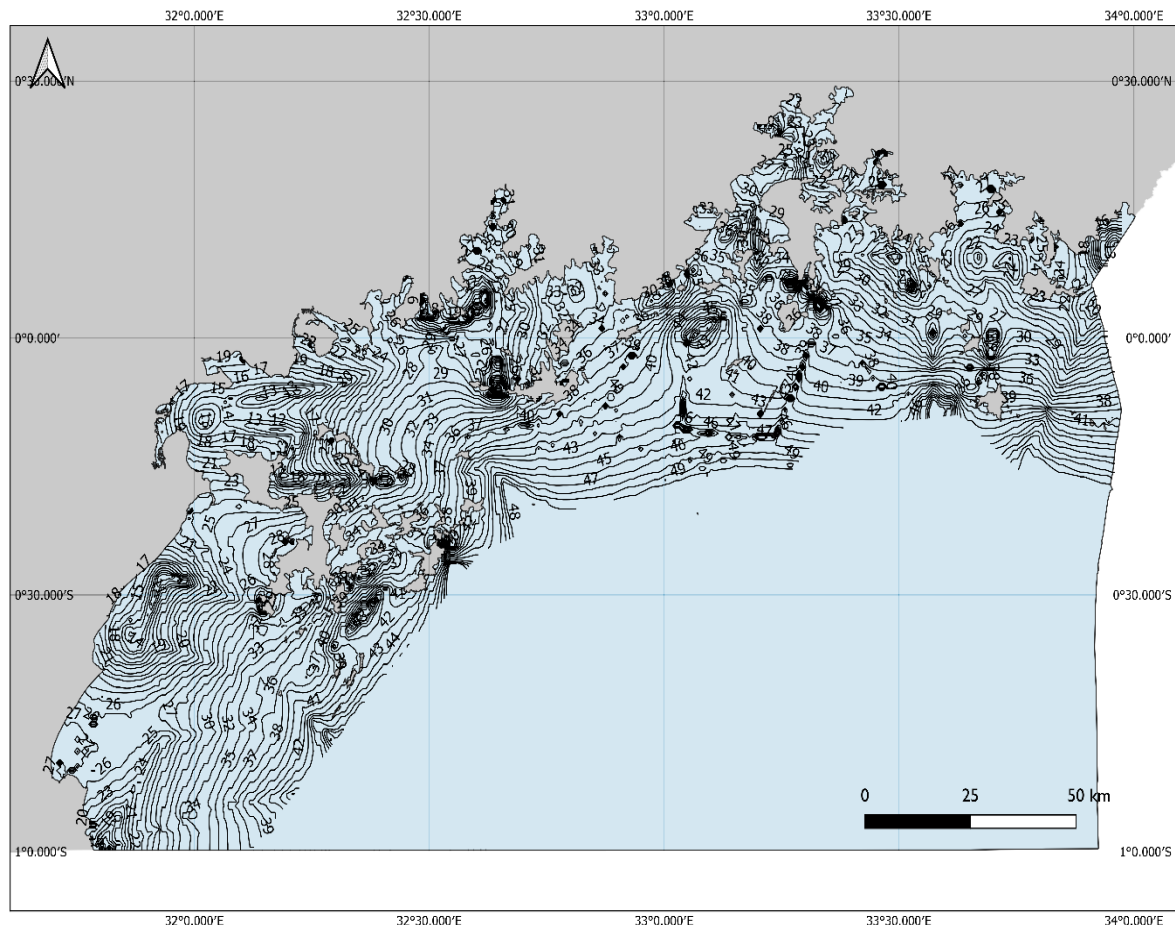


Figure.5.3: Contour Lines over the study extent (open area beyond 50m depth excluded to focus on area occupied by the ES studied)

5.2.3.2 Costs

Costs were the values that had to be minimised in pursuit of achieving a target for an ES. Two cost types were set up in the model; the opportunity cost (cost of alternation forgone) of replacing the ES in the area it originally occupied and the water quality cost based on trophic state defined by chlorophyll a concentration. As a function of an ES's abundance (in km²), the

opportunity cost of replacing it was calculated. Each PU was assigned a cost value depending on how much of the ES lay in that PU (except for fisheries) as explained below.

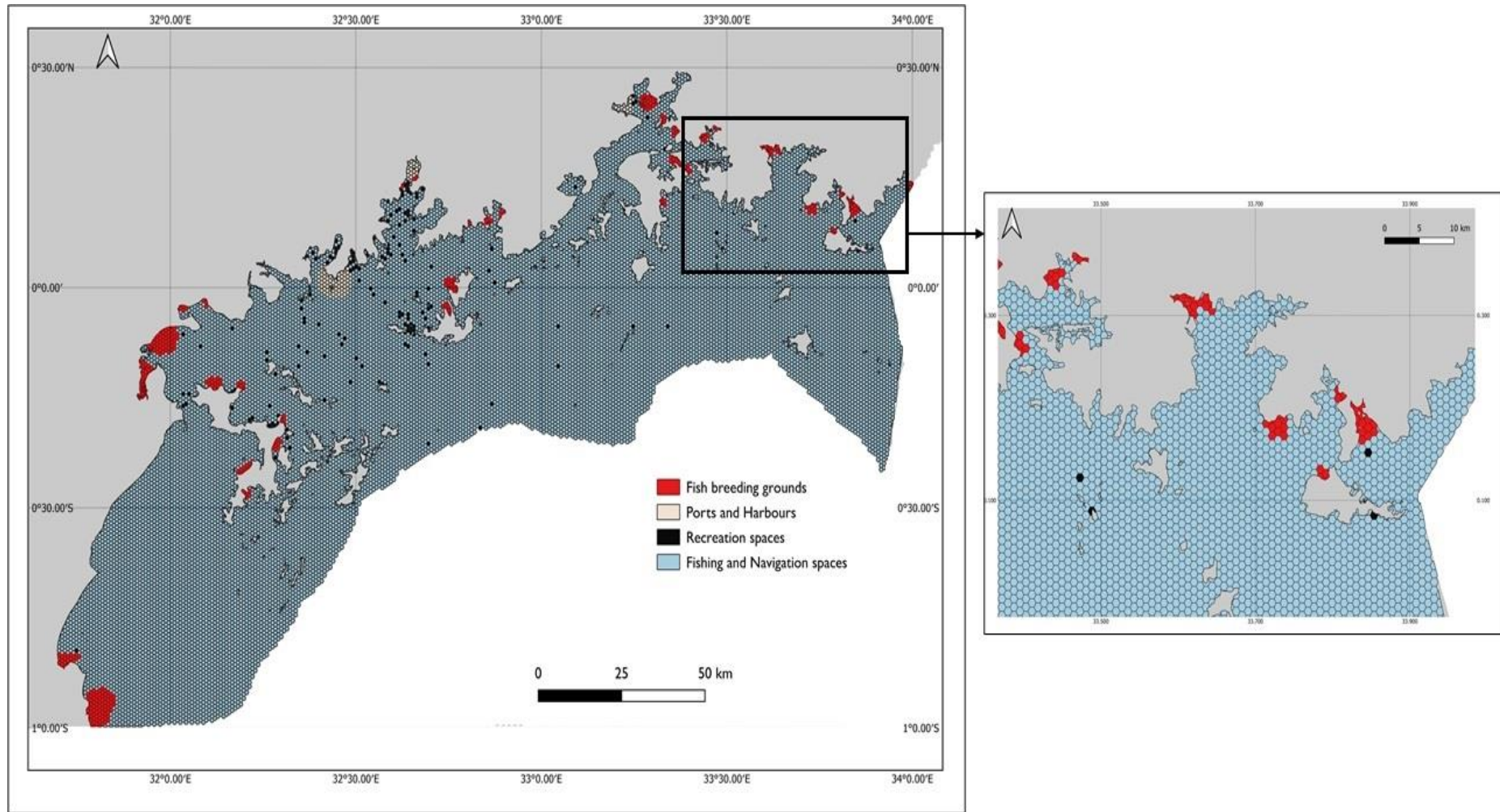


Figure 5.4: Location of ES with focus on the hexagonal planning units (one full planning unit was 0.86km²).

The space available to ports and harbours was first mapped by creating a buffer around the current port locations. The area was calculated using the geometric area module in field calculator in QGIS. It is this geometric area that formed the cost of replacing this space with another ES i.e., opportunity cost of losing space meant for ports. The same was done for navigation, the initial area being any of the PU under the planning area and the cost calculated as the geometric area that each of the PU occupied.

The cost on fisheries was included as the opportunity cost of displacing fishers per PU using the formula in *Equation 5.2*. Data for the number of fishers in a district (administrative unit) (X) was obtained from the 2017 frame survey (DiFR, 2017); a routine census done to establish the status of fisheries on the lake. The cost of displacing fishers (Y) was calculated as a product of the total population of fishers reported in that district (P) and the ratio of the size of the PU (a) to the total area the district covered (A).

$$Y = P \left(\frac{a}{A} \right) \quad \text{Equation 5.2}$$

The resulting cost value was relative to the size of the PU and the population of fishers in the district i.e., smaller PUs had a relatively smaller cost compared to the larger ones. Smaller districts like Mayuge to the far East had higher population of fishers compared to the area occupied under the district while others like Kalangala, to the west, had wider areas with a relatively lower number of fishers reported (Figure 5.5). It was more costly to replace fisheries in the former than the latter.

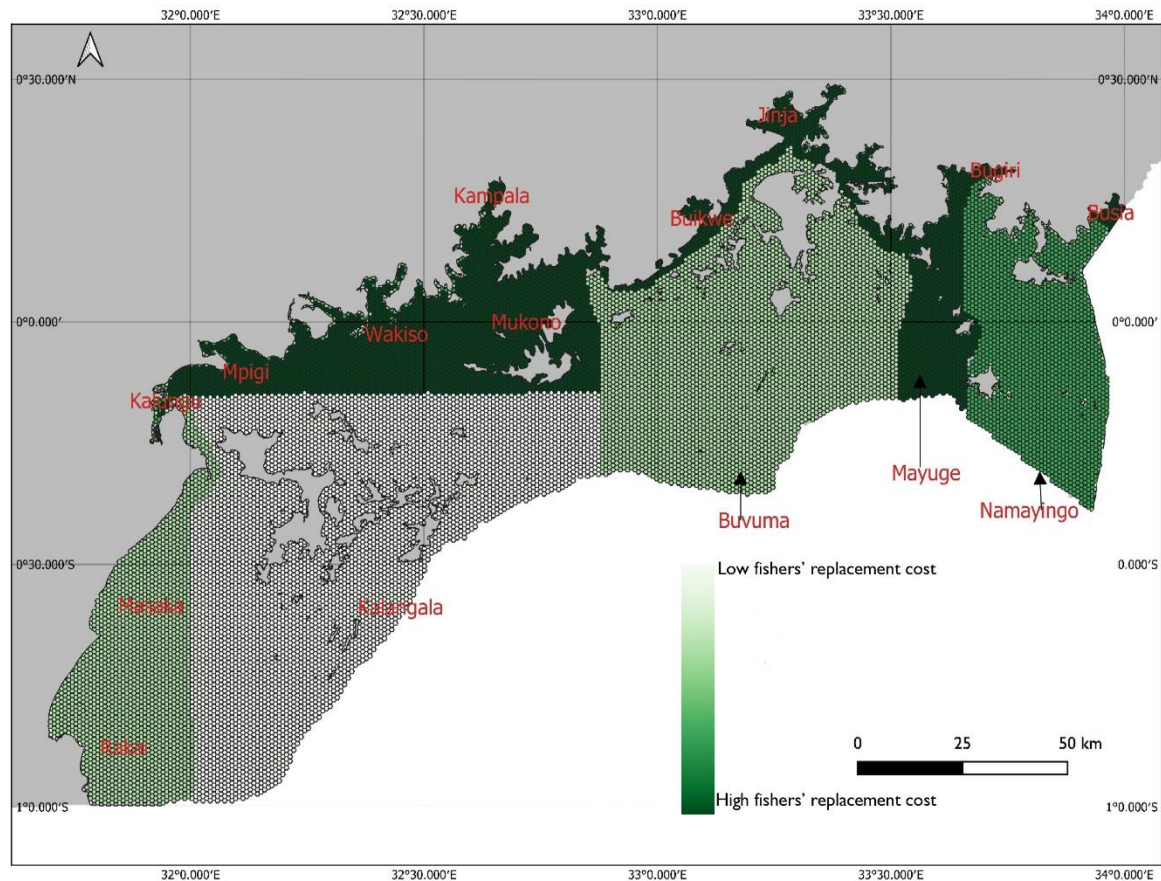


Figure 5.5: Relative population of fishers per PU generated from frame survey data on number of fishers per administrative bloc for 2017 (DiFR, 2017).

To create the cost layer for recreation services, the INVEST model's recreation module was used. In the model, the value of the natural environment was quantified using a proxy for visitation i.e., the rate of visitation across a grid cell was calculated based on geotagged photos posted on the Flickr website (Sharp *et al.*, 2014). In the model, as photos are uploaded by revellers to the website, a coordinate was assigned. Basing on these uploads, a measure of average annual number of photo-user-days per 1km hexagonal PU in the planning area was generated for the period 2005-2017. To create the layer for the opportunity cost of replacing spaces important for the recreation ecosystem services (Figure 5.6), a Boolean layer was generated out of the photo-user-days averaged over the 12 years such that the area in any of the PUs with a photo-user-day was weighted the same in terms of importance for recreation ES. The resulting map was validated by areas close to renown tourist spots, major towns, and cities, where use of smartphones is common being included in the recreation cost layer.

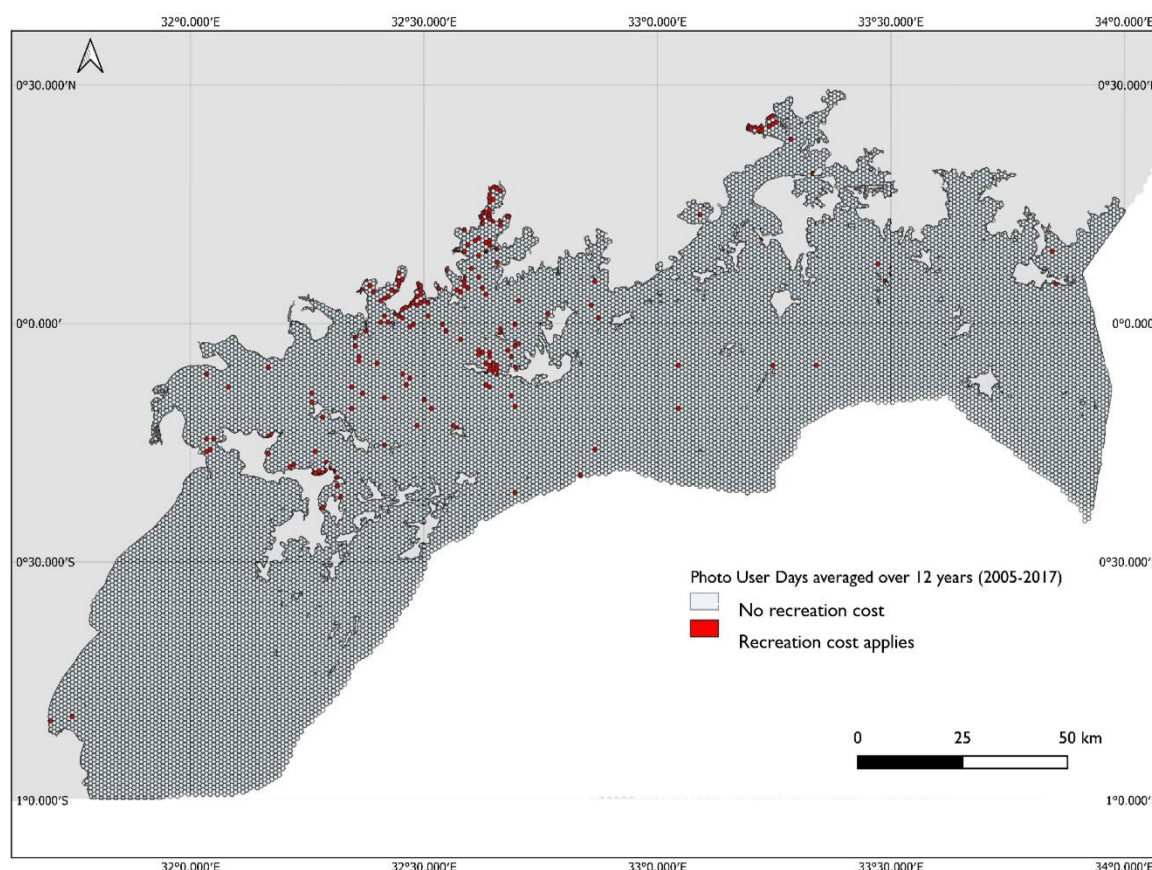


Figure 5.6: Recreation activities Boolean layer created with the INVEST model from which the opportunity cost to replace leisure activities was generated.

The areas with potential for cage farming used were results of modelling for site suitability done in Chapter 4 of this thesis. Factors considered were distance from markets, distance from water hyacinth hotspots, distance from existing fish farms, bathymetry and viewshed while the constraints considered were locations of fish breeding grounds, protected areas, a shoreline buffer, and navigation buffers. For inclusion in the current study, the fuzzy suitability output was reclassified into 5 classes that represented weights of suitability (Table 5.3). The opportunity cost of replacing cage farming was weighed. In areas with higher aquaculture suitability, replacement costs were higher (Figure 5.7), i.e., factor of 4 higher than those with very low suitability (Table 5.3).

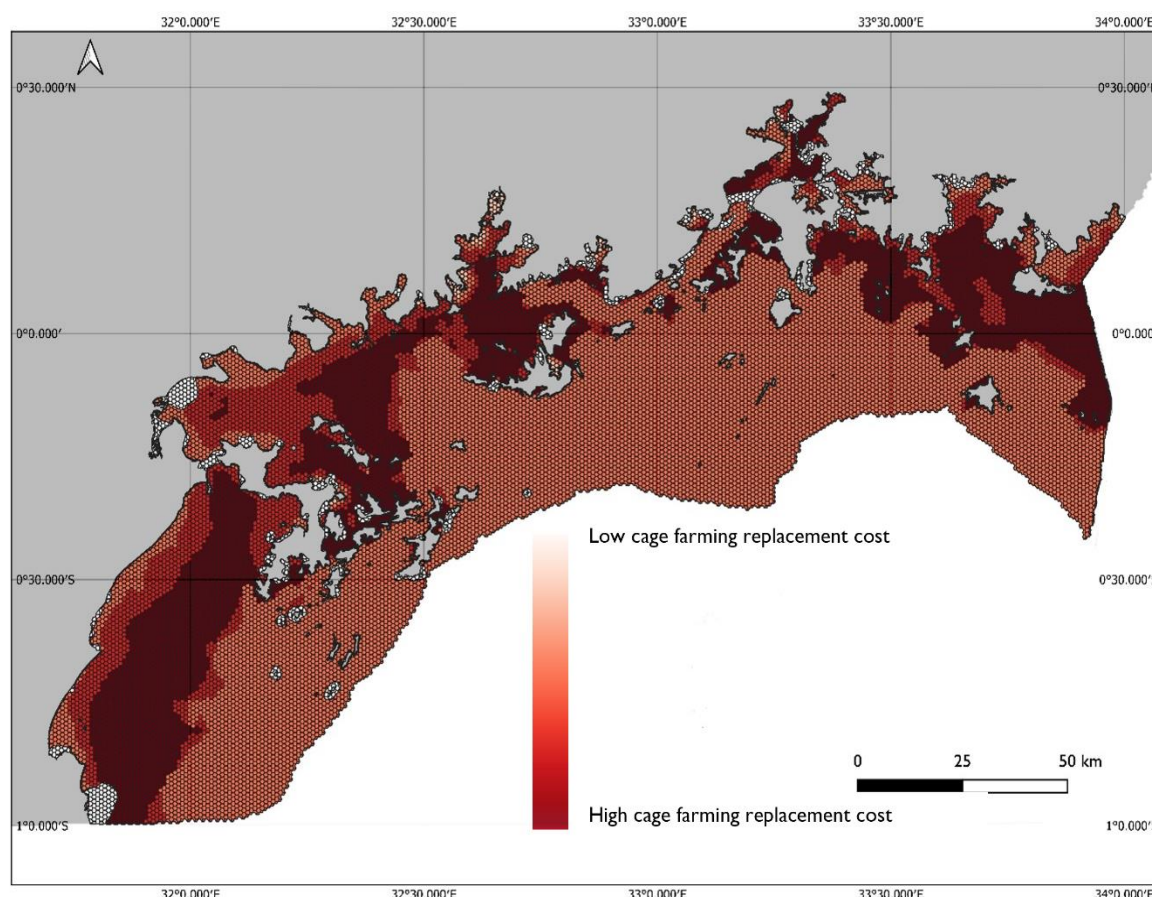


Figure 5.7: Cage aquaculture suitability layer from which weighted cage farming cost was generated.

Table 5.3: Reclassification of the cage farming suitability layer to generate a weighted cage culture cost layer.

Fuzzy suitability scale Intervals	Category	Reclassification to produce cost layer
0-0.01	Constraint	Area factored by 0
0.01-0.25	Very low suitability	Area factored by 1
0.25-0.50	Low suitability	Area factored by 2
0.50-0.75	Moderate suitability	Area factored by 3
0.75-1	High suitability	Area factored by 4

To cater for ecosystem health, a water quality cost was included as a proxy to ecosystem health. This was done to assess if difference in water quality in the sections covered under different MAs had an impact on the optimal solution (output). Using the trophic index layers of 2017 and 2018 generated from analysis of chlorophyll a concentration, (EEA, 2021) , a cost value per PU was created (Figure 5.8). Like the cage farming cost layer, to vary the importance of water quality across MA types, weighting was done. PUs assigned to MA type 1 (Available

MA) had their trophic index value weighted by 1 (maintained). Priority ES in this MA type, i.e., fishing, navigation, ports, and recreation services were assumed least affected by water quality. A water quality factor of 5 was applied to the PUs in MA type 2 (Target MA), where cage farming was the priority ES, while a factor of 10 was applied to PUs in MA type 3 (Protected MA), where fish breeding and nurseries were the priority ES. This weighting was based on how sensitive the priority ES could be to water quality. Poor water quality could negatively impact fish breeding and nurseries more than fishing, for example.

Costs in each PU were measured in different units, so to standardize them onto a uniform scale, the cost in each PU was divided by the largest value of that cost in all PU and multiplied by 100; thus, the scale was standardized into 0-100. In a PU, the final cost would be a total of all the opportunity costs of the ES in that PU, the cost of dislodging fishers and the cost based on water quality, an indicator of ecosystem health.

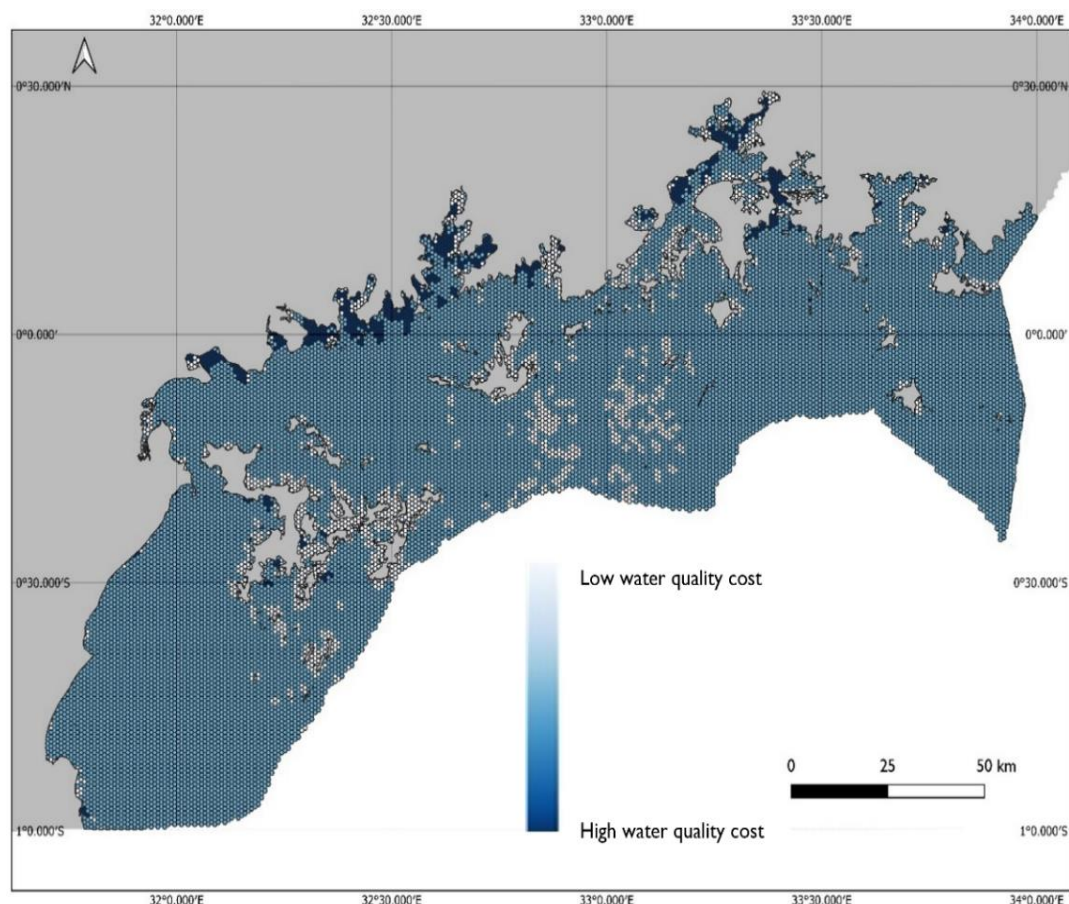


Figure 5.8: Water quality cost layer based on Chlorophyll a concentration assessed over a long time periods by Copernicus Land Monitoring Service (EEA, 2021).

5.2.3.3 Targets and penalties

Performance across ES was assessed when fishing, cage farming, and breeding grounds for fish were given high priority. Initially, the baseline run had minimal targets for the ES of interest (fishing, cage farming and fish breeding grounds), while other ES were maximized at 100% (Table 5.4). The penalty (which was a multiplier) for not meeting the ES target was included. The penalty factor was varied throughout calibration runs and eventually fixed at 100 for all ES. Setting the highest penalty ensured maximum performance was achieved across all ES targets at the least costly, optimal level. Where the penalty was low, it was not sufficient to induce the ES target to be achieved, hence this was increased.

Table 5.4: ES, targets, and penalties used in the base run.

Objective	Base run target	Penalty
Fisheries	70%	100
Cage Fish farming	50%	100
Fish breeding and nurseries	100%	100
Navigation	80%	100
Ports and Harbours	100%	100
Recreation services	100%	100

5.2.3.4 Management Areas

Management Area types (MA), also known as networks or zones, were the spatial representations of the multi-use scheme determined by the management area rules. In any MA type, several ES could exist, depending on the potential risk posed, or if they were legally provided for. To design a solution using MarZones, ES had to be assigned to a MA type. Three MA types were devised (Table 5.5): Available (MA type 1), Target (MA type 2) and the Protected Management Area (MA type 3).

The MA types were benchmarked on Boyes et al. (2007) and Hermoso et al. (2018) based on the interaction between ecosystem services and aquatic environments and the possible magnitude of damage caused by conflicting uses. Provisions of the law were also taken into consideration. Fish breeding and nursery grounds, for example, are legally no fishing, no cage farming areas. Therefore, the MA type where fish breeding was a priority, had restricted use (0%) for fishing or cage farming. However, in the cage farming zone, a fraction of fishing and fish breeding were allowed.

Activities that occurred without stringent regulations were allowed to be allocated to a MA type (100% cleared by authorities), whereas activities that required licensing could only be allocated a fraction because they required clearance from authorities. 'Restricted' activities (0% cleared) were not permitted in that MA type. This interpretation was for purposes of setting up software but may not fully reflect regulatory provisions by authorities. By law, most activities on Lake Victoria are subject to some level of licensing or permitting.

Table 5.5: Allocation of ES to different Management Areas.

ES	Available-(MA type 1)	Target – (MA type 2)	Protected-(MA type 3)
Cage culture	Restricted	Occurs	Restricted
Fishing	Occurs	Occurs with a license	Restricted
Fish breeding and nurseries	Occurs	Occurs	Occurs
Navigation	Occurs	Occurs	Occurs
Ports and Harbours	Occurs	Restricted	Restricted
Recreation services	Occurs	Occurs	Occurs

Based on the provisions in the MA type (Table 5.5), the contribution of each MA type towards meeting the ES was set (Table 5.6). These were the minimum quantity of the ES that had to be included in a MA type. The MA type targets had to total up to ES target (Table 5.4 for corresponding ES targets). These targets were varied according to the priority of the MA type. Each MA type had a priority ES i.e., that ES whose target took priority to achieve from that MA type. For the core ES, fishing was priority in MA type 1, cages in MA type 2 while breeding and nursing grounds took priority in MA type 3. Other ES were spread out among these. These MA type details were used to define the zone contribution file specifications. Table 5.6 shows the translation of Table 5.5 in the zone contribution file included in the input folder.

The priority ES was set only once and adjusted only during calibration if, for instance, performance of another ES was poor and more needed to be contributed from that MA type to the ES. For instance (Table 5.7), in MA type 3, 100% priority went to fish breeding and nurseries (ES id 3), smaller commitments for other ES like navigation (id 4) and recreation services (id 6) could be achieved here. Priority in MA type 2 was cage farming (id 2), hence 100% commitment to meeting the target while no recreation services were permitted hence 0 for id 6. Other ES were allocated fractions of commitment during calibration. Similarly, for MA

type 1, the committed ES were fishing (id 1), navigation (id 4), ports (id 5) and recreation (id 6) while no cage farming were planned (id 3) and breeding grounds only occurred partially (id 2).

Table 5.6: MA type targets per ES. The proportion of the ES that was to be contributed from each MA was set according to provisions in Table 5.5

ES id	ES	MA type 1	MA type 2	MA type 3
1	Fishing	50%	20%	0
2	Cage culture	0	50%	0
3	Fish breeding and nurseries	20%	10%	70%
4	Navigation	70%	5%	5%
5	Ports and Harbours	100%	0	0
6	Recreation	60%	30%	10%

As a setting in the software, MA type priority does not need to add up to 1 or 100% or to the ES target, but it represents commitment from the MA type due to what is priority in it. Commitment of any of the ES could be spread across MA type, but where the highest proportion was targeted, it formed the key ES.

5.2.3.5 Other Inputs

The Boundary Length Modifier was set at 1. A boundary length file was generated using the QMarxan plugin in QGIS. The boundary length file contained information on the boundary 'costs' between two planning units. This was the inverse distance between the PUs. If PU were being considered for inclusion in the same MA type in a solution, ones close together would be cheaper to select into the solution than those further apart. To ensure spatial compactness among MA types, a network boundary cost file was included. This defined the overall arrangement (or clustering) of the MA types in the solution.

Table 5.7: Example of the Zone contribution file defining arrangement in Table 5.6.

MA id	ES id	Fraction of ES
1	1	1
1	2	0
1	3	0.25
1	4	1
1	5	1
1	6	1
2	1	0.25
2	2	1
2	3	0.5
2	4	0.25
2	5	0
2	6	0
3	1	0
3	2	0
3	3	1
3	4	0.25
3	5	0.25
3	6	0.5

Calibration of the MA type boundary cost file was a manual process that involved adjusting the value between two MA types and assessing connectivity in the spatial outputs before redoing it for another two MA types, until satisfactory compactness was attained. As recommended by Serra-Sogas *et al.*, (2020) starting with two networks (1 & 2), the value was moved from 0 to 0.001; results were viewed and then introduced a value between network 1 and 3 from 0 to 0.001. Final value between network 1 and 2 was set at 0.001 and network 1 and 3 was 0.1. These provided good compactness among networks 1, 2 and 3 and were maintained.

Other calibration processes included attempts to lock some of the PUs. This meant that they couldn't be tried for inclusion in any other MA type in the solution but only the ones they were locked to. When this was done, only 2 out of the 6 ES targets were met. This solution was not optimal enough and hence, all the planning units were left unlocked.

5.2.4 Scenarios

Scenarios were run to identify priority areas that enhanced co-benefits and minimised trade-offs. Changes in priorities for the MA type across the core ES of fishing, cage farming and fish

breeding and nursery grounds (Table 5.8) were done. After an optimal solution was attained where all ES were set at 100% i.e., Scenario 4, a water quality cost was incorporated in the costs' configuration of the software. The inclusion of the water quality cost after testing performance across all ES till the maximum target was aimed to assessing if ecosystem health could have an implication on model output.

Table 5.8: Scenarios and target representing changes in priorities.

Objective	Base scenario	Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Fisheries	70%	70%	70%	100%	100%	All targets set at 100% and introduced a water quality cost layer-varied among MA types
Cage Fish farming	50%	70%	90%	90%	100%	
Fish breeding grounds	100%	100%	100%	100%	100%	
Navigation	80%	100%	100%	100%	100%	
Ports and Harbours	100%	100%	100%	100%	100%	
Recreation services	100%	100%	100%	100%	100%	

5.2.5 Outputs

5.2.5.1 Analysis of the outcomes

MarZones was run for 10,000 iterations, for each scenario, producing two most useful outputs per scenario: the best solution and the selection frequency file. The best solution was a representation of how the PUs were placed to meet the targets across all ES in the least costly way. The best solution targets to avoid trade-offs as much as possible.

The selection frequency shows how many times (out of 10,000) each PU was selected to contribute towards of meeting the target, values ranged from 0 to 10,000. Selection frequency maps for each zone are presented in the results section on a scale of 0-100%. Based on Vallecillo *et al.*, (2018), the total area of the PUs which were selected over 90% was calculated for each zone.

5.1 Results

5.1.1 Performance of ES targets across Scenarios

From the analysis, at least each ES target was achieved to a minimum of 40% (Figure 5.9) with the worst performance achieved for recreation services at 32%. Introducing the water quality cost in scenario 5 resulted in achievement of 2 out of 6 ES targets. Fish breeding and nurseries and recreation services performed the worst at 52.0% and 32.1% while fishing was attained at 97% and cage farming at 99%. In the base scenarios, targets for fishing, cage farming and navigation were 50%, 70% and 80% respectively, while other ES were at 100%. All targets were met at 100% except for recreation services and fish breeding grounds met at 53% and 68% respectively. In scenario 1, targets for both fishing and cage farming were increased to 70%, while other ES were set at 100% and achievement level was like the base scenario. In scenario 2, targets for cage farming were increased to 90%, fishing left at 70% and the other ES at 100%. Cage farming targets were met by 91%, recreation services 45% and fish breeding at 50% while other ES targets were met by 100%. In scenarios 3, 4 and 5, still ES targets of fish breeding areas and recreation services weren't fully met.

In the setup of the management framework, 56% of the space was set up under MA type 1 (Available), 42.8% under MA type 2 (Target) while 1.2% was under MA type 3 (Protected). The priority ES in MA type 3 was fish breeding and nursery grounds. Limited spatial commitment of only 1.2% from MA3, meant a limited ability to contribute to the targets of any of the ES placed in this network. When the priority ES had a wider expanse (more Pus) to select from (such as cage farming), ES targets were met in most scenarios.

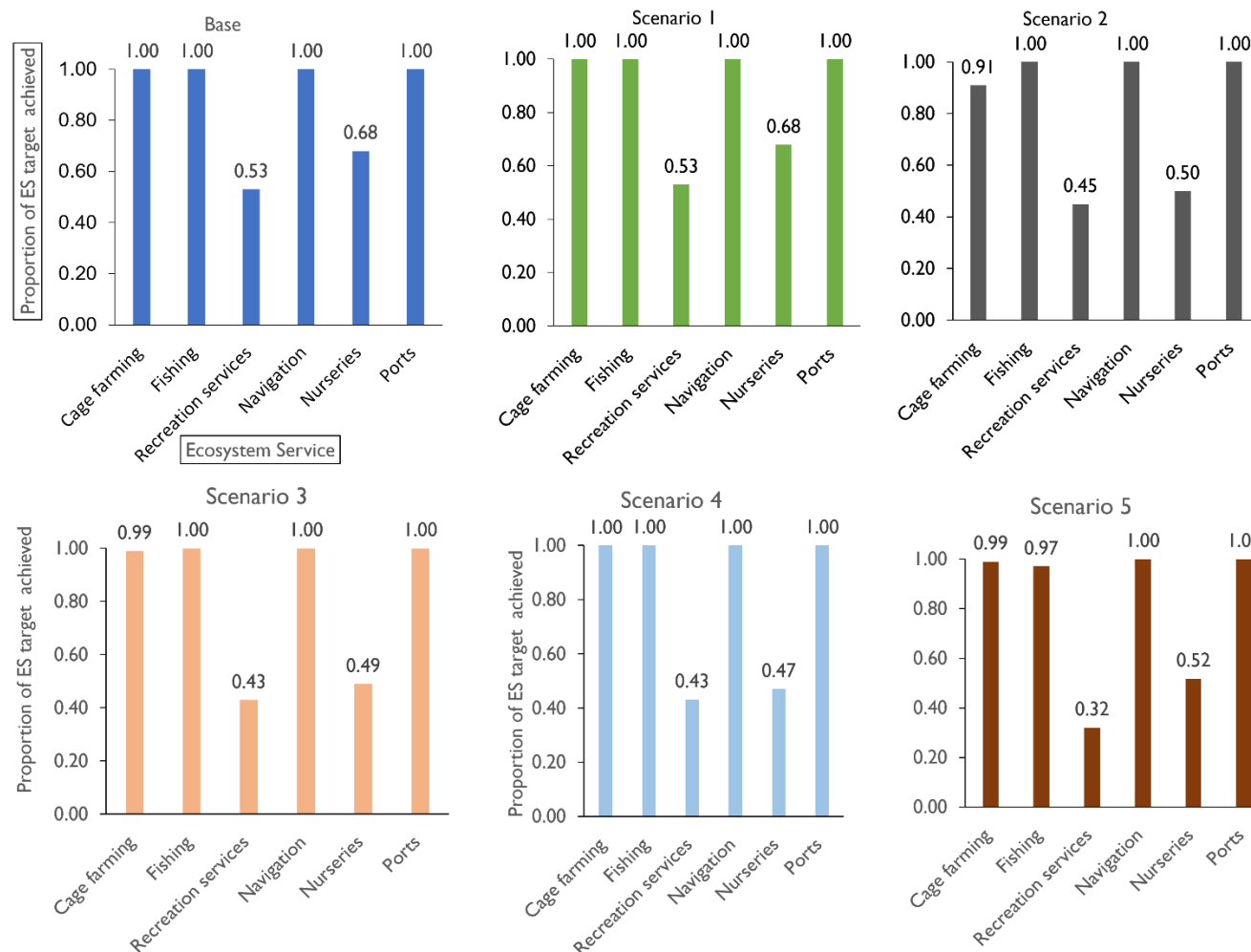


Figure 5.9: Level of achievement of ES targets across scenarios. Worst performing ES in all were recreation services and fish breeding and nursery grounds while the least impacted ES were Ports and Navigation. This could be due to limited space required for the latter ES. Similarly, ports and navigation were setup in the model to happen in synergy with other ES.

5.1.2 Best solution

In the best solution, across all scenarios (Table 5.9), the largest areal commitment was for the Available management area network (MA type 1) followed by the Protected area network (MA3) and lastly the Target area network (MA type 2). The Target area (MA type 2) contributed to cage farming as the key ES, with least commitment to navigation, fishing, and recreation services. Whereas Available area (MA type 1) was almost equally committed to all ES apart from cage farming. However, compared to the setup of 56% of the space under MA type 1 (Available), 42.8% under MA 2 (Target), MA type 1 had a larger bulk of ES compared to other networks. Similarly, MA3 was committed to fish breeding and nurseries, navigation, and recreation services. In general, in the best solution, more area was allocated into meeting the ES priority for the MA type before other ES were considered. In Scenario 5 (Table 5.9), more area needed to be allocated in MA type 2 and MA type 3 but not in MA type 1. In the setup, the burden of water quality was weighted highest in MA type 3, followed by MA type 2 and least in MA type 1.

Table 5.9: Area covered under management area types for different scenarios.

Scenario	Area best solution (km ²) (% allocation)		
	MA type 1	MA type 2	MA type 3
Base scenario (solution with at least 40% target achieved on all ES)	8,997 (59.8%)	1,090 (7.2%)	4,951 (33%)
Scenario 2 (Cage farming 90% and increase fishing from 70 to 100%)	10,388 (69.0%)	712 (4.7%)	3,938 (26.2%)
Scenario 3 (100% target for fishing and cage aquaculture)	7,923 (52.7%)	815(5.4%)	6,300 (41.9%)
Scenario 5 -Water Quality Constrained	5,011(33.3%)	3,967 (26.4%)	6,060 (40.3%)

Best solution maps (Figure 5.10) were created to show the management area types alignment, and the connectivity among the networks in the optimal solution. As targets for the core ES were increased from scenario 1 through to 3, the arrangement in the MA type networks were more visually comparable. Targets for cage farming in scenarios 1 and 2 were lower, causing less visibility of the MA type 2 network on the map. Introducing a weighted cost in scenario 5

increased on the connectivity among PUs in the MA type, with clearer identification of the Target MA type network from others. Scenario 5-(Figure 5.10D) is most representative of current conditions; the weighted water quality constraint influenced a better spatial alignment among the management areas but led to concentration of features of the Protected Area (MA type 3) in areas earlier set up for Available features (MA type I).

5.1.3 Selection Frequencies

Selection frequency maps (Figure 5.11,5.12 and 5.13) represent the number of times out of 100, a given PU was selected to contribute to a solution for any of the management area networks in the different scenarios. PUs that were selected more than 90% of the time to contribute to the solution for a network, in each scenario formed the irreplaceable area of that management area network in any scenario. The total area (Table 5.10) of these PUs was calculated. The smaller the area, the higher the possibility of alternative solutions (a more flexible arrangement) while a larger irreplaceable area meant, the solution bound a wider space to be able to attain the objectives of that management area network.

Table 5.10: Total area of the PUs selected more than 90% to contribute to the optimal solution per scenario.

Scenario	Irreplaceable Area (km ²)		
	MA type 1	MA type 2	MA type 3
Base scenario (solution with at least 40% target achieved on all ES)	304.1	0	189.8
Scenario 2 (Cage farming 90% and increase fishing from 70 to 100%)	1.75	0	18.2
Scenario 3 (100% target for fishing and cage aquaculture)	28.9	0	184.14
Scenario 5 -Water Quality Constrained (100% target for all ES)	10.34	0	399.7

The highest irreplaceable area was in scenario 5 for MA type 3 (Figure 5.13), where fish breeding grounds and nurseries were the key ES. On the contrary, MA type 2 (Figure 5.12) didn't need to bind any PU to meet its ES targets, a possibility of a flexible network where cage farming was the priority ES.

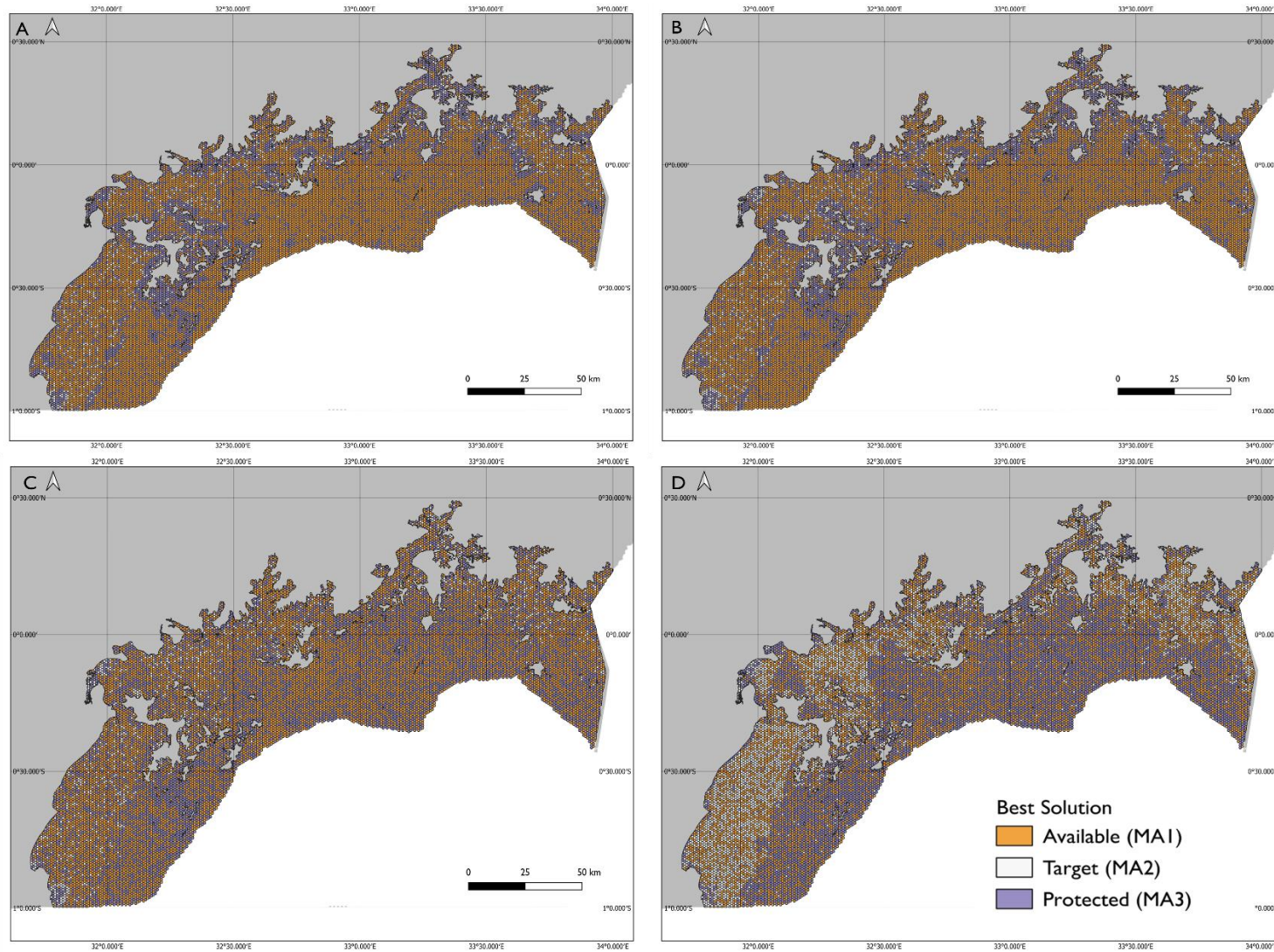


Figure 5.10: PU configuration in the Best Solutions for A (Scenario 1), B (Scenario 2), C (Scenario 3) and D (Scenario 5-water quality constrained)

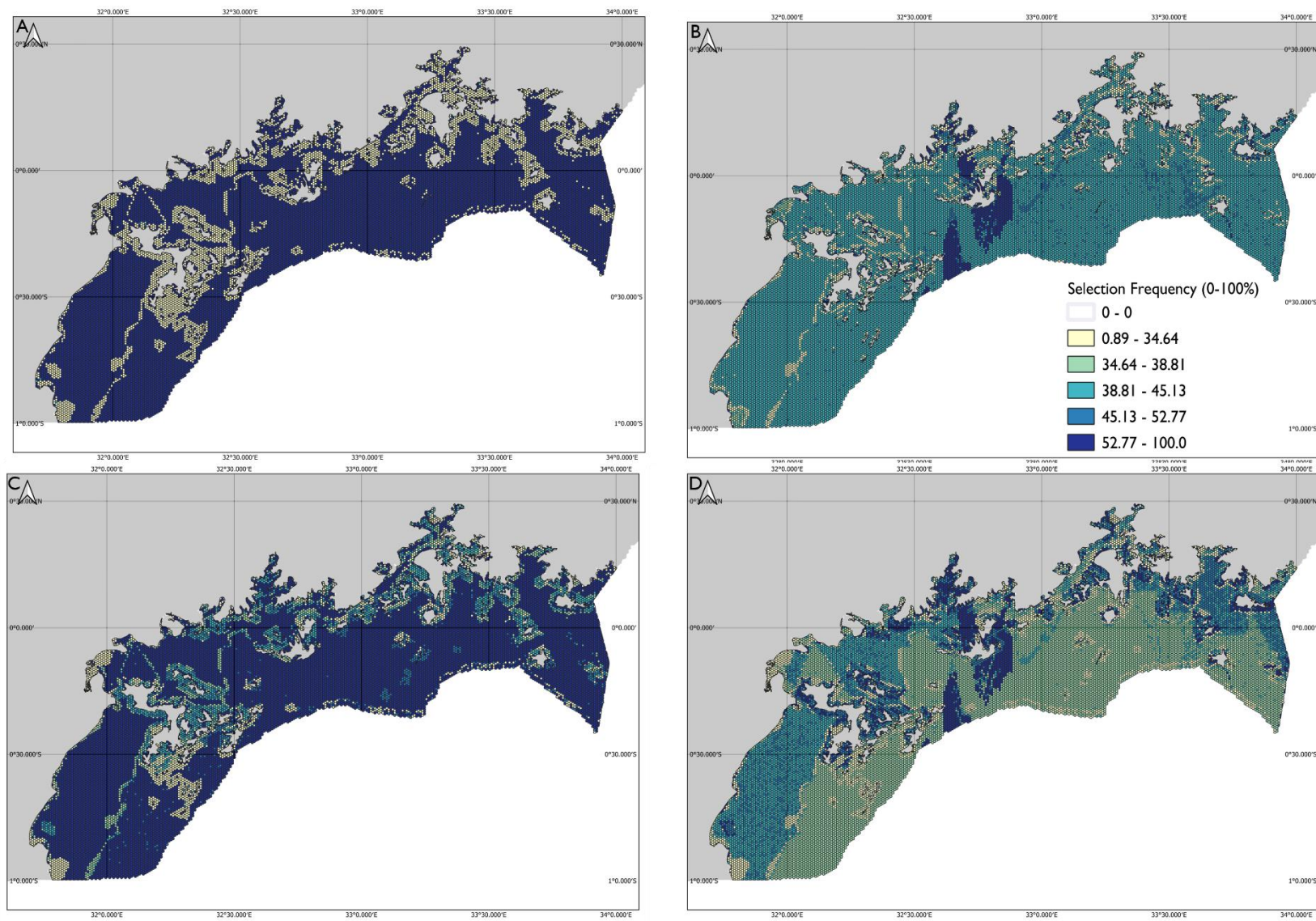


Figure 5.11: Selection frequency for the Available MA (MA type 1). A- Scenario 1; B-Scenario 2; C-Scenario 3; and D-Scenario 5

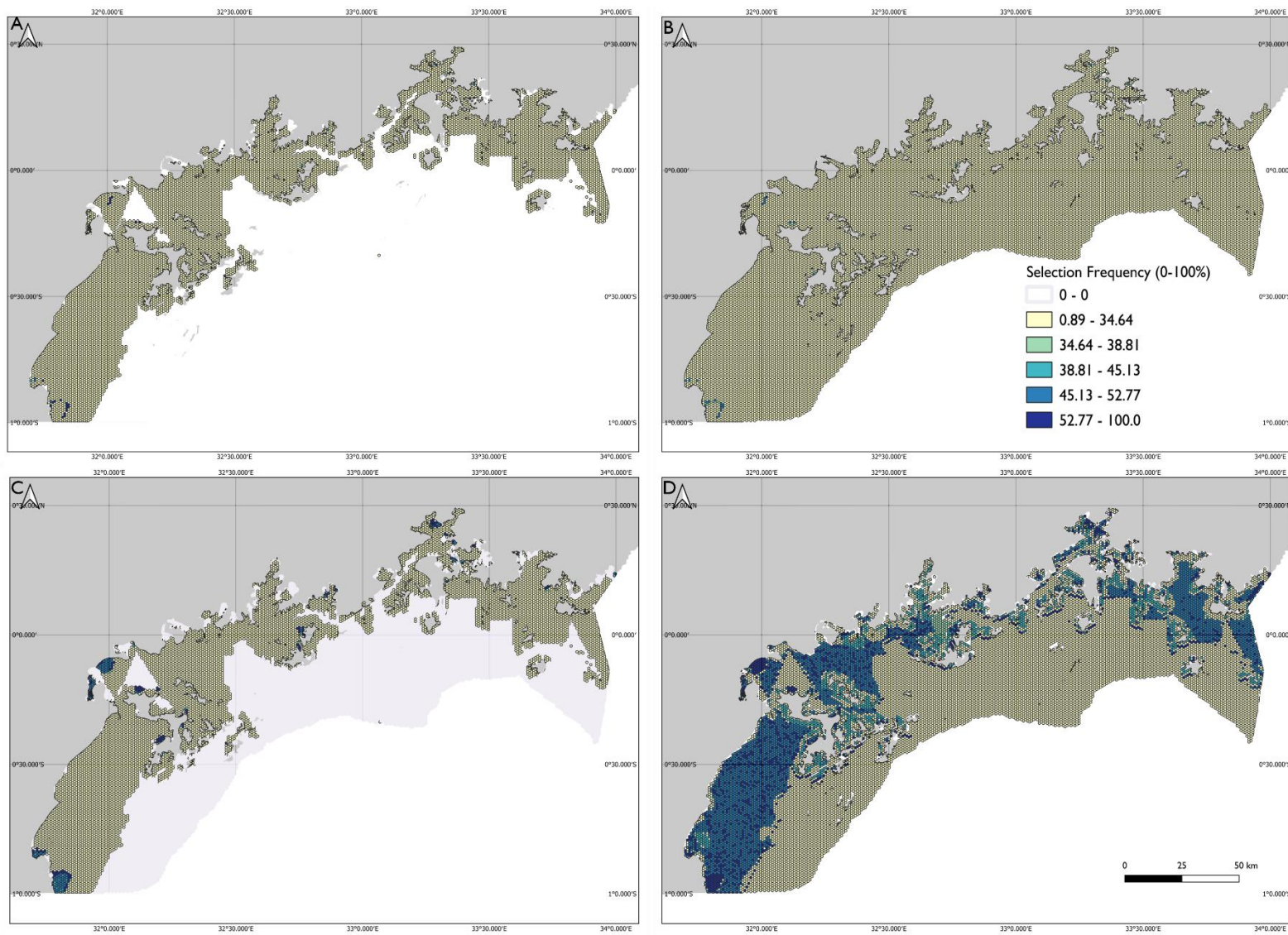


Figure 5.12: Selection frequency for the Target MA (MA type 2). A- Scenario 1; B-Scenario 2; C-Scenario 3; and D-Scenario 5

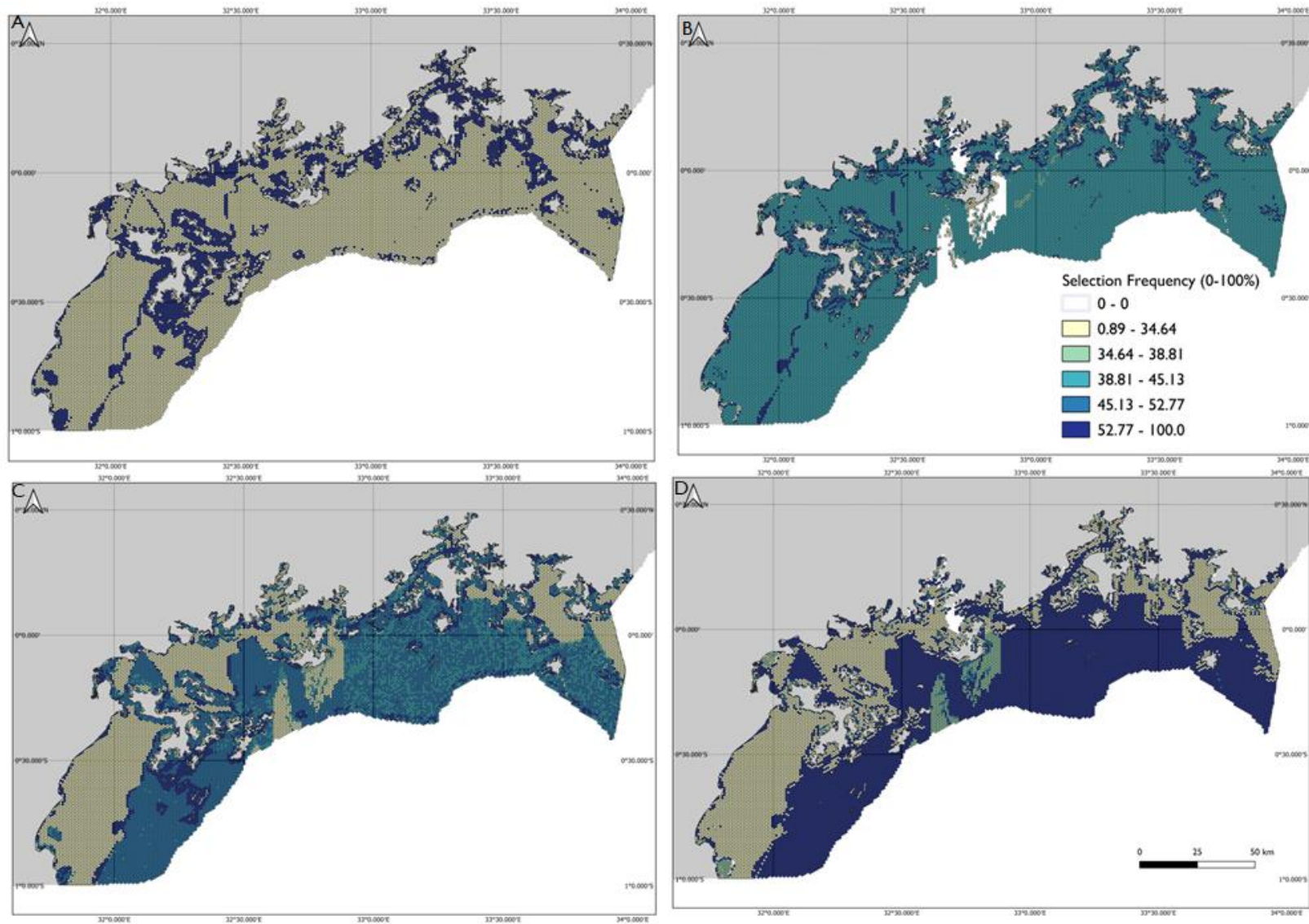


Figure 5.13: Selection frequency maps for the Protected MA (MA type 3). A- Scenario 1; B-Scenario 2; C-Scenario 3; and D-Scenario 5

5.2 Discussion

Lake Victoria provides various ES to its users, yet changes in the environment, increased population and deteriorating ecosystem health is assumed to deter a sustainable supply of some ES of socio-economic importance. This study uses a spatial optimisation tool to analyse trade-offs and synergies, identify conflict-prone areas on the lake, and assess ES achievement under scenarios of holistic management to assess cage farming integration with other ES. Multi-use planning for trade-off & synergy analysis was used to maximize socioeconomic benefits from cage farming, fishing, and commitment to fish breeding and nursing grounds. The ES evaluated were spatially explicit but had a potential to conflict with cage farming i.e., navigation, ports & harbours, space for recreation services, fish breeding grounds, and fishing. A cost analysis was carried out on replacing any ES within the planning area based on their occupancy, location, or legal provision. The socio-economic cost included in the model was the cost of moving fishers from their areas of operation, using data from the 2017 fishing frame survey (DiFR, 2017).

Defining meaningful amounts or proportions of an ES to exist in an area is challenging because it is determined by many external factors (Janssen *et al.*, 2021). Demand, supply and flows of ES are highly dynamic (Villarreal-Rosas *et al.*, 2020). As a measure of projected increase in demand, scenarios where objectives were changed were used. According to studies like Adame *et al.*, (2015), Ter *et al.*, (2014), and Vallecillo *et al.*, (2018), 50% of the ES target needs to be met. Hypothetical but meaningful targets based on lake use pathways both current and future were used in this study. The minimum achievement across the ES was set at 40%, for the solution to be deemed optimal due to poor performance achieved on some of the ES. As a result of the level of risk associated with priority ES in those areas, boundary length modifiers (an additional cost) were incorporated to separate management area networks.

Results indicate that fishing and cage farming could be maximised with limited impact on other ES, but fishing breeding & nursery grounds and recreation services. These had the least met targets in all scenarios. This could be attributed to two factors; space allocated for the ES and the quality of the environment. These ES were priority in the protected management area network which mostly occupied the littoral parts of the lake. Nearshore waters of Lake Victoria are classified as eutrophic (Deirmendjian *et al.*, 2021; Onyango & Opiyo, 2021). Meeting high priorities (100% target with a maximum penalty of 100 if not met) amidst poor ecosystem health (high water quality burden in the littoral zone) were determining factors of performance.

This indicates that where ecosystem health impacts the supply of some ES, unavoidable trade-offs must be made (Mato & Kassenga, 2019).

In all ES, only recreation services and fish breeding grounds & nurseries fell short as targets across ES were progressively increased. This could be an indication that regardless of the increase in ES targets, there was enough space for the ES initially setup in wider expanses. It can be concluded that if larger areas were available for the various ES, targets would be met with limited trade-offs unless other defining factors like water quality were involved. However, where the demand for ES is high, space is always a major limiting factor. Therefore, while setting targets, compromises must be made, bearing in mind other needs that must be met. Brugere et al, (2015) conclude that regardless of the importance of increasing fish production in Lake Victoria (through fishing or cage farming), factors governing this production cannot be traded off to achieve the fish production objective. These results suggest that this condition fails when the ecosystem service cannot occur in synergy with fish farming or fishing. This is the case, for example, with recreation services and fish breeding grounds.

With spatial optimisation, visual interpretation of ES selection frequency maps like those generated in this study could inform allocation of activities in such a way that further ecosystem deterioration is avoided, and conflicts are abated (Watts *et al.*, 2009). With the selection frequency maps, it was possible to identify parts of the lake that were more prone to conflicts. The selection frequency maps indicated the most critical areas for meeting the objectives of priority ES of the management area. For instance, for the Available management area (MA type 1), in scenario 5, the water quality constrained scenario which was most representative of the current situation in the lake, areas close to towns and cities showed high frequency of selection (>50%). The core activities in this zone were fishing followed by fish breeding grounds. Parts near cities around Lake Victoria are most impacted by eutrophication (Kabenge, Wang and Li, 2016; Simiyu *et al.*, 2022), and by allowing activities that could potentially add organic loading (such as cage farming) within such areas, the potential for further lake degradation could increase. As a precaution, these sections of the lake could be avoided in future licensing activities. It is possible that species that breed in deeper waters such as *Lates niloticus* (Nile perch) (Breder and Rosen, 1966) might have a higher possibility of reproducing in deeper waters due to higher selection frequencies shown in open waters for features of the Protected management area (MA type 3) involving fish breeding and nurseries. For other littoral breeders like tilapia, more grounds for fish nurseries needed to be set up if targets were to be met. By visually interpreting the results, planning for licensing activities on the lake could be improved.

Irreplaceable areas represented the PUs selected several times (out of all the 10,000 runs done) to contribute to the optimal solution and this was based on Vallecillo et al. (2018). These were critical in guiding the identification of the most conflict-prone parts of the lake if management area network targets were to be met. The amount of space that had to be secured for each management area network was calculated. For instance, with an ecosystem health constraint (scenario 5), more space would be required if targets for features in the Protected management area (where breeding grounds were a priority) were to be met. Large irreplaceable areas meant limited flexibility in the formation of the management area network i.e., very limited alternative solutions (Vallecillo *et al.*, 2018). According to the estimate, in scenario 5, the Protected Management Area (MA type 3) needed 399.7km², while the Target Management Area (MA type 2) (where cage farming was prioritized) required no irreplaceable PU, and the Available Management Area (MA type 1) required only 10.34km². Consequently, given the water quality condition in the lake, more space might need to be set aside for priorities in MA type 3 (fish nurseries) than in MA type 2 (cage farming) or MA type 1 (fishing).

5.3 Conclusion

A priori spatial optimisation contributes to more cost-effective solutions in EBM, aids fewer trade-offs and co-benefits can be mapped. This approach allows for placement of safeguards for ecosystem integrity prior to the occurrence of extremes such as species extinctions. Using spatial prioritisation networks that allow management interventions to be varied would contribute to mitigating current and future challenges in the ES of Lake Victoria. With data availability on actual economic value of an ES, this framework could be used for cost benefit analysis in resource planning. In future research, other drivers of environmental change, the spatial & temporal changes caused by these drivers, as well as other determinants of state can be considered. The use of proxy data, plausible ranges, scenarios, or environmental model outputs provides an alternative when actual information about the ES is not available. It has been demonstrated through the application of spatial prioritisation that aquaculture can be planned effectively within the framework of other ecosystem services. Due to the large populations dependent upon these lakes for their livelihoods, this approach should be adopted for cage culture in African Great Lake systems. A major challenge lies with the space available for fish breeding & nursery grounds and recreation services in the current ecosystem health state. The MarZones software allowed for inclusion of various costs that had to be optimized, thus providing cost effectiveness solutions where competing objectives were involved. With actual financial cost estimates, it is a tool that could enable implementation of cost benefit analysis and the 'Polluter Pays Principle' stated in the NEMA Act 5 (NEMA, 2019), Section 81 (5).

CHAPTER 6

ASSESSING THE IMPACTS OF CAGE CULTURE ON SEDIMENT AND WATER QUALITY IN LAKE VICTORIA, UGANDA

6.1 Introduction

Cage farming is reported as a source of nutrients and organic waste, chemicals (such as therapeutics, or antimicrobials), feral fish, bacteria, and parasites (Nobile *et al.*, 2020). The problem of eutrophication in freshwater bodies is exacerbated by a combination of low water flows, a long residence time, and low dilution rates. (Beveridge, 2008). Organic wastes from cage farming contain carbon, nitrogen, and phosphorus in dissolved (soluble) or particulate form (waste feed, and faecal matter) (Azevedo *et al.*, 2011). Several factors affect the uptake of feed, including size, shape, appearance, palatability, water quality, facility design, and feeding practice (Cho and Bureau, 1998; Herath and Satoh, 2015). When ingested, the feed is digested and absorbed into the bloodstream. The undigested portion and metabolic outputs are released as waste (Cho and Bureau, 2001). Similarly, during feed administration, a portion is lost in the water column (Beveridge, 2008). Solid waste feeds and faecal matter make up the bulk of organic wastes that settle and impact cage sites.

Phosphorus contained in waste is a major concern in freshwater ecosystems. This phosphorus contributes to photosynthesis by phytoplankton when available in the upper layers of the water column (epilimnion). An increase in phosphorus concentration is linked to accelerated eutrophication, and the proliferation of harmful blue green algae (Cooperrider *et al.*, 2020). Eutrophication has been linked to anoxic conditions and fish kills in shallow waters below 20 meters in Lake Victoria (Kolding *et al.*, 2008). Therefore, monitoring water and sediment quality is required to mitigate the additional risk of organic enrichment in cage culture operations currently taking place in shallow semi-enclosed bays.

One of the current strategies authorities employ to estimate cage impact on the lake is periodic monitoring of water quality. This yields a large volume of scientific information often embedded in technical reports (such as NaFIRRI, 2014; Egessa *et al.*, 2017). Monitoring information is rarely used by decision makers; summarizing it into key indices of change in water or sediment quality ensures more effective communication and utilisation (Tyagi *et al.*, 2013; Yan *et al.*,

2022). An overall score that shows the level of acceptability for a particular purpose is calculated from physical, chemical, and biological variables assessed in environmental monitoring (Banda and Kumarasamy, 2020). For a given use, such as supporting aquatic life or suitability for potable water, this score is used to classify the wider ecosystem as poor, marginal, fair, or good. CCME (Canadian Council of Ministers of the Environment, 2001) is a common water quality index, upon which other indices are developed for monitoring or legislation purposes, such as those developed by Feng *et al.* (2018); Vasistha *et al.*, (2020); Kikuda *et al.*, (2022) and Gamvroula and Alexakis (2022). In evaluating aquaculture impacts, a common approach utilised at farm level, is the assessment of key parameters attributed to changes in the food web. In freshwater systems, this could be the total nitrogen or phosphorus concentration or the concentration of chlorophyll *a*. The Tilapia standard used by the Aquaculture Stewardship Council (ASC, 2019) measures the chlorophyll *a* concentration of a receiving oligotrophic water body. The challenges arise when aquaculture is taking place in a system with high nutrient concentration (e.g., a eutrophic lake), added from within the lake itself or from the catchment.

Regardless on it being earmarked for cage farming, the shallow and inner bays and gulfs of Lake Victoria have been classified as eutrophic (Gikuma-Njuru *et al.*, 2005; Hecky, 1993; Kabenge, Wang and Li, 2016; Mugidde *et al.*, 2005; Okechi *et al.*, 2022). Second, the lack of accredited testing laboratories and trained personnel in the East African region (Kang'ethe *et al.*, 2021) is a major threat to effective environmental monitoring. Even with written guidelines or legal provisions for monitoring, the process is expensive and rarely implemented. Assessment results are often complex and difficult to interpret for decision makers; thus, monitoring information rarely makes it into management plans (Biber, 2013).

Furthermore, various catchment processes simultaneously happening in proximity to cage culture farms contribute their own nutrient load to the water column. Establishing the fate of cage-generated wastes becomes complicated. Segregating waste loading from catchments and other sources from that generated at cage farms is essential. However, implementation of known techniques, such as isotope analysis (Alp and Cucherousset, 2022), is limited by technical capacities. The monitoring of environmental quality at cage farms must be robust, precise, and cost-effective to facilitate licensing and ensure production does not go beyond site assimilative capacity.

In this study, water and sediment quality at a cage fish farm is compared to conditions at the control site to determine if there is a significant difference. To reduce information and ease communication, a quality index is created. In addition, a benthic macroinvertebrate analysis is carried out and conditions are compared for the cage and control sites. Finally, using nutrient

mass balance models, fluxes of C, N and P nutrients are modelled. The aim of this study is to evaluate the three approaches to organic waste assessment at cage sites. This will contribute to effective monitoring of farms with the objective of setting production limits. Using a commercial production site as a case study, the most robust, precise, and cost-effective option(s) for on-farm licensing decision support is explored. This study is not discrediting the importance of any of the other techniques. Instead, it explores the most appropriate approach for setting on-farm production limits, specific for eutrophic lakes.

6.2 Materials and Methods

6.2.1 Sampling site

The study was conducted at a commercial cage fish farm located at Latitude 0°03'16.38" N, Longitude 32°33'18.93" E; 1135m elevation in Garuga, Tende Bay on Lake Victoria (Figure 6.1). The farm has commercially produced Nile Tilapia (*Oreochromis niloticus*) since December 2014 in square production cages (6x6x6) m. Located 600m from the lake shoreline in the semi-enclosed bay, with each cage having an average cage net depth of 6m and suspended 3m to the lake bottom. Within an average 8months production cycle, the site started with an average of 7tons in 2015, selling the fish locally within towns and cities.

Samples of water and sediment were taken at the cages and at a control site 300m away from the cages in replicates in October & December 2019 and January to February 2022 (Figure 6.2). Sampling was done at predetermined production cages and control points along a depth profile of 1m, 5m and at 10m at each site. The average total depth at the cages and control was 11.2m and 10.6m respectively. In situ measurement of physico-chemical parameters of dissolved oxygen, pH, temperature, conductivity, redox potential were taken using portable metres. Light attenuation was measured using a Secchi disk and total depth was measured using an Echo sounder.

Sampling for water and sediment, preparation, storage, and analysis were done following Stainton, Capel and Armstrong, (1977). Water using a 2-litre Van Dorn water sampler, water was taken at 1m, 5m and 10m depth at the cages and control sites in three and two replicates respectively. It was poured into 500ml translucent polyethylene screw-cap Nalgene pre-acid washed bottles that were pre-rinsed in sample water and were kept in a cooler away from light and transported to the laboratory for preparation (Figure 6.3).

Water samples for dissolved nutrient Soluble Reactive Phosphorus (SRP), were filtered through Whatman 42mm GF/C glass fibre filter within 4hours. The unfiltered water was kept for the analysis of Total Nitrogen (TN) and Total Phosphorus (TP).

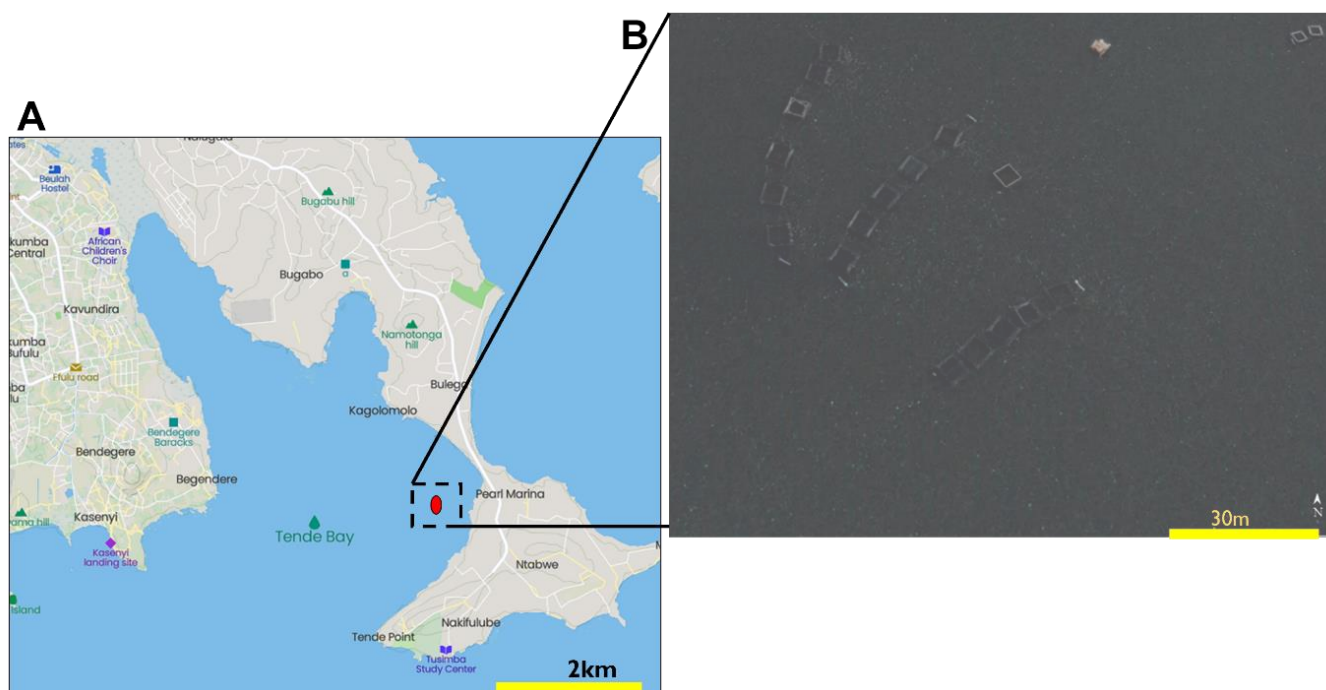


Figure 6.1: Location of sampling cages in Tende bay, Gerenge (A) and a zoomed in Google image of the production cages at which sampling was done (B)

6.2.2 Water sample analysis

Analysis for TN, SRP, TP, BOD and Ammonium in water were done at the National Fisheries Resource Research Institute, Jinja in Uganda, using spectrophotometric methods (Stainton, Capel, and Armstrong, 1977) . This involved the phenol- hypochlorite method for ammonia, the acidified molybdate method for SRP and the persulphate digestion method for TN and TP. For each analysis, a set of standards was prepared and treated like the samples. For SRP, to 10mls of filtered in a plastic vial, 2mls acidified molybdate solution was added. Time was allowed for colour development and reading done on a UV spectrophotometer (model UV-18000) at a wavelength of 885nm.

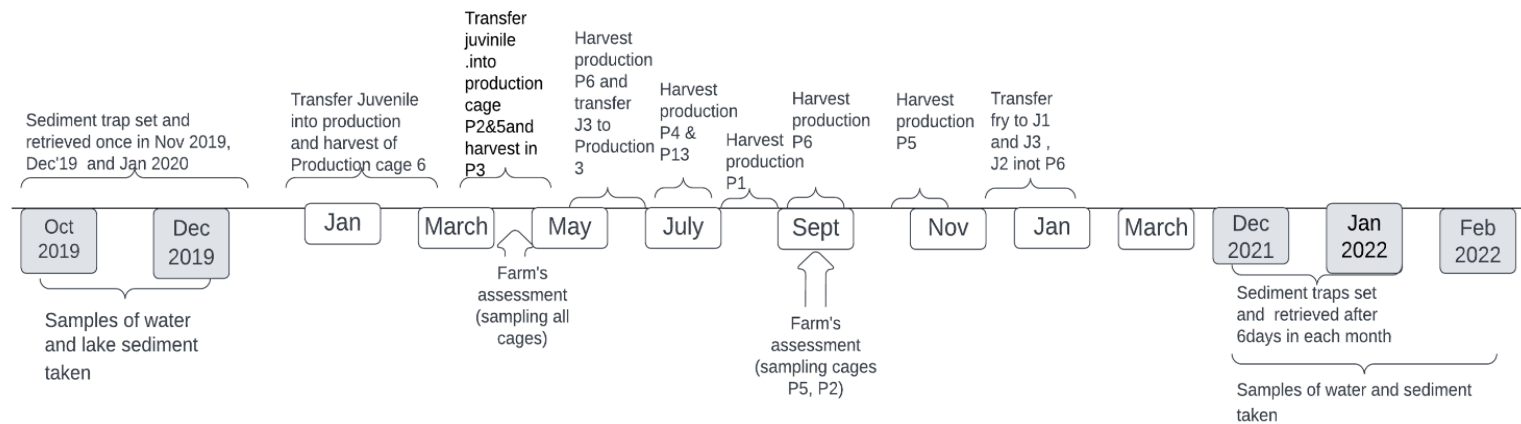


Figure 6.2: A chronology of activities of the farm within which sampling for water and sediments was done (in shaded months).

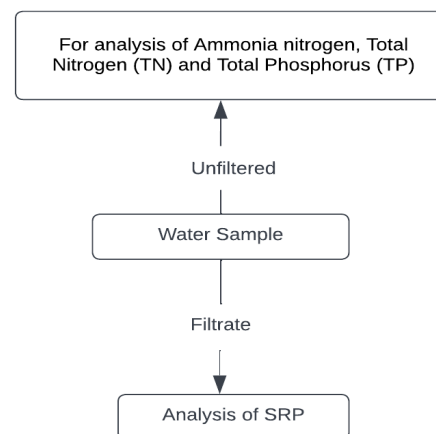


Figure 6.3 Apportioning the water sample drawn at each sampling depth for nutrient analysis (adapted from Stainton, Capel, and Armstrong, (1977))

For Ammonium-nitrogen analysis, to 10mls of filtered water sample, 0.4mls of phenol solution was added, followed with 0.4mls of sodium nitroprusside solution. 1mls of oxidizing solution was added and the mixture covered & kept away from light for an hour to allow for colour development. Using a spectrophotometer at a wavelength of 640nm, reading was taken.

For TN, unfiltered water samples and standards in the range of 200, 500, 800, 1000, 2000, and 4000, 5000 $\mu\text{g/l}$ was prepared. 10mls of each was transferred into digestion bottles to which 0.4gms of sodium persulphate was added. Digestion was done for 1 hour at 120°C in autoclave at a pressure of 1.2Kpas. The mixture was left to cool, 0.25gms of Devardas alloy was added and the mixture was left to stand overnight. 1ml of the solution was pipetted off and topped with deionized water to make 10mls. Analysis was done as for ammonium-nitrogen analysis.

For TP, unfiltered water samples and standards in the range of 200, 500, 800, 1000, 2000, and 4000, 5000 $\mu\text{g/l}$ was prepared. 10mls of each was transferred into digestion bottles to which 0.4gms of sodium persulphate was added. Digestion was done for 1 hour at 120°C in autoclave at a pressure of 1.2Kpas. The mixture was left to cool then diluted to 2 times. 5mls was pipetted and diluted with 5mls distilled water to make 10mls. To the mixture, 2 drops para-nitro phenol solution was added. Titration was done using 4M (0.42mls) sodium hydroxide solution till permanent yellow colour was obtained. The yellow colour was eliminated by adding one drop of 20% sulphuric acid. Colour reagent was added, and analysis followed as for SRP.

6.2.3 Sediment sampling and analysis

Sediment samples were collected from the lake bottom and in sediment traps set under the cages and at a random reference site, 300m away from the cages for C, TN, and TP analysis. The lake bottom samples were collected using an Ekman grab at the cage and control site. The sediment traps were fabricated using locally available material (Figure 6.4). Each trap had four arms to which a PVC 6cm cylindrical tube were tied. On each arm, starting from the sampling bottle, a diameter 500 mL Nalgene container was attached. The ratio of the bottle was set to fit the adapter socket that connected it to the pipe. The gap between the connector and the bottle was tightened using Plumber's Teflon (PTFE) tape.



Figure 6.4: Sediment trap fabricated using locally available material. The painted iron bars were susceptible to rusting if available steel provides a better option.

To the connector, a small hole was drilled and covered with tape on deployment. This was removed to let water out on retrieving the trap before taking the sampling bottles off. To avoid losing the bottles, a sleeve was added to hold them in position. Two traps were set, one under the cages lowered with a nylon rope that was tied to opposite ends of the cage's PVC pipe to hold it in place while another was set at a reference site, lowered by a rope, and kept in position with a heavy rock improvised as a sinker. To the top, a buoy was attached to easily locate the trap for retrieval. For two sampling periods, the traps were retrieved once every month in November, December and January 2019 & December, January, and February 2022 after staying for 6 days. Sediment samples collected from the lake bottom under the cages and at the control site, taken to the Uganda Fisheries laboratory, Entebbe, and dried in an oven for 24hrs at 80°C and weighed. Further analysis for Total Nitrogen, Carbon and Total Phosphorus on the sediment samples was done at the Institute of Aquaculture, University of Stirling.

For TN analysis, the sediment sample was pre-dried in oven at 50°C for 2 hours. It was ground into a very fine powder to allow maximum digestion. 0.02g was weighed off using a sensitive analytical balance. The weighed powder was transferred into digestion bottles then dissolved in 10mls of fresh distilled deionized water. Standards in the range of 2,000 5,000, 8,000 10,000 20,000 40,000 50,000 µg/l was prepared and 10mls measured off and transferred into digestion bottles. To all digestion bottles, 0.4g of persulphate was added and the mixture

autoclaved for 2 hours at 120°C. The mixture was left to cool and 0.25g of Devardas alloy added. The mixture was left to stand overnight at room temperature for reduction to take place. The subsequent aliquots were transferred using a pipette to clean vials, using a dilution of 100. To this 10mls of deionized water was added. Samples were analysed using phenol method for ammonium nitrogen described above. Final concentration was derived using regression analysis but available N in sediment was expressed as %N.

For TP in sediment, the sample was pre-heat in an oven at 50°C for 2 hours. It was ground into very fine powder to allow maximum digestion. 0.02g was weighed off using sensitive analytical balance. The weighed powder was transferred into digestion bottles and dissolved in 10mls of fresh distilled deionized water. Standards in the range of 500 1,000, 2,000 5,000 8,000 10,000 µg/l was prepared from which 10mls was measured and transferred into digestion bottles. To all digestion bottles, 0.4g of persulphate was added the mixture autoclaved for 2 hours at 120°C. The mixture was a left to cool and diluted 10 times. 2 drops of para-Nitrophenol were added and titration done with 4M NaOH till permanent yellow colour was obtained. One drop of 20% sulphuric acid was added to eliminate the yellow colour. A colour reagent was added, and analysis followed as for SRP. The final concentration was calculated using regression analysis but available P in sediment was expressed as %P.

For organic matter analysis, subsamples (2cm³) of sediment were placed in pre-weighed ceramic crucibles and weighed again to obtain weight of wet sample and crucible. The sample was then dried overnight (15 hours) at 105°C in an oven, allowed to cool in a desiccator and weighed to get the dry weight (DW₁₀₅). Next, the samples were heated to 550°C in a muffle furnace for 4 hours and allowed to cool overnight. The samples were then placed in a desiccator, cooled to room temperature, and weighed again to obtain ignition weight (DW₅₅₀). The percent change in mass from the initial drying to the final combustion was the loss on ignition (LOI) value (Equation. 6.1.)

$$\text{Loss on Ignition} = \frac{DW_{105} - DW_{550}}{DW_{105}} \times 100 \quad \text{Equation 6.1}$$

For sampling of benthic macro-invertebrates, a Ponar grab sampler (open jaw area of 238cm²) was used to collect two hauls from each sampling point. These were released into a basin, concentrated and a sample drawn into a labelled sample bottle containing 5% formalin solution, a preservative. In the lab, samples were rinsed with water and individually placed in

a flat-bottomed white tray using a pair of forceps. Identification was done under a binocular microscope at a magnification of x25 using Mandahl Barth, (1954); Ravera, (2013).

6.2.4 Data Analysis

6.2.4.1 Water and Sediment quality

For the water quality physico-chemical data, statistical analysis was done using SPSS (v. 28.0.0.0). All data was tested for normality using Shapiro-Wilk ($p < 0.05$) test. Where normality conditions were met, Student t-test was used to compare conditions between the cage and control sites. To compare differences along the depth profile using one-way ANOVA test was used. Those that were not normally distributed, natural log transformation was done. Where normality still failed after the log transformation, non-parametric Mann Whitney test at $p < 0.05$ was used to assess the difference between sites. To compare changes in sediment TC, TP and TN over the years, Friedman test was run. All statistical significance was set at $p > 0.05$.

For the benthic macro-invertebrates, Shannon-Weiner diversity index (H')(Shannon-Weiner & Weaver, 1949)(Equation 6.2) was used to estimated differences in diversity between sites.

$$H' = \sum_{i=1}^R p_i \ln p_i \quad \text{Equation 6.2}$$

Where p_i is the proportion of individuals belonging to the i^{th} species

Throughout the calculation, log (base 2) was used. Effectiveness, a measure of diversity was calculated as $\exp(H')$ (Gardener, 2014). Species richness (S) was simply the total number of species at the site while Heip's Evenness Index (HE)(Heip, 1974) (Equation 6.3) that indicated how evenly species richness contributed to the biomass between the two sites was calculated as:

$$HE = \frac{\exp(H') - 1}{S - 1} \quad \text{Equation 6.3}$$

6.2.4.2 Aquaculture score

The aquaculture score (index) was generated to reduce the water quality data into a single value that could be easier to interpret and communicate. It was a transformation of quantitative data into a qualitative scale, more appreciable for decision support. The index was based on

the CCME methodology (CCME, 2001). It utilised three aspects showed in Equations 6.4-6.9; the scope ($F1$)- the number of variables that do not meet the water quality objective; the amplitude ($F3$)- the amount by which the objectives are not met, and the frequency ($F2$) which is the number of times the objectives are not met (CCME, 2001). The objective of the index was support for aquatic life.

$$F1 = \left(\frac{\text{Number of failed variables}}{\text{Total number of variables}} \right) \times 100 \quad \text{Equation 6.4}$$

$$F2 = \left(\frac{\text{Number of failed tests}}{\text{Total number of tests}} \right) \times 100 \quad \text{Equation 6.5}$$

For $F3$ the calculation involves first, establishing the number of times by which each parameter concentration is greater or less than the value of the objective also known as the *Excursion*

$$\text{Excursions}_i = \left(\frac{\text{Objective}_j}{\text{FailedTestvalue}_i} \right) - 1 \quad \text{Equation 6.6}$$

Second, calculating the sum of the excursions and dividing by the total number of tests to generate the normalised sum of excursions (nse)

$$nse = \frac{\sum_i^n \text{Excursion}_i}{\text{Number of tests}} \quad \text{Equation 6.7}$$

$$F3 = \frac{nse}{0.01nse + 0.01} \quad \text{Equation 6.8}$$

$$\text{Water Quality Index} = 100 - \left(\frac{\sqrt{F_1^2 + F_2^2 + F_3^2}}{1.732} \right) \quad \text{Equation 6.9}$$

The variables used were DO, temperature, pH, conductivity, redox, Ammonium, TP in water, TSS, and turbidity. Due to lack of a national standard for water quality for support of aquatic life at the time of this study, to generate the index, guidance on parameter ranges were sourced from various references (Table 6.1). but the overall goal was that the optimal value prescribed as fit for survival of aquatic life. The outcome was compared to the CCME index ranges (Table 6.2.) to generate a score showing conditions at the cages and control sites.

Table 6.1: Parameters for Water quality index generation.

Parameter	Ranges	Reference
DO (mg/l)	5-6mg/l	Alabaster & Lloyd, (1982); CCME, (2001); USEPA, (1986)
Temp(°C)	29.4-32.2°C (Upper limit)	Alabaster & Lloyd, (1982)
pH	6-9.5	Alabaster and Lloyd (1982) and USEPA (1986), CCME (2001)
Cond. µScm ⁻¹	150-800	Huron River Watershed Council, (2013)
Redox(mV)	(400)- 800mV	Sigg, (2000)
Ammonium (mg/l)	0.0026-0.05mg/l	USEPA, (1986)
TP-water (mg/l)	0.035-0.1mg/l	CCME, (2001)
TSS (mg/l)	Max increase of 25mg/l from background levels	CMEC, (2001)
Turbidity	Max increase of 2NTU from background levels	CCME, (2001)

Table 6.2: CCME index ranges and the corresponding interpretation (CCME, 2001)

Index Range	Category	Interpretation
0-44	Poor	Water quality is poor, conditions always depart from natural levels
45-64	Marginal	Water quality is threatened often depart from desirable conditions
65-79	Fair	Sometimes conditions depart from desirable levels
80-94	Good	Conditions rarely depart from natural desirable levels
95-100	Excellent	Conditions close to pristine

6.2.5 Nutrient Balance Model

Waste flux from the cages were estimated using nutrient balance models based on feed input, feed composition and output. To establish the amount of P, Equations 6.10-6.16. were followed (Varol & Balci, 2020). Appropriate values were used in the same equations to establish C and N fluxes (Table 6.3). Results from the mass balance models were compared to observations taken on samples collected in sediment traps that were retrieved every 6 days. Other parameters used are presented in Table 6.4. The same results were compared to other studies done on Tilapia grown in similar conditions.

Table 6.3: Description of variables and values used in the nutrient balance model.

Description	Unit	Value	Source
Feed nutrient Content			
Nitrogen	C _N	8%	Beveridge, (2004)
Phosphorus	C _{PH}	1.2%	From manufacturer
Carbon	C _C	44.6%	Gondwe <i>et al.</i> , (2011a)
Digestibility coefficients			
Protein	ADC _P	92.60%	Köprücü & Özdemir, (2005)
Phosphorus	ADC _{PH}	54.4%	Montanhini Neto and Ostrensky, (2015)
Carbon	C _{RR}	72%	Montanhini Neto and Ostrensky, (2015)
Body nutrient content			
Nitrogen	B _N	3%	Beveridge, (2004)
Phosphorus	B _P	0.34%	Beveridge, (2004)
Carbon in fish	C _{FF}	49.2%	Gondwe <i>et al.</i> , (2011a)
Farm Management			
Feed Used	FG	82,856kg (2019/2020) & 88,830Kg (2020/2022)	Farmer's records
FCR	FCR	1.5	Provided by farmer
Feed waste	FW	20%	(Montanhini Neto and Ostrensky, 2015)-revised from 18%

$$Uneaten\ P\ (U) = (FG * \%FW) * \left(\frac{C_{PH}(\%)}{B_P} \right) \quad \text{Equation 6.10}$$

$$Consumed\ P\ (C) = (FG - (FG * \%FW)) * \left(\frac{\%C_{PH}}{B_p} \right) \quad \text{Equation 6.11}$$

$$Faecal\ P\ (F) = (FG - (FG * \%FW) * \left(\frac{C_{PH}(\%)}{B_P} \right)) * (1 - ADC_{PH}) \quad \text{Equation 6.12}$$

$$\text{Digested } P (D) = FG * \left(\frac{B_P}{FCR} \right) \quad \text{Equation 6.13}$$

$$\text{Solid } P = \text{Feacal } P + \text{Uneaten } P \quad \text{Equation 6.14}$$

$$\text{Dissolved } P = C - F - D \quad \text{Equation 6.15}$$

$$\text{Total } P (Y) = \text{Solid } P + \text{Dissolved } P \quad \text{Equation 6.16}$$

Table 6.4: Values used to calculate the percentage of nutrient waste in the sediment traps set under the fish cages.

Variable	Value
Number of days traps were deployed	6
Cage Area (m ²)	36
Sediment trap Area (m ²)	0.003
Average dry weight of sample in the trap (g)	2.6

6.3 Results and discussion

6.3.1 Water Chemistry

Physico-chemical tests done for the sampling months of October, December 2019, and January and February 2022 were compared at the cage and control site (Table 6.5). The Shapiro-Wilk normality test indicated temperature, redox potential, DO, and TN-water were normally distributed. The student t-test (Table 6.6) was used to assess differences between sites. Apart from redox potential, the means between the cage and control sites were not significantly different for other parameters. Conductivity, pH, Ammonium, SRP, TP-water and TSS were not normally distributed, hence, to check for difference between the sites, the non-parametric Mann Whitney test was done (Table 6.7). Similarly, among this set of parameters, there was no significant difference in conditions between the cage and control sites. Results are presented as mean and the standard deviation while the charts error bars represent standard error.

The mean temperature at the cages was 26.3±1.23°C compared to the control at 25.8±0.65°C. However, the difference between sites was not statistically significant (p=0.117). October average temperature at the cage and control sites were 26.0 ± 0.5°C and 25.6 ± 0.6 °C while December averages were 25.9±0.3°C and 25.8±0.4°C respectively. In December,

temperature was the same at the surface but higher at the cage site than control site as depth increased. The mean difference between temperature at the surface and in the deeper layers (10m) was not statistically difference ($p=0.101$).

Generally, the average temperature was within range for survival of most freshwater fish species (Alabaster and Lloyd, 1982), and that for optimal for Tilapia growth ($22-30^{\circ}\text{C}$) (McGinty et al, 1989; Nivelle *et al.*, 2019). The lake is monomitic, with complete mixing occurring once a year between June and August (Talling, 1966; Nyamweya *et al.*, 2016). However, no sampling was done in this period due to COVID19 pandemic restrictions.

Dissolved oxygen depletion is a key measure of the effects of organic enrichment in freshwater systems (Gondwe et al 2011b; Hellawell, 2012). The average DO at the cages was $6.507\pm 2.10\text{mg/l}$ while the control site had an average of $7.20\pm 1.81\text{mg/l}$ (Figure 6.5). The difference in mean DO between the cages and control was not statistically significant (Table 6.6). At an average temperature of 26°C , oxygen levels in the water column at the control and cage sites were approximately 97% and 94% near saturation respectively.

Compared along depth profile, the difference in mean DO between the surface (0.5m), midwater (5m) and deeper (10m) layers was statistically significant ($p=0.008$) (Table 6.6) with a surface average of $7.63\pm 1.09\text{mg/l}$ and a bottom average of $5.39\pm 2.67\text{mg/l}$. The difference in DO along the depth profile could be related to the oxygen demand created by settling organic matter.

Dynamics of DO are very important in intensive cage culture, but mostly because production cages at the study site were in the shallower, protected inshore regions prone to algal blooms (Kabenge, Wang and Li, 2016) . Reduced DO levels have been linked to massive fish kills (Chen et al 2020) which have been reported in wild stocks on Lake Victoria (Ochumba, 1990; MAAIF, 2021) .

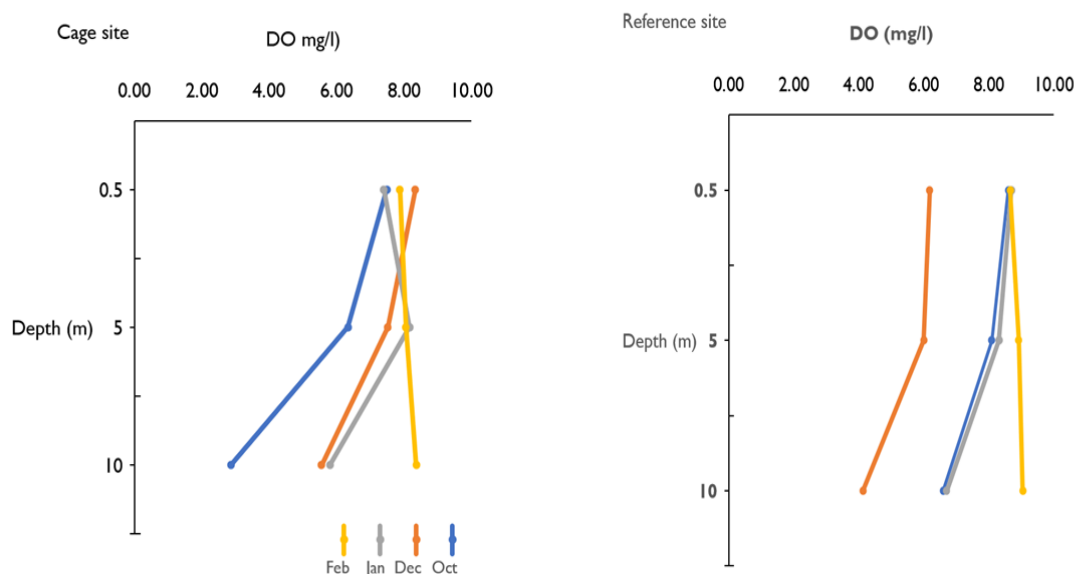


Figure 6.5: DO concentration along the depth profile at the cage and reference sites. December at the reference site exhibited levels less than any other month while cage DO levels were within range with other months, lowest in October in deep layers.

Dissolved oxygen distribution in the lake is highly correlated with the seasons of stratification in the lake occurring diurnally, seasonally, and annually (Talling, 1966; Hecky *et al.*, 1994). Most of the shallower parts of the lake (depth $\leq 10\text{m}$) and bays in the Ugandan section of the Lake are protected from strong winds by presence of islands (Talling, 1966), and the influence on dissolved oxygen and the thermal gradient is determined by diurnal variations with night-time dissolved oxygen levels sharply dropping due to limited algal photosynthetic activity (Hecky *et al.*, 1994; Ssebuggwawo *et al.*, 2000). The influence of the land and lake breeze plays a key role in mixing in these parts of the lake (Ssebuggwawo *et al.*, 2000).

During periods of thorough lake mixing in waters $>20\text{m}$ depth (Talling, 1966) in the months of July and August, the mixed layer can create uniform oxygen concentration (Hecky *et al.*, 1994). Similarly, dissolved oxygen distribution in the lake is highly correlated with the annual seasons of stratification in the lake (Talling, 1966; Hecky *et al.*, 1994). Surface waters in the past were found to be near saturation (94-100%) with atmospheric oxygen (Hecky *et al.*, 1994). However, with increased loading from the catchment (Kundu *et al.*, 2017) oxygen profiles will most likely change, especially during seasons of deep mixing in the lake or heavy rains.

The average BOD at the cage site was $1.27 \pm 0.94 \text{mg/l}$ compared to $1.18 \pm 0.75 \text{mg/l}$ at the control. Although not statistically significant, BOD levels at the cages are affected by organic enrichment (Hellawell, 2012). Comparable levels between sites could be due to a high algal

abundance that die and decay with conditions of lower oxygen worse at night when no algal photosynthetic activity is taking place.

The redox potential was the only parameter that exhibited a statistically significant difference in the means between the sites, higher at the control; $+103.95 \pm 3.8\text{mV}$ than the cage site $+82.11 \pm 26.74\text{mV}$. Very low redox potential (-100 to -710mV) is linked to dissociation of Manganese and Iron-bound sediment P under hypolimnetic anoxic conditions (Out et al 2017)

The average Secchi depth at the cage was $1.60 \pm 0.35\text{m}$ than the control site; $1.56 \pm 0.35\text{m}$ on average. Highest levels were recorded in December, with an average of 2.1m , followed by February, January and lowest in October i.e., 1.70m , 1.64m and 1.40m respectively. Averages at the control site were highest in December averaging at 2.0m , followed by February at 1.50m , January at 1.33m and least on October 1.29m .

Early stratification has been reported in Lake Victoria, for the month of October (Ssebuggwawo *et al.*, 2000; Deirmendjian *et al.*, 2021) while December exhibits deep thermal stratification (Nyamweya *et al.*, 2016), this could explain the trends seen in DO, and Secchi depth between these periods, however, with no statistical significance between sites.

Table 6.5: Means of water quality variables at the cage and control sites for the sampling months of Oct. and Dec. 2019 and Jan. and Feb 2022

	Oct-19		Dec-19		Jan 2022		Feb 2022	
Parameter	Cage	Control	Cage	Control	Cage	Control	Cage	Control
DO	5.56±2.49	7.77±1.17	7.14±1.43	5.43±1.78	8.1±0.24	8.87±0.20	7.12±1.21	7.9±1.06
Temp (°C)	26.0±0.54	25.6±0.62	25.9±0.32	25.8±0.66	28.6±1.30	26.6±0.60	25.4±0.21	25.4±0.15
TSS (mg/l)	5.23±2.57	6.96±3.53	4.30±0.98	5.69±3.06	3.07±0.98	5.48±3.46	2.9±1.84	4.8±1.98
pH	8.4±0.44	8.6±0.26	8.7±0.20	8.4±0.26	8.46±0.00	8.67±0.98	9.36±0.29	9.8±0.15
Cond/μScm ⁻¹	100.21±0.70	101.12±0.70	100.87±0.72	102.9±3.61	100.5±0.00	100.3±0.00	96.7±0.15	96.5±0.61
Redox/mV	82.11±26.74	104.92±7.79	78±11.04	103.67±3.54	85.07±19.2	102.87±4.97	94.27±25.0	100.6±0.66
Ammonium(mg/l)	0.12±0.19	0.2±0.48	0.45±0.04	0.03±0.02	0.43±0.00	0.063±0.03	0.08±0.06	0.06±0.03
SRP (mg/l)	0.01±0.005	0.008±0.00	0.041±0.01	0.036±0.00	0.03±0.02	0.033±0.00	0.0085±0.00	0.02±0.02
TP - water(mg/l)	0.15±0.02	0.2±0.01	0.06±0.01	0.07±0.03	0.165±0.02	0.085±0.03	0.165±0.02	0.066±0.04
TN- water(mg/l)	0.22±0.10	0.25±0.09	0.81±0.03	0.89±0.05	0.93±0.19	0.704±0.32	0.76±0.38	0.740±0.27

Table 6.6: Mean and t-test for equality of means (p<0.05) between the cage and control sites for parameters normally distributed.

Parameter	DO (mg/l)		Temp(°C)		Redox/mV		TN-water	
Site	Cage	Control	Cage	Control	Cage	Control	Cage	Control
N	18	18	18	18	9	6	16	16
Mean value ± SD	6.51±2.09	7.20±1.81	26.34±1.23	25.82±0.65	82.111±26.7	103.950±3.7	0.473±0.33	0.583±0.32
2-sided p-value	0.299		0.117		0.071		0.352	

Table 6.7: Comparison of conditions at the cage and control sites for parameters tested using non-parametric Mann-Whitney U statistical tests. NH₄⁺, SRP, TP and TSS in mg/l while conductivity was in μScm^{-1}

		pH	Cond.	Ammonium	SRP	TP-water	TSS
Mean	Cage	8.6±0.48	99.789±1.53	0.220±0.21	0.018±0.01	0.140±0.04	4.41±2.1
	Control	8.8±0.64	100.817±3.00	0.102±0.27	0.024±0.01	0.102±0.05	5.66±3.04
	N	15	15	15	12	16	16
<i>p</i> <0.05	<i>p</i> -value	0.90	0.21	0.25	0.54	0.14	0.23

No significant difference was seen in Secchi depth between the cages and the control site. Even when Secchi depth transparency is a key parameter for monitoring cage production impact (Gondwe et al, 2011b), nutrients (N and P) seize to be a limiting factor for algal growth beyond certain limits (Downing and McCauley, 1992). Then, other factors like light could be limiting for algal growth. To an extent, the presence of cages sheds off light, compared to the exposed control area that receives more direct sunlight hence are more suitable for algal growth hence consistently lower Secchi depth in the control site (Figure 6.6). Deirmendjian *et al.*, (2021) reports a Secchi depth of $1.7\pm0.6\text{m}$ at near shore waters (8-31m depth). This value is within the same range as was established at the cage and the control sites. However, assessing Secchi depth in Lake Victoria as an indicator of organic enrichment from the cages is limited by the heavy pollution from allochthonous sources.

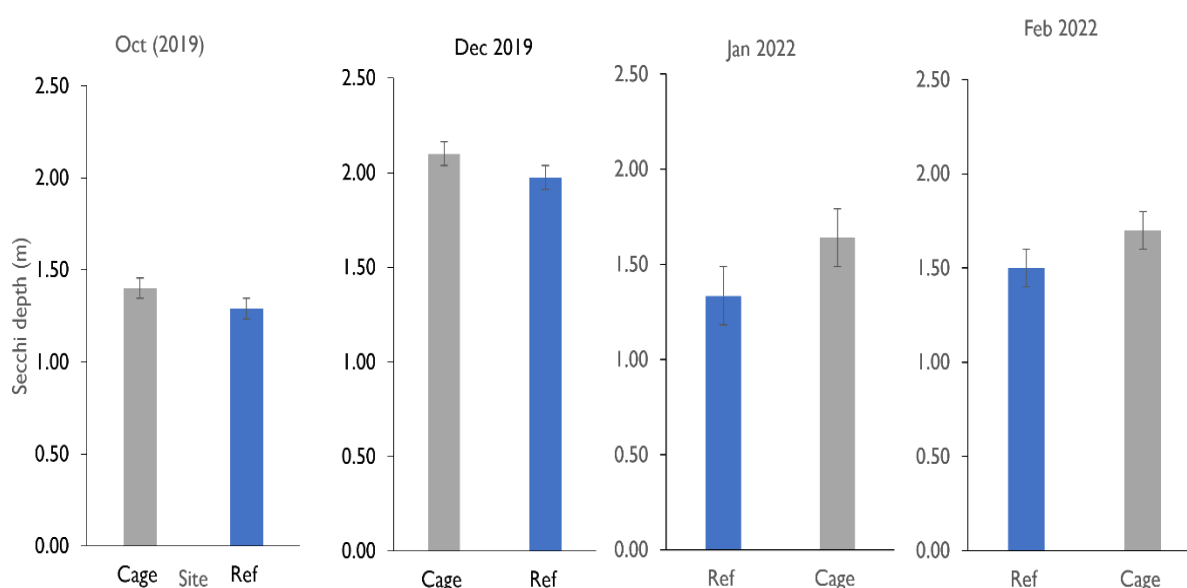


Figure 6.6: Secchi depth at the cage and control sites for the sampling months. December exhibited highest Secchi depth for both sites compared to any other month.

Average pH at the control and the cage site was 8.7 ± 0.6 and 8.6 ± 0.5 respectively (Figure 6:7), however, the difference in the means was not statistically significant (Table 6.7). October averages at the control site and the cage site were 8.6 ± 0.28 and 8.4 ± 0.37 respectively. In December, the control and cage site had pH levels of 8.79 ± 0.2 and 8.4 ± 0.1 whereas the cage site and the control were 9.4 ± 0.3 and 9.8 ± 0.2 in February respectively. February averages at the cage and reference sites are different from ranges reported for both October and December and this could be attributed to sampling using different equipment, whose calibration could have varied.

The pH ranges at both sites were over the safety range of 8.5-9 suggested by (Alabaster and Lloyd, 1982; Beveridge, 1987). Elevated pH levels in the sites could be linked to increased algal photosynthetic activity (Beveridge, 1987) stimulated by presence of nutrients and sunlight (with better exposure at the control).

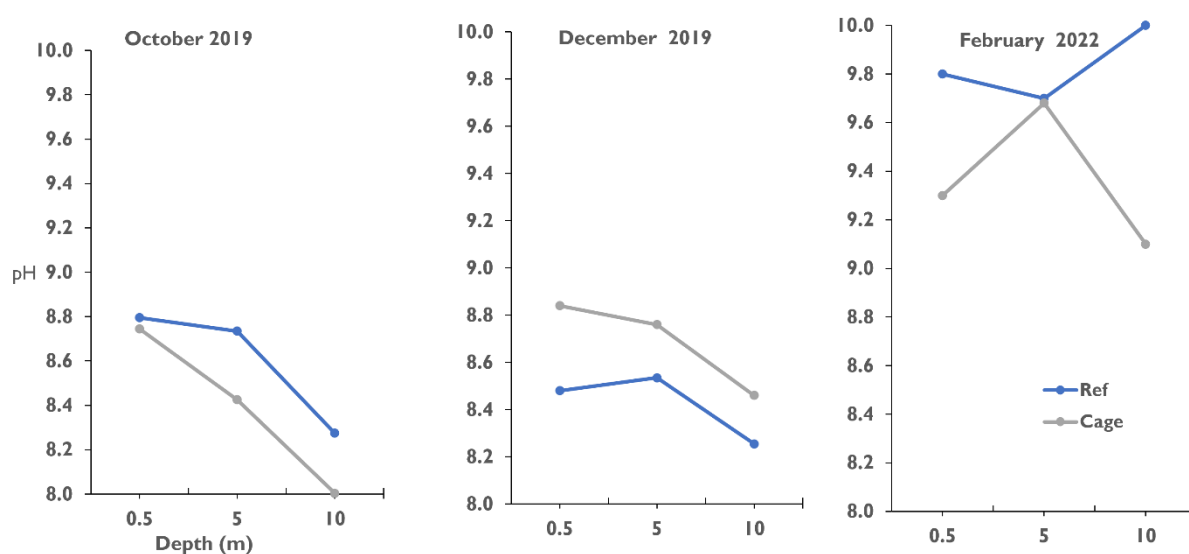


Figure 6.7: pH levels at the surface, midwater and bottom compared for the cage and control sites. February exhibited the highest averages in both sites over the sampling period.

Average conductivity at the control was $100.8 \pm 3.0 \mu\text{Scm}^{-1}$ while the cages had an average of $99.79 \pm 1.52 \mu\text{Scm}^{-1}$. The difference in the mean values was not statistically significant. In October, conductivity at the control was $101.12 \pm 0.71 \mu\text{Scm}^{-1}$ while the cage site average at $100.21 \pm 0.37 \mu\text{Scm}^{-1}$ (Figure 6.8) December for the control site and the cage site were $102.9 \pm 0.24 \mu\text{Scm}^{-1}$ and $100.9 \pm 0.72 \mu\text{Scm}^{-1}$ respectively. In February was $96.7 \pm 0.15 \mu\text{Scm}^{-1}$ and

96.5±0.6 μScm^{-1} for the cage and control sites respectively, the lowest recorded at both sites for the sampling period. For both sites, the top layer of the water column (0.5m) exhibited highest conductivity for the sampling period.

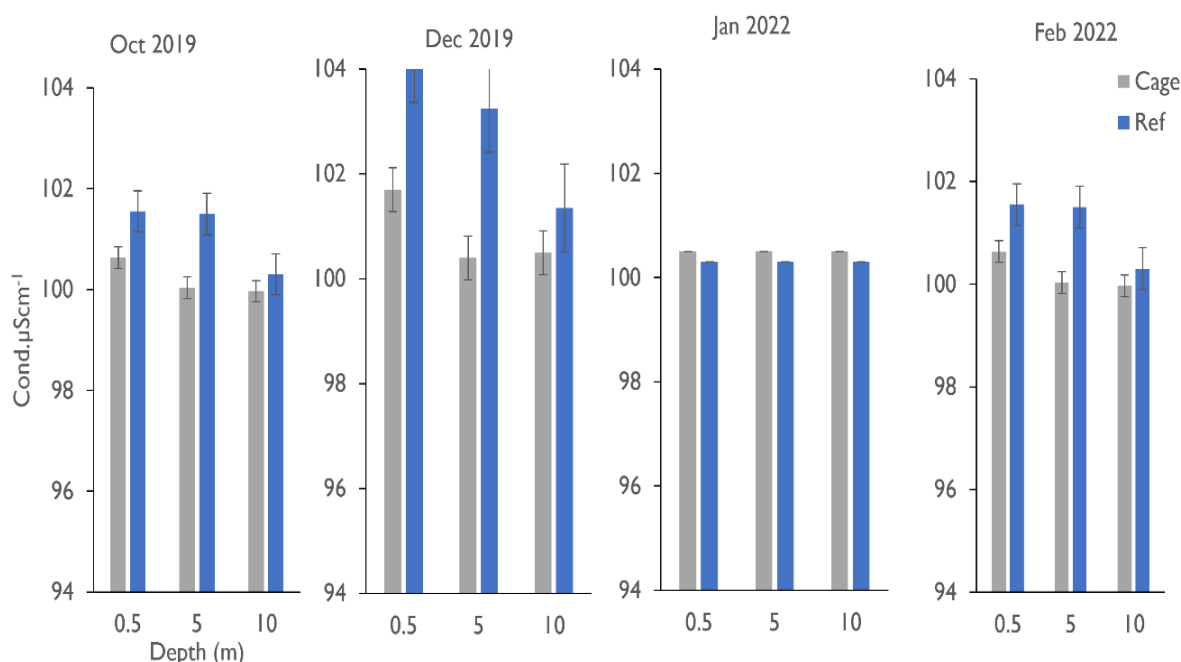


Figure 6.8: Comparing conductivity (μScm^{-1}) in the water column for samples taken Oct, Dec.2019 and Jan, Feb 2022

Average total suspended solids at the cages than the control site was 4.41±2.2mg/l and 5.66 ±3.04 mg/l respectively (Figure 6:9). Highest levels in TSS were recorded at a water depth of 10m in both sites. However, there was no statistical significance in the mean difference in the two sites (Table 6.7).

Comparing conditions in TSS before farm production commenced (baseline) (Figure 6:9), TSS levels are seen in December 2022 at the cage site. Correlating with Turbidity (USEPA, 1986), and Secchi depth (ASC, 2019), TSS, is one of the criteria used in monitoring cage impact on the environment. Higher TSS at the cages could be due to presence of waste feed, faeces, plankton, and mineral particles (Boyd, 2004). A lack of statistical significance in the TSS levels between the cage and the control site could be linked to allochthonous sources. These could include runoff from the catchment fostering a situation in the control site that's not very different from the cages. Kabenge, Wang and Li, 2016 has reported TSS levels as high as 50.2±2.61mg/l in a non-cage culture bay on the lake.

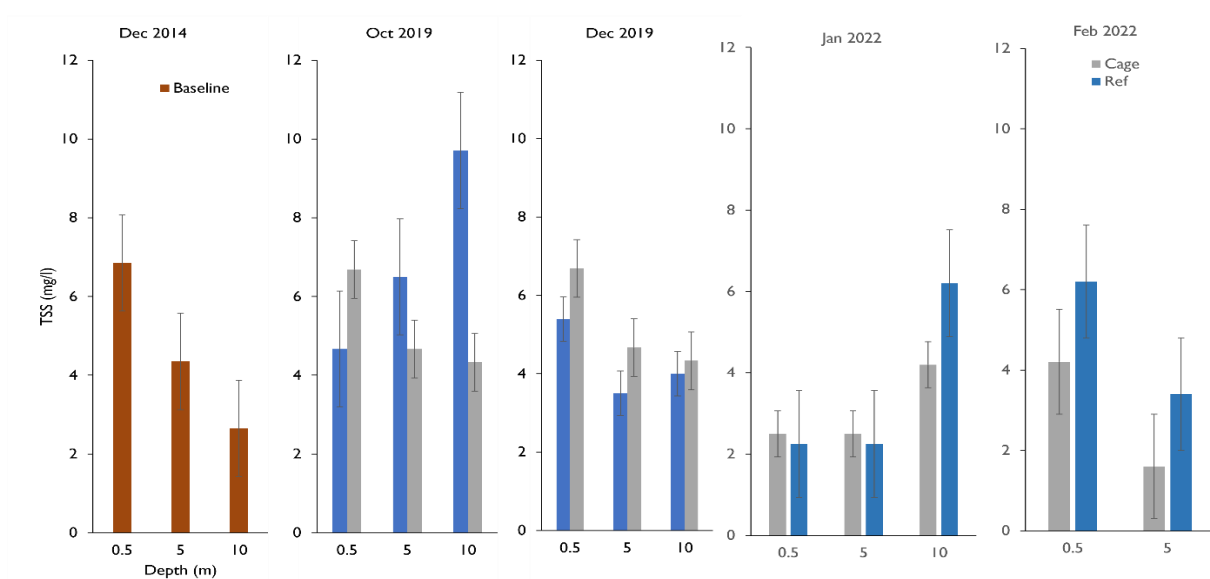


Figure 6.9: Total suspended solids at the cage and control sites. Comparison is made with the baseline conditions. In the deeper layers, for the months of Oct. and Jan, the control site exhibited high TSS. Ranges at the cages are still comparable with baseline conditions throughout the sampling period.

The results in the study fail to conclusively attribute changes in physico-chemical parameters in the water to presence of the cage farm. Similar findings of a lack of statistical significance in the difference of means in water quality variables of DO, TSS, Secchi depth, TC, TN, TP as tested at the cages and control sites were reported by Gondwe *et al.*, (2011a) who conducted an assessment on the impacts of cage culture on the water column Lake Malawi. A similar conclusion was reached by Kashindye *et al.*, (2015) who attributed dynamics in N and P, Chlorophyll a at a cage site on Lake Victoria (Tanzania) to seasonal land fluxes and phytoplankton activity rather than presence of the cages. The evidence was insufficient to support the existence of azoic sediments devoid of oxygen. This could be due to the quality feed used at the farm, the volume of production being within sustainable limits or a quick assimilation of the waste or uptake by wild fish (Gondwe, Guildford and Hecky, 2012). Generally, sites located in the shallow bays of the lake are prone to receiving catchment pollution. This coupled with highly dynamic internal conditions, and nocturnal mixing (Talling, 1966; Hecky *et al.*, 1994) means that comparing water quality conditions at cage and control sites alone is not enough while monitoring the impacts of cage culture in Lake Victoria. Similarly, because the farm where sampling was done had mean depth less than 20m, both for the cage and reference sites, mixing in such depths has been attributed to diurnal wind action (Ssebuggwawo *et al.*, 2000), rather than seasonal thermal stratification (Talling, 1966).

6.3.2 Nutrients

The average Ammonium levels at the cage and the control site were $0.15 \pm 0.2 \text{ mg/l}$ and $0.11 \pm 0.3 \text{ mg/l}$ respectively (Figure 6:10). October averages were $0.121 \pm 0.1 \text{ mg/l}$ at the cages and $0.051 \pm 0.1 \text{ mg/l}$ at the control site. At the cage site, concentration in Ammonium decreased with depth while for the control site, deeper layers exhibited higher Ammonium levels than in the upper water layer. In December, average Ammonium was $0.309 \pm 0.3 \text{ mg/l}$ and $0.027 \pm 0.01 \text{ mg/l}$ at the cage and control sites respectively. For February, average concentration at the cages decreased to $0.082 \pm 0.06 \text{ mg/l}$ and increased to $0.0604 \pm 0.03 \text{ mg/l}$ at the control site. The difference in average ammonium levels between the sites was not statistically significant (Table 6.7). A high Ammonium concentration at the cages could be attributed to the organic waste leaching from the cages in dissolved form. The toxicity of the un-ionised fraction of Ammonia (NH_3) increases with increase in temperature and pH (Alabaster and Lloyd, 1982; Levit, 2010), hence, care must be taken on farm due to high stocking densities in intensive cage production in tropical lake systems with high water temperatures and relatively high pH levels. Exposed to a lethal concentration of 2-4mg/l unionized ammonia for 24hrs, Evans *et al.*, (2006) reported 93-100% morality. Devi *et al.*, (2017) reports unionized ammonia of 0.02mg/l to be most ideal for warmwater fish culture.

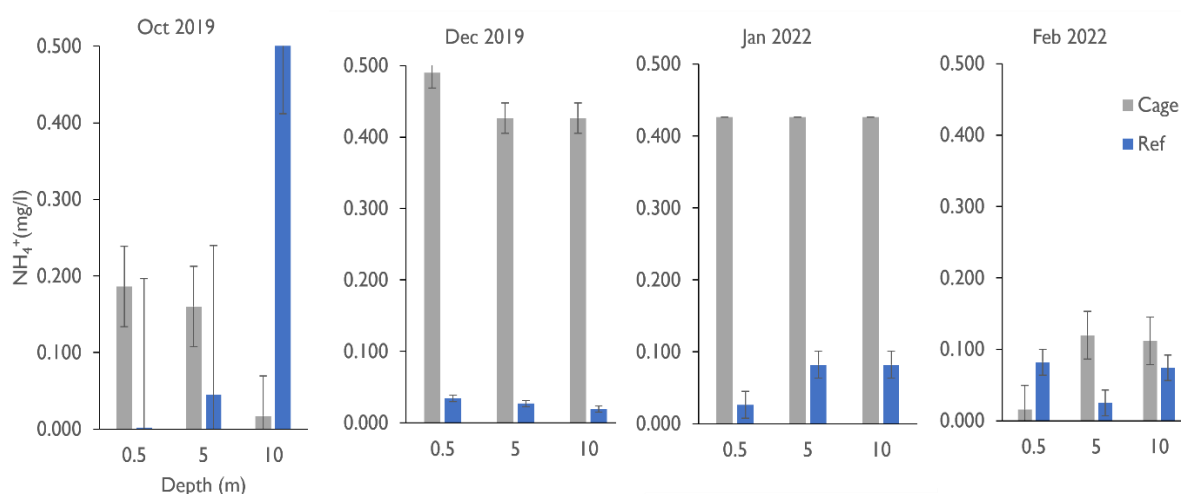


Figure 6.10: Ammonium concentration at the cages compared to the control sites, high levels in the former is attributed to organic waste deposition.

Average SRP concentrations in the water (Figure 6:11) column increased at the cages ($0.0176 \pm 0.015 \text{ mg/l}$) than the control ($0.024 \pm 0.142 \text{ mg/l}$). This difference over the sampling period was not statistically significant (Table 6.7). Average SRP at the cage site in October

was $0.0102 \pm 0.003 \text{ mg/l}$ while the control had $0.0082 \pm 0.003 \text{ mg/l}$. December average was $0.041 \pm 0.01 \text{ mg/l}$ and $0.036 \pm 0.00 \text{ mg/l}$ at the cage and control sites respectively.

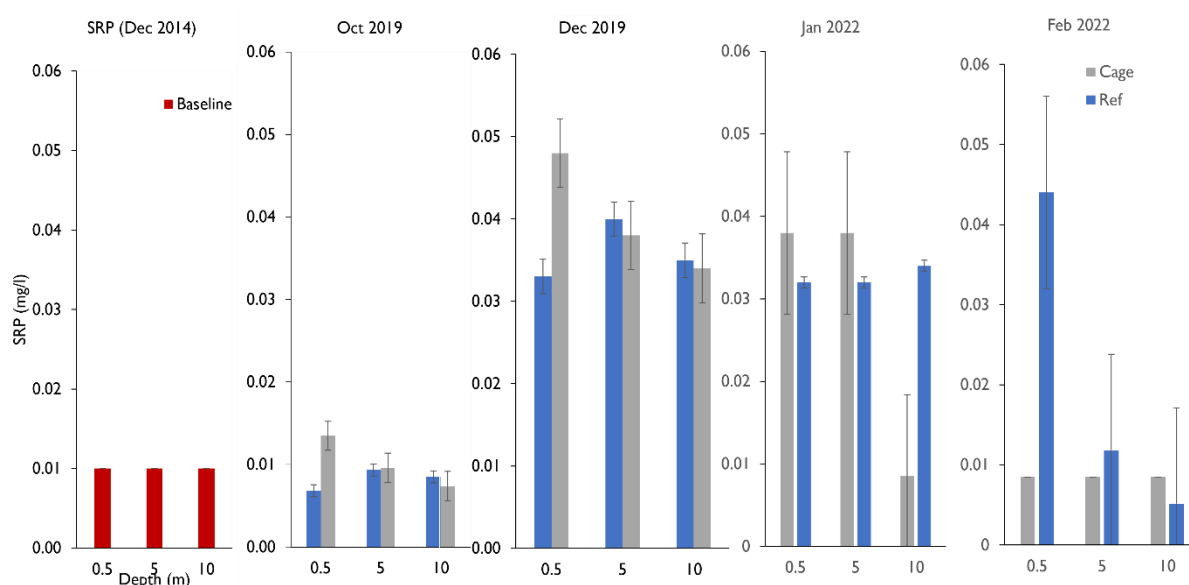


Figure 6.11: Average SRP concentration at the cage and control sites for the months of Oct and Dec. 2019 and Jan and Feb 2022 compared to the situation in Dec 2014 before production on the farm commenced.

Average TP (Figure 6:12) in water was higher at the cages was $0.14 \pm 0.042 \text{ mg/l}$ while the control had $0.102 \pm 0.051 \text{ mg/l}$. October levels at the control were $0.2 \pm 0.01 \text{ mg/l}$ while the cage site had $0.15 \pm 0.02 \text{ mg/l}$ and December was $0.07 \pm 0.03 \text{ mg/l}$ at the control and $0.06 \pm 0.01 \text{ mg/l}$ at the cages. However, this difference in TP was not statistically significant (Table 6.7) with median levels of (0.1480, and 0.1645 mg/l) for Dec 2019, and January 2022 respectively.

Average TN concentration (Figure 6:13) in water at the cage site was $0.53 \pm 0.36 \text{ mg/l}$ while that at the control site was $0.58 \pm 0.32 \text{ mg/l}$. This difference in TN between sites was not statistically significant (Table 6.7). Solid waste from the cages could undergo rapid diffusion, uptake, transformation, or removal (Azevedo *et al.*, 2011) staying within the epilimnion of the lake for only a brief time before it is passed on into the hypolimnion. This makes estimation of nutrient loading into the water column a complex process. Beveridge, (2004) recommends the assessment of water quality conditions and sedimenting particles or use of mass balance models to quantify waste from the cages.

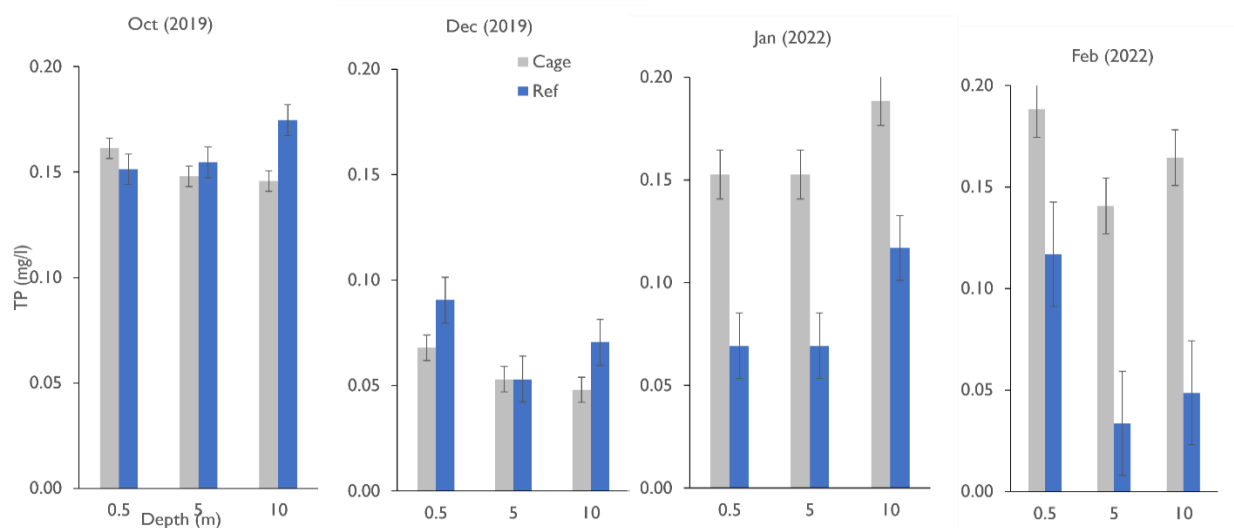


Figure 6.12: Total Phosphorus in water at the cages compared to the control sites. With no significant difference in conditions at both sites, the explanation for relatively high TP in both sites in October could be because it is a period soon after lake mixing while in December, the lake is starting a period of stratification.

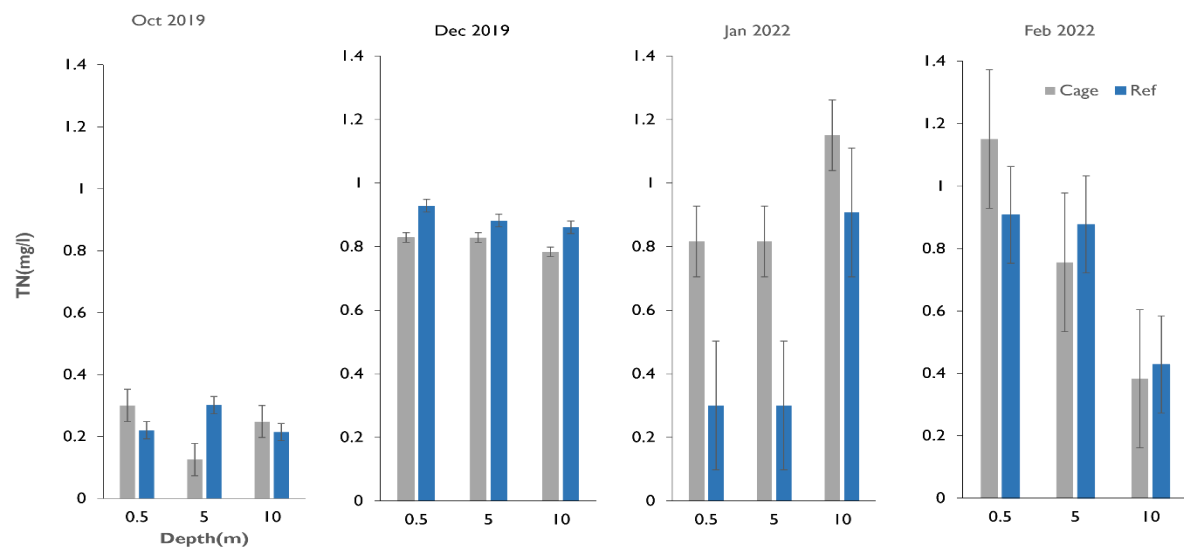


Figure 6.13 Total Nitrogen in water at the cages compared to the control sites, no significant difference in conditions at both sites.

6.3.3 Sediment Quality

Feed-use data from the farm was reported to increase by almost 6 tons between 2019 and 2021 (Table 6.8). In 2019 average %TP in the lake sediment was higher at the cage (1.12%) than the control site (0.19%) and again higher at the cage (2.39%) than control site (1.97%) in 2022 (Figure 6.14). %TC was higher at the control (3.94%) than the cage site (1.29%) in 2019 but lower at the control (17.6%) than the cage site (18.5%) in 2022.

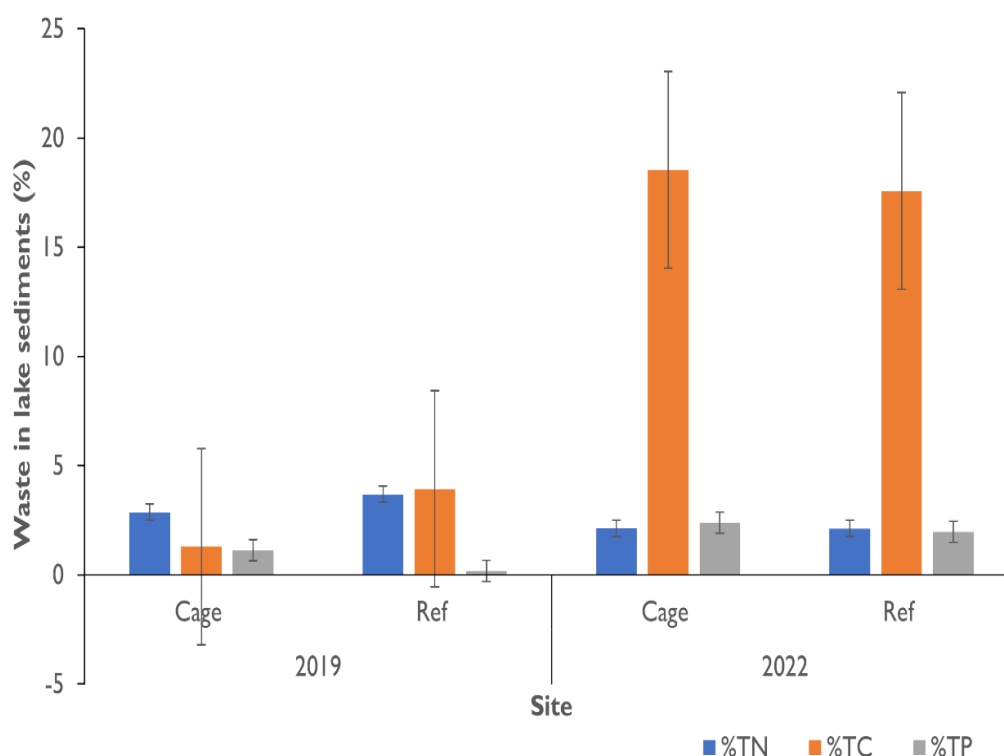


Figure 6.14: %TN, %TC and %TP in lake sediments at cage and control sites between 2019 and 2022

At the cages, average % TP in the lake sediment was 1.12% in 2019, and 2.39% in 2022. TC in the lake bottom sediments under the cages increased from 1.29% to 18.55% while TN decreased slightly from 2.87% to 2.13% (Table 6.8). However, using Mann-Whitney test at $p < 0.05$, the increase in TC was not statistically significant ($p = 0.98$) and neither was the increase in % TP or TN in the lake sediment at the cage site ($p = 0.67$)

Table 6.8: Average % TN, %TP, and %TC (by dry weight) in the lake sediment at the cages.

Year	Total Feed Used	%TP	%TN	%TC
2019/20	82,856	1.12	2.87	1.29
2021/22	88,830	2.39	2.13	18.54

Establishing the fate of phosphorus released from aquaculture farms in freshwater systems is made complex by differences in hydrodynamic conditions, the feed used and limnological

conditions at a site. Hence, site by site assessment would be required. Phosphorus rapidly changes form once released into the water column with the dissolved form available for plant uptake. However, under conditions of anoxia condition in the hypolimnion, even the P bound to sediments becomes bioavailable (Otu, *et al.*, 2017). Understand sediment diagenesis would improve knowledge on the fate of P lost from cages into the environment.

6.3.4 Benthic Macro-invertebrate diversity and abundance

The bottom substrate was mostly soft and flocculate with organic matter or feed remains. Samples were drawn at an average depth of 11m and 12.2m for the cage and control sites respectively. Highest average abundance at the cages was exhibited among sub-class *Oligochaeta* (616 individuals /m²) compared to 364 individuals /m² at the control (Table 6.9). Insect species *Amphipsyche sp.* (23 individuals/m²) of order Trichoptera and *Bulinus trigonus* (5 individuals/m²) of class Gastropoda were found only at the cages but not control sites. Similarly, Crustaceans of class *Ostracoda* were only present at the cages and not the control site. Species *Helobdella sp.* of class Hirudinea were abundant at the cages (89 individuals/m²) than the control site (14 individuals/m²).

Using Shannon-Weiner's diversity Index, H' (Table 6.10) the cage site had a higher number of individuals and a slightly higher species diversity compared to the control. However, the Heip's evenness index was higher at the control site than at the cages. A slightly higher Shannon-Weiner's diversity index at the cages ($H'=1.8$) compared to the control ($H'=1.7$), infers to only a slight difference in diversity between the two sites. A comparably high Shannon-Weiner's diversity index $H'=1.7$ at the control site indicates that nutrient loading may be due to other sources (point or non-point) (Scheren *et al.*, 2000) playing a major role in nutrient enrichment away from the cages. A higher number of individuals at the cage site among benthic macro-invertebrates at the cages (1008) compared to the control (651) could be due to the organic enrichment providing more food for organisms to thrive. However, the Heip's evenness index (HE) of 0.92 at the control site compared to 0.76 at the cages indicates that although the former has fewer species (richness of 6), they were more evenly distributed than the latter (richness of 8).

A high abundance of leeches (Class: Hirudinea) at the cages could be related to tolerance of some species to pollution (Mackie, 2001), the slowed currents caused by presence of the cage installations but also, the caging material that provide a substrate for attachment of these organisms.

Table 6.9: Average abundance of difference species at the cage and control sites (individuals /m²)

Species	Cage	Reference
Gastropoda		
<i>Bellamya Unicolor</i>	79	105
<i>Bulinus Trigonus</i>	5	0
<i>Melanoides tuberculata</i>	5	7
Diptera		
Chironominae	0	21
<i>Chaoborus sp.</i>	140	140
Trichoptera		
<i>Amphipsyche sp.</i>	23	0
Hirudinea		
<i>Helobdella sp.</i>	89	14
Oligochaetes	616	364
Ostracods	51	0

Table 6.10: Benthic macro-invertebrate richness, diversity, and equitability between sites.

Site	Total number of individuals	Richness	Diversity (H')	Heip's Eveness Index (HE)
Cage	1008	8	1.8	0.76
Control	651	6	1.7	0.92

Presence of Chironomidae larve, a macro-inverteberate with intermediate tolerance to pollution (Ravera, 2013) at the control but not cage site could be another indicator of the impact of organic loading. A high abundance of leeches (oligocheates) which are the most tolerate and last macroinvertebrates to disappear (Ravera, 2013) in polluted freshwater systems was found at the cages, indicating the sediment's ability to support life but also a progression in benthic enrichment compared to the control site. Similarly, a high abundance of Ostracods is a direct indicator of increased alkalinity at the cages (McCormack *et al.*, 2019) Larve of Trichoptera (caddisflies) which are classified as sensitive to pollution were found at the cages, implying the level of pollution generated is still within limits to support benthic life. However, as more nutrients are deposited at the cages, competition for space will lead to dominant species proliferation, lowering coexistence and decreasing diversity (Moss, 1998), although this could be driven by other factors as well (Aschehoug and Callaway, 2015). Importantly, biomonitoring is impacted by other factors like presence of predators, or changing invertebrate life stages (Ravera, 2013), and hence when assessing impacts of cage farming based on

evaluating changes in the benthic composition these factors should be taken into consideration.

The benthic conditions under the cages are a representation of biotic and abiotic conditions but also, a direct indicator of the impact of organic loading. Benthic macroinvertebrate assemblages are key bioindicators of anthropogenic disturbance (Fierro *et al.*, 2018) due to their high diversity, high sensitivity to biotic and abiotic factors, limited mobility, and their presence all year round (Mandaville, 2002). Several studies have employed the use of biomonitoring to establish changes in water quality associated to cage farming (Clerk, Selbie and Smol, 2004; Nabirye *et al.*, 2016; Hettige *et al.*, 2020; Varol and Balci, 2020). Mandaville, (2002) provides a practical compilation of metrics on macroinvertebrate tolerance levels based on various indices and could be a valuable tool for generating on-farm monitoring criteria for cage farming.

6.3.5 Aquaculture Index-based on Water Quality Assessment

To provide a simple and convenient summary of a complex dataset, a cage culture-status score was generated based on a procedure like that of the CCME index (CCME, 2001). Following the CCME categories, both the cage and control sites fell within the same score range yielding a value of 50.7 and 53 (Table 6:11) for the cage and reference sites respectively. These readings on the CCME score are both within the marginal category with a range of 45-64, where water quality conditions are frequently impaired and often depart from desirable levels.

Summarising the water quality data into a single score provided a convenient means of communicating the findings that could be easily communicated for decision support. The score demonstrated a better and easier way in which technical data could be reduced into a single score to ease integration of findings into decision making. Several examples exist on how water quality indexes are used to monitor aquaculture impacts (like Ferreira *et al.*, 2011; Ma *et al.*, 2020; Tallar & Suen, 2016).

Overall, the water quality at the cages and the control according to the index generated were not different, implying the presence of other sources of waste influencing conditions at the control site. Similarity in the cage culture score at the cage and control site (both marginal) agrees with the results from the traditional water quality monitoring that found no difference in conditions between sites. This gives more evidence of how complex it might be to base on

water quality assessment in identifying ecological difference in sites receiving cage organic loading from non-cage sites.

Table 6.11: Cage culture score based on the CCME scoring methodology. No difference was seen in conditions at the cage and control site.

Site	Cage culture score	CCME meaning	Implication
Cage	50.7	Marginal-Water quality is frequently threatened or impaired	Conditions usually depart from natural or desirable levels
Control	53	Marginal-Water quality is frequently threatened or impaired	Conditions often depart from natural or desirable levels

6.3.6 Mass Balance Estimate of the Nutrient Flux

The feed used increased between 2019/20 and 2021/22 from and so did the modelled outputs of carbon, phosphorus, and nitrogen for the same period (Table 6.12). The FCR was the same at 1.5 Model predications indicated an increase in C, TP and TN as more feed used between the years. Of the TP waste generated in 2021/22, 70% was solid while 30% was in dissolved form while, 65.4% of TN was in dissolved form and 34.6% was predicted as the solid form. It is the dissolved form of TP that is available for plant uptake. Taking the FCR of 1.5 for every ton of fish produced in between 2019/20 and 2021/22 (wet weight), an average of 86.97kg of TN, 14.11kg of TP and 274.12 kg of C per ton of fish was added into the environment at this farm site.

Compared to levels estimated for Tilapia culture by Beveridge, (1987) i.e., 23-29 kg TP /ton of fish, modelled estimates of TP at the study farm were low but higher than ranges found by Gondwe, Guildford and Hecky, (2011a) ; 10.75kg TP, 26.5kg TN, and 25kg Carbon/ton of Nile Tilapia fish, Montanhini Neto and Ostrensky, (2015) found 14.26 kg TP, 44.95kg TN and 1040.63kg of Carbon/ton of fish produced at tropical cage farms.

Table 6.12: Estimated of nutrients generated from nutrient mass balance model. Between 2019/20 and early 2021/22

Year	Waste generated (kg/ton of fish)		
	TP	TN	TC
2019/20	13.62	83.95	264.60
2021/22	14.60	90.00	283.66

These TP levels were close to findings in this study (14.11kg/ton of fish) but higher for TN (86.97kg/ton). ASC, (2019) recommends a standard of TP ≤ 27 kg/ton of Tilapia fish produced, which based on this standard would qualify the farm as confirming. Unfortunately, the feed type and specifics like the proximate content of each ingredient, digestibility coefficients, percentage feed lost to the environment, makes it hard to use a standard developed under different conditions to the conditions of the farm under evaluation. In other words, whilst Aquaculture Stewardship Council sets a standard of TP as ≤ 27 kg/ton of fish (ASC, 2019), when farm management practices and feeds used are highly variable, using a uniform metric is limiting. Hence, values for waste generated on farm will have to be calculated depending on farm practices although this still provides no base for setting production limits.

6.3.6.1 Validation of Mass balance Models

Using the observations from sediment traps set under the cages, results indicate that on average, the nutrient mass balance model overestimates %TP, %TN and %Carbon (Figure 6:15) by approximately 18.1, 29.6 and 38.6 folds respectively. This overestimation could be due to uncertainties in the observation data created during sampling. The traps were not set centrally under the cages due to lack of proper diving equipment, hence, some of the waste was not trapped. The diameter of the pipe (6cm) could have been too narrow to let in all the waste. There studies have used pipes 8cm in diameter. On another occasion, a bottle of the samples fell off while retrieving the sediment traps. Both challenges affected the weight of the final sediment sample. Hence model performance was validated by comparing its performance to findings in other studies done on Tilapia cages in tropical lakes rather than results from sediment traps set in this study.

Table 6.13: Comparing modelled to observed nutrient loading generated from mass balance model and those observed from the sediment traps.

	2019/20			2021/22		
	%TP	%TN	%C	%TP	%TN	%C
Observed	1.12	3.69	29.8	3.22	2.8	17.87
Modelled	29.59	90.52	825.52	31.72	97.05	885.04

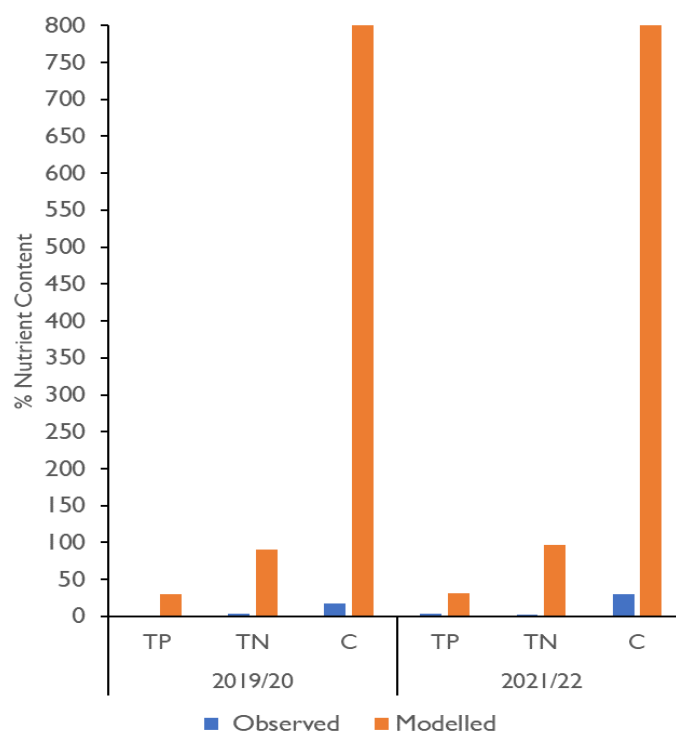


Figure 6.15: A comparison of the modelled and observed waste generated at a cage farm. Uncertainty in the sampling process could explain the difference in performance especially given modelled values of TN, TP and carbon are comparably close to those from other studies.

6.4 Conclusion

In this study, an assessment of water and sediment quality together with nutrient mass models was done to establish the impacts of cage culture on the environment. This study aims to contribute to the development of better on-farm monitoring protocols for tilapia cage culture production on Lake Victoria. As a result of assessing water quality parameters alone, there was not enough evidence to suggest that cage production had negatively affected sediment or water quality at the site. By analysing the benthic species assemblage, a better understanding of the differences in conditions between sites caused by cage farms was established. One indicator was the presence of pollution-tolerant species in the cage sites that were not present at the control. To relate farm activity to ecological changes when setting production limits, it is ideal to combine the nutrient balance models with the benthic monitoring method.

Results from water quality assessments (temperature, redox potential, DO, TN-water, conductivity, pH, ammonium, SRP, TP-water and TSS), indicated conditions at the cages were not different from the control sites. These results resonated with what Kashindye *et al.*, (2015)

found on the Tanzanian side and what Okechi *et al.*, (2022) found on the Kenyan side, both comparing conditions at cages and control sites to assess the impact of cage farming in Lake Victoria. This study found that on site carbon deposition at the lake bottom (0m from cages) increased from 1.29% to 18.54% between 2019/20 and 2021/2022 as production on the farm increased. Mwebaza-Ndawula *et al.*, (2013) reported an increase in the number of cages from 6 to 147 at a commercial site in the northern waters of Lake Victoria, however, with no significant change in water quality upstream and downstream the cages within the same period. Furthermore, Musa *et al.*, (2022) reported an increase of 386.5% Total Organic Carbon between the start and end of the production period at cage site in Kenyan waters but no significant difference in N and P levels in water when compared to other sites, even as close as 50m from the cages. Studying the impact of cage farming on another tropical Lake Malawi, Gondwe, Guildford and Hecky, (2011) reported no significant difference in TSS, NH_4^+ , NO_3^- , SRP, particulate phosphorus, particulate carbon, and particulate nitrogen at cage and control sites, despite reporting nutrient loading in particulate and dissolved form between periods of the study. Organic loading was attributed to the use of feed between the periods with faeces and waste feed lost directly to the lake bottom.

As the control site had similar conditions to the cage sites that were receiving organic loading from fish production, this indicated that the site was receiving nutrients from sources other than those associated with the cages. It would thus be impossible to determine the difference in conditions or the relationship between organic enrichment in the water column and the presence of cages based solely on assessing the quality of the water or nutrient levels in the sediment.

Traditional water quality assessments involved extensive fieldwork which was costly and constrained by fewer technical staff and laboratory services in the region. The variability in lake conditions, such as hydrodynamics, benthic characteristics, and allochthonous sources, affected the effectiveness of water and sediment quality assessment as a robust tool for monitoring cage production. When applied, the volume of technical information generated in water quality assessments needs to be reduced into a score that is adaptable. Using the CCME methodology (CCME, 2001), the water quality data at the cage and reference site were reduced to an aquaculture score. However, no difference was seen between the sites. This led to the conclusion that similarly high nutrient loading was happening in the control site, which was 300m from the cages. Musa *et al.* (2022) found no significant difference in nutrients, water chemistry and sediment quality even as near as 50m from the cages.

Since Lake Victoria and similar eutrophic lakes in the Africa Great Lakes Region have complex biotic and abiotic dynamics, traditional water quality assessments alone will result in erroneous

production limits or undermined farm impacts. When licensing is the goal, more than a physico-chemical evaluation or sediment nutrient assessment must be conducted. Water quality monitoring will still be required for establishing changes in conditions since these have a direct implication on farm health and welfare. With a large-enough database, a multivariant analysis to correlate water quality conditions and macroinvertebrate assemblages will be required. Among data to be collected will be currents and wind speed and direction to establish particle movement and impact of diurnal wind movements on waste dispersal. A tiered on-farm monitoring system that includes assessment of organic loading, based on nutrient balance modelling results, visa vie changes in the benthic conditions could be adapted.

CHAPTER 7

REGRESSION MODELLING TO INFORM WATER QUALITY MONITORING AT CAGE FARMS

7.1 Introduction

Environmental externalities of cage farming, mostly organic loading, are an issue of concern in aquaculture (Quiñones *et al.*, 2019; Kee *et al.*, 2022). This is particularly pertinent in shared lake systems reported to have major hydrological and water quality monitoring challenges (Awange, 2020). With slow water currents and long residence time in lakes (Vinçon-Leite and Casenave, 2019), organic enrichment from cage fish farms has a gradual cumulative impact on the ecosystem. In the AGL region, cage culture can be an invaluable source of revenue and fish for food and nutrition security if well planned and monitored (Cowx and Ogutu-Owhayo, 2019). There is a gradual increase in the number of cage farms on Lake Victoria in recent years. To ensure that other uses of the lake are not further compromised, sustainable cage farming will require implementation of Best Aquaculture Practices, stringent monitoring, and regulation (Njiru *et al.*, 2018; Musinguzi *et al.*, 2019).

The current practice for monitoring at Tilapia cage farms in Lake Victoria involves sampling for water physico-chemical conditions, zooplankton and phytoplankton abundance and sediment analysis for benthic macroinvertebrates population as presented by Sekiranda *et al.* (2004); Mwebaza-Ndawula *et al.*, (2013); Kashindye *et al.*, (2015); Musa *et al.*, (2022) and Okechi *et al.*, (2022) . From such monitoring assessments, the outcome is values given as ranges for variables and their possible impact on aquatic life. This monitoring provides the empirical evidence upon which models could be developed and validated to further aid management decision making (Zessner, 2021). As with many African Great Lakes, monitoring in Lake Victoria is prohibitively expensive, constrained by a lack of testing and calibration laboratories and a shortage of trained personnel, which reduces the frequency of monitoring (Plisnier *et al.*, 2018).

Phosphorus is most decisive limnological metric responsible for in-lake primary production in freshwater systems. Similarly, it is the nutrient of interest as far as organic enrichment of cage farming is concerned. The amount of Total Phosphorus (TP) released from cage farms in form of waste feed and faeces depends on nutrition (type and quality of feed given, its digestibility, P-content) and husbandry techniques. In its solid form, some of this waste P is consumed by

wild fish and other invertebrates while the rest is deposited to the sediment particularly under the cages (Beveridge, 2008) and only returning to the water column following perturbations in the sediment, anoxia, or biotic activity (Otu, et al 2017). The dissolved form is what is available for plant uptake in the euphotic zone of the epilimnion.

Regression statistical modelling involves establishment of interrelationships among variables (Schuenemeyer and Drew, 2010). Multilinear regression considers how multiple independent variables account for variance in a dependent variable by means of the regression coefficients attached to them. As an output from MLR statistical modelling, a relationship is built between predictor (independent) variables $x_1, x_2 \dots x_n$ and a response (dependent) variable y . This relationship is represented by a simple mathematical equation (Equation 7.1), where β_0 is the intercept and, β_1 is the slope parameter for x_1 and estimates x_1 , and so forth, while ε is the error (or residue) calculated as the difference between the predicted and actual value of the dependent variable. Rather than generating new sets of raw data, the model equation can be used to predict the dependent variable when data on independent variables are used (Giancristofaro and Salmaso, 2003). The MLR provides information about correlation among variables but not causation (Tranmer *et al.*, 2020). Essentially, the model shows how variance in the dependent variable could be explained by variance in the independent variable but does not mean the latter causes the variations in the former.

$$y = \beta_0 + \beta_1 x_1 + \beta_2 x_2 + \beta_3 x_3 + \dots \beta_n x_n + \varepsilon \quad \text{Equation 7.1}$$

To contribute to economically feasible and technically acceptable tools for monitoring cage impacts, statistical modelling is used to explore options for sustainable on-farm monitoring efforts in Lake Victoria. Total phosphorus is regressed against other physico-chemical parameters like DO, NH_4^+ , SRP, temperature, redox, conductivity, TN, pH.

The aim is to identify which of the variables monitored best explains changes in TP in the water column. This does not mean that other parameters are not important as they may have different environmental effects and implications on the health/management of the fish. Focus is to reduce to a few significant parameters, with an aim of reducing monitoring costs and devising tools to encourage on-farm rapid assessment. Secondly, Lake Victoria is a highly heterogenous system, hence, the resultant statistical models generated are tested for performance to assess usability in different parts of the lake. While patterns in the dataset were explored, a prediction goal was added when the models were tested with out-of-sample observations to check for generalisability.

7.2 Methodology

7.2.1 Study area

Regression models to assess variability in TP levels in the water column due to cage farming were developed for two sites with different hydrodynamic conditions i.e., one farm in a semi-enclosed bay (Bay Farm) and the other at the river mouth (River farm). Bay Farm is 0.05457° N 32.54258° East on the northern part of the lake (Figure 7.1). The farm has been in production since 2015, with 10 technical staff, producing on average 50 tons/ annum. It is a relatively small farm with 15 rectangular 6mx6mx6m dimension cages. The farm uses pelleted feed to produce Nile Tilapia (*Oreochromis niloticus*) up to an average of 8 months per production cycle.

The river mouth farm located in the Napoleon gulf is at the mouth of River Nile (Figure 7.1 B), the only outlet from the lake. This is a larger farm that has been in production since 2006, culturing Tilapia in both square and circular cages. In 2019 the farm had 29 square cages made of metallic material and 94 other cages made of Polyvinyl Chloride (PVC). With an average stocking density of 78-400 fish/m², the farm produces an average of 1200-1500 tons per annum.



Figure 7.1: Locations of the farms where samples were taken (A) Bay farm and (B). River farm. In proximity are major towns and cities of (Entebbe for (A) and Jinja for (B) and other human developments like fish landing sites and urban developments

7.2.2 Data

Sampling for physico-chemical analysis at Bay farm was done in October, December 2019 and January, February 2022. Variables reported were DO, TP, SRP, Ammonium, conductivity, redox potential, TSS, Temperature, TN, Secchi depth and pH. Details of sample handling and analysis are presented in Chapter 6 of this thesis. For River farm, secondary data was generated from monitoring reports of cage activity obtained from the National Fisheries Resources Research Institute. Data was obtained for March, June, September, and December for 2018. Parameters included $\text{NO}_3\text{-N}$, DO, TP, SRP, Ammonium, conductivity, TSS, Temperature, TN, Secchi depth and pH. In both cases, TP was regressed against other parameters since it is a key parameter for monitoring organic enrichment.

7.2.3 Model setup

Two models were developed representing conditions of slow and fast water flow rates in a bay and at the river mouth. Multiple Linear Regression (MLR) was done in SPSS Version 28 (IBM Corp, 2021). The data was first tested for assumptions of MLR including, normality among observations, homoscedasticity (equal variance), linearity, lack of multicollinearity, lack of outliers or influential points (Laerd Statistics, 2015) following steps (Figure 7.2) and detailed in the subsequent sections.

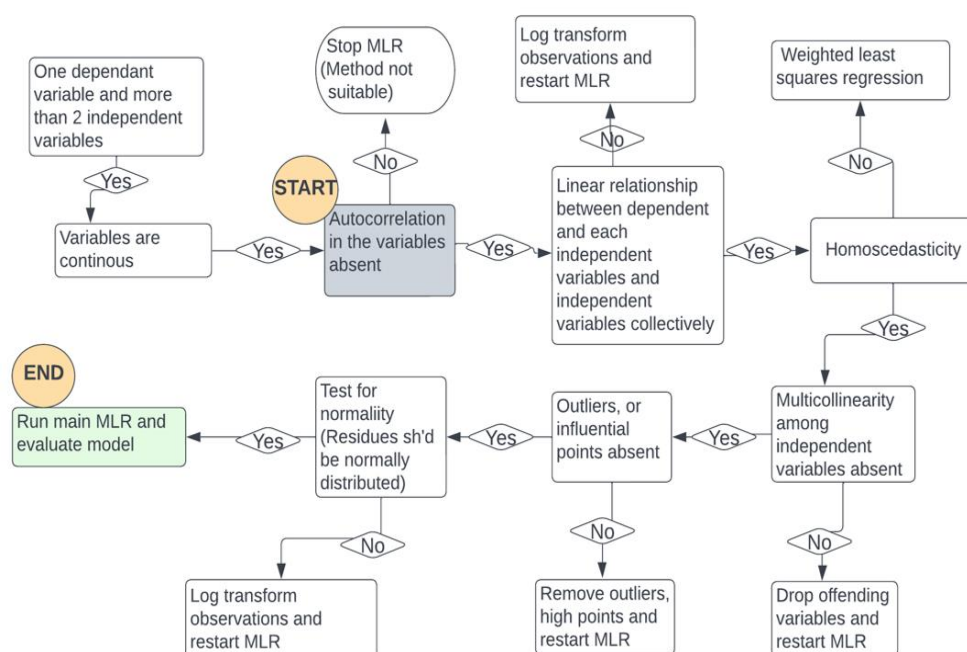


Figure 7.2: Flowchart for developing a MLR model. shows the assumptions that must be met and the actions to take in case of non-conformity. For every restart, the process begins from the START through to the END step.

7.2.4 Statistical tests

The process begun with generating a MLR model having TP as the response variable (dependent) and DO, SRP, NH_4 , conductivity, redox potential, TSS, Temperature, TN, Secchi depth and pH as the independent variables. It is using the values generated in the first MLR model i.e., regression standardized residuals, standardized predicted value, Cook's distance, Centred leverage points and the weights (when weighted regression had to be done). Correlation matrices were generated for the variables to check for multicollinearity among the independent variables and between TP and each of the independent variables.

Normality of the standardized residuals was tested for using the Shapiro-Wilk test at $p < 0.05$. To establish that there was no autocorrelation in the residuals, a Durbin-Watson statistic was run. This was followed by a check on if there was a linear relationship between TP and each of the independent variables and TP and the independent variables all together. Equal error variances were checked for by plotting studentized residuals against the unstandardised predicted values of TP generated by the model. Multicollinearity, which occurs when two of the independent variables are highly correlated with each other was checked for by inspecting the correlation coefficients with statistical significance at $p < 0.05$ and the Tolerance values. Outliers and influential points were catered for by excluding all values with a generated leverage > 0.2 (Laerd Statistics, 2015) and a Cook's distance > 1 (Schuenemeyer and Drew, 2010).

Visual inspection of histogram superimposed over a normal curve was used to check for normality of residuals, additionally the mean and standard deviation of the standardised residuals generated in the model was assessed for a value close to 0 and 1 respectively. In the dataset where heteroscedasticity was detected by visually inspecting plots of the residuals against the predicted values, the Breusch-Pagan test was done for confirmation. The test is used to confirm that the variance in the errors is not dependent on values of any of independent variables. With the Breusch-Pagan test, if $p > 0.05$, the null hypothesis stating that the variance of the residuals is equally distributed, meaning homoscedasticity is present (Hayes and Cai, 2007). When violated, log transformation was done, and the model set up and rerun on transformed observations.

7.2.5 Model validation and evaluation

A random sample of each of the datasets from the Bay and River dataset was selected using SPSS. 78% of the dataset was used for model setup and the remaining 22% used for validation. To check for performance of the model under different water flow conditions, the data from either site was tested in models developed for opposite conditions i.e., out-sample validation. With the regression equations from the modelling, predicted TP values were calculated, and compared to the observed. The Standard deviation, RMSE and coefficients of efficiency and variation were evaluated. Goodness of fit of the models was evaluated using four criteria; visual inspection of plots of observed against modelled TP values and how they fit along a 1:1 line, coefficient of determination (R^2) of the model, Root mean square error (RMSE)-a predicted value error indicator Equation 7.2 and a dimensionless indicator Nash and Sutcliffe (NSE) coefficient of efficiency (Nash and Sutcliffe, 1970).

$$\text{Root Mean Square Error (RMSE)} = \sqrt{\frac{\sum_{i=1}^N (O_i - P_i)^2}{N}} \quad \text{Equation 7.2}$$

$$SD = \sqrt{\frac{\sum_{i=1}^N (O_i - \bar{O})^2}{n-1}} \quad \text{Equation 7.3}$$

$$n_t = \frac{SD}{RMSE} - 1 \quad \text{Equation 7.4}$$

$$NSE = 1 - \left(\frac{1}{n_t + 1} \right)^2 \quad \text{Equation 7.5}$$

Where N is sample size, O_i is Observed, P_i is predicted and $\bar{O}$ is mean of the observed values, SD; Equation 7.3 is standard deviation of the observed values and n_t Equation 7.4 is number of times SD (variability among observations) is greater than the RMSE (presented as mean model error) and the NSE denoted by Equation 7.5.

The final performance assessment was done in the FITEVAL tool (Ritter and Muñoz-Carpena, 2013), which utilises bootstrapping in the observed values. This involved sampling, and resampling within the observed data to create several ‘new’ datasets O_i for which the predicted value P_i and the corresponding RMSE, NSE, SD and n_t statistics were calculated. In the FITEVAL tool, the probability distribution of the adapted NSE performance intervals (Table 7.1) were generated. The NSE intervals were adapted to less stringent values due to uncertainties in the regression models as guided by Harmel *et al.*, (2014).

Table 7.1: NSE criteria for the goodness-of-fit evaluation adapted from Ritter and Muñoz-Carpena, (2013)

Performance rating	n_t	NSE ranges (Ritter and Muñoz-Carpena, 2013)	NSE ranges in present study
Very good	≥ 2.2	≥ 0.90	0.65-1.00
Good	1.2-2.2	0.8-0.9	0.5-0.65
Acceptable	0.7-1.2	0.65-0.8	0.3-0.50
Unsatisfactory	< 0.7	< 0.65	< 0.3

To cater for the uncertainty inherent in measuring water quality data (Harmel *et al.*, 2006), a coefficient of variation (CV) for the observed and predicted values was included. This was calculated as in Equation 7.6.

$$CV(\%) = \frac{\text{Standard Deviation of values} \times 100}{\text{Mean of values}} \quad \text{Equation 7.6}$$

A bias of 3% was included in the performance evaluation to cater for model uncertainties.

7.3 Results

7.3.1 Model calibration for Bay Farm

7.3.1.1 Assumption testing

Multicollinearity among independent variables; and TP and within the independent variables was checked for by evaluating the correlation matrices. Normality tests on the errors showed normal distribution by inspection of the histogram superimposed over the normal curves (Figure 7.3 A) and P-P Plot (Figure 7.3 B.). Similarly, a mean and standard deviation close to 0 and 1 respectively were indicators of normality. The stepwise option was selected on setting the model up, to reduce variables to only those that had statistically significant correlation with the changes in Total Phosphorus and avoid forcing those whose contribution was not significant.

There was independence of residuals in the observation, as assessed by a Durbin-Watson statistic 1.463. Using the Durbin-Watson statistics significance tables (Savin and White, 1977), the value of 1.463 was higher than the upper limit with two regressors at $p < 0.05$. Inspection of the correlation matrices among independent variables and a Tolerance value of not less than 0.1 showed no multicollinearity among the variables. There was homoscedasticity, as assessed by visual inspection of a plot of studentized residuals versus unstandardized predicted values and the partial plots of TP against the significant independent variables (SRP and DO; Figure 7.4 A and B). Homoscedasticity was confirmed using the Breusch-Pagan test where the squares of the residuals regressed against all independent variables yielded a $p > 0.05$; i.e., $p = 0.076$. Cook's distance values were all less than 1. Two cases that had violated the Leverage requirement (were > 0.2), and these were excluded, and the model was rerun.

Table 7.2: Analysis of Variance (ANOVA) in Bay Farm model

	Model	Sum of Squares	df	Mean Square	F	Sig.
1	Regression	0.105	2	0.052	33.817	<.001^c
	Residual	0.051	33	0.002		
	Total	0.156	35			
Dependent variable: TP						
Independent variable: (Constant), SRP, DO						

The final model statistically predicted changes in TP at Bay farm, $F(2,33) = 33.82$, $p < 0.001$ (Table 7.2). From the model, SRP and DO can significantly explain changes in TP

levels at low water flow rates. With the R^2 value of 0.672 (Table 7.3) it means SRP and DO explain 67.2% of variability in TP at Bay farm

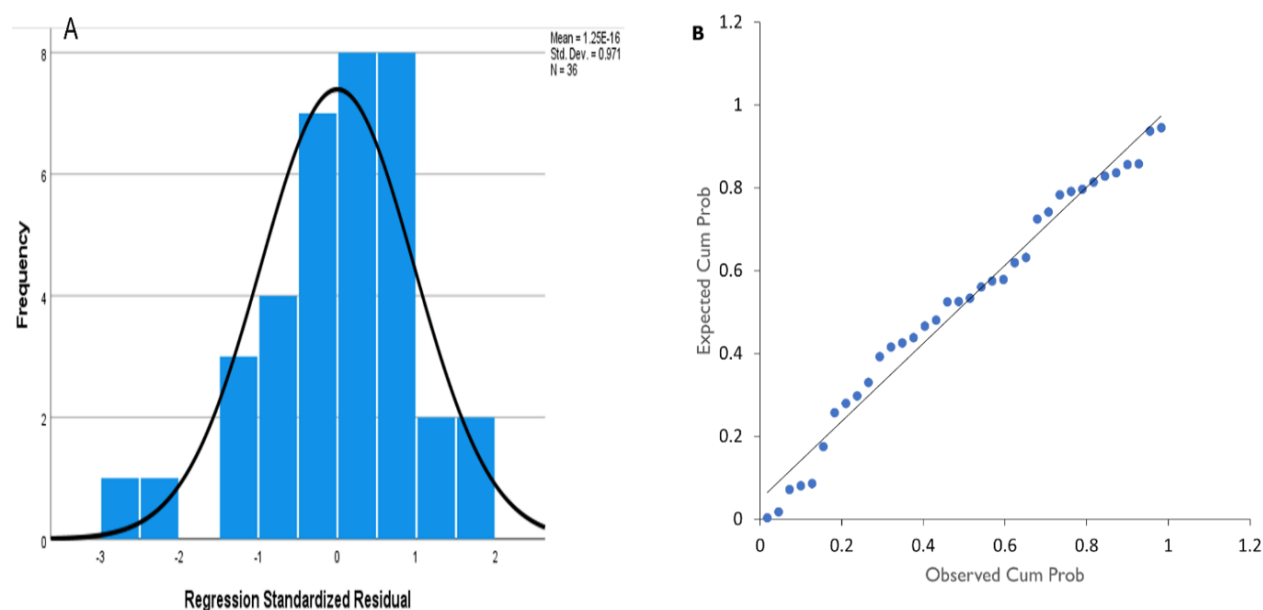


Figure 7.3A: Histogram superimposed with a normal distribution curve, $N=36$, SD of 0.971 and mean of 1.25×10^{-16} . All residuals were within $\pm 3SD$. **(B).** Normal P-P Plot showing how regression standardised errors lie along a fitted 1:1 line for the Bay Farm model.

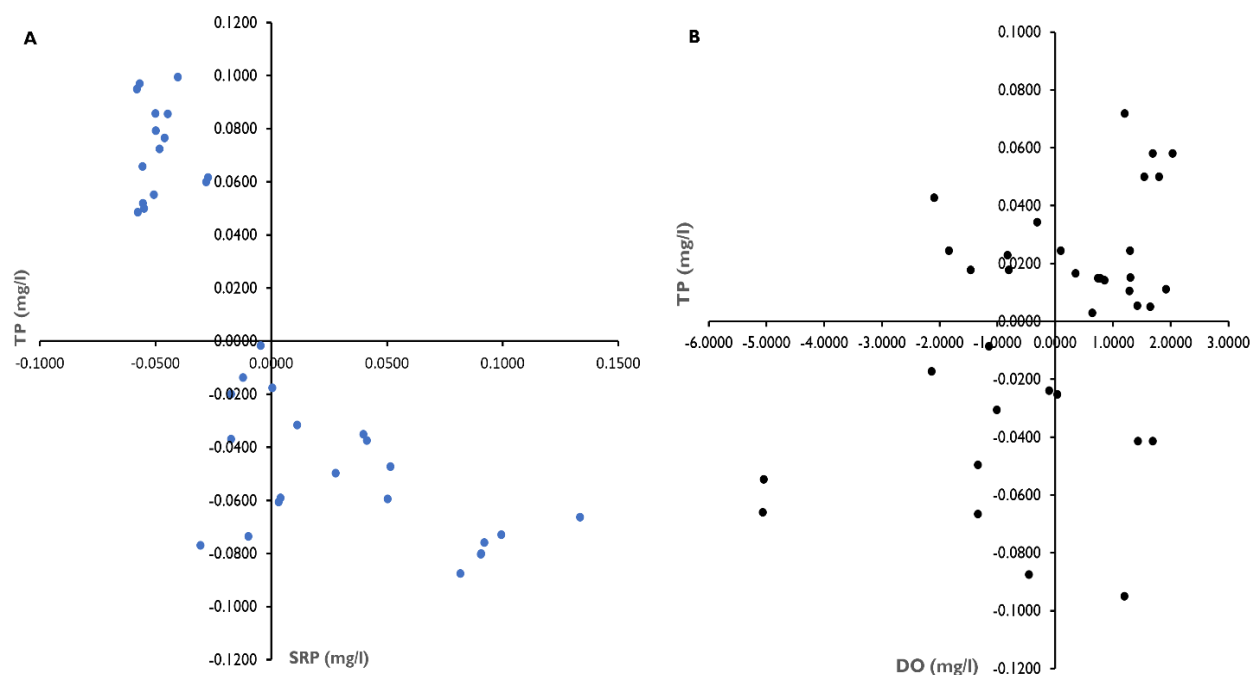


Figure 7.4: Partial plots of TP against SRP (A) and DO (B) for Bay model. Both plots show linearity and homoscedasticity in the model with SRP having a negative correlation and DO a positive one

To maintain the units of measurements for each of the variables, the unstandardized coefficients in Table 7.4 were used and the resultant model Equation 7.7 was denoted as:

$$Y_1 = 0.071 - 0.969x_1 + 0.01x_2 \quad \text{Equation 7.7}$$

Where: Y_1 is the concentration of Total Phosphorus in water (TP) (mg/l), x_1 is the concentration of soluble reactive phosphorus (SRP) (mg/l) and x_2 is the concentration of dissolved oxygen (DO)(mg/l).

Table 7.3: Bay model performance and the retained significant variables.

	R	R ²	Adjusted R ²	Std. Error of the Estimate	Durbin-Watson
Final Model	0.820 ^b	0.672	0.652	0.0393314	1.463
Independent variable (Constant), SRP, DO					
Dependent variable: TP					

The model as presented as Equation 7.8 and it shows that there is a negative relationship between predicted TP and SRP i.e., for every unit increase in SRP, the TP declines by 0.969mg/l when DO is held constant. With DO, the relationship is positive, and every unit increase in DO, the predicted TP increases by 0.01mg/l when SRP is held constant.

$$\widehat{TP} = 0.071 - 0.969SRP + 0.01DO \quad \text{Equation 7.8}$$

This doesn't mean DO or SRP cause the changes in TP but presents the relationship between the variables. In presence of data on DO and SRP, using Equation 7.8 the changes in TP could be predicted.

Table 7.4: Coefficients of the Bay Model indicating levels of significance and coefficient limits.

Model	Unstandardized Coefficients		Sig.	95.0% Confidence Interval for B	
	B	Std. Error		Lower Bound	Upper Bound
1 (Constant)	0.071	0.028	0.016	0.014	0.128
SRP	-0.969	0.120	0.000	-1.213	-0.724
DO	0.010	0.004	0.015	0.002	0.017

7.3.2 Model calibration for River farm

7.3.2.1 Assumption testing

The variables included in the MRL model were TP, TN, TSS, SRP, NO₃-N, NH₄-N, conductivity, pH, Temperature, Secchi depth and DO. The data was taken through similar assumption testing as was the Bay model. With N=32, normality, independence of observations, homoscedasticity, linearity, presence of outliers or influential cases and multicollinearity among the independent variables was tested. A regression model of TP against all other independent variables was run to generate values used in assumption testing.

To check for homoscedasticity using the Breusch and Pagan test, squared values of the residuals were regressed against the independent variables in the model. Results showed $p < 0.05$ ($p = 0.009$) (Table 7.5). This confirmed heteroscedasticity in the model.

Table 7.5: Heteroscedasticity detected in regression squared residuals against all independent variables.

Analysis of Variance						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	26.347	2	13.173	5.631	0.009 ^c
	Residual	65.506	28	2.340		
	Total	91.853	30			
Dependent variable: Residuals squared						
Independent variable: (Constant), Temp, NH ₄ -N						

Failing the assumption of homoscedasticity, natural log transformations observations were made, and the model run again. Regression was first between natural log TP against natural logs of all the independent variables. Then, the squared residue values were regressed

against the independent variables. Similarly, heteroscedasticity was confirmed with a failed null hypothesis; and in this case, $p=0.031$ (Table 7.6) (for homoscedasticity, p -value must be >0.05). The weighted least square option was subsequently used.

Table 7.6: Heteroscedasticity detected in transformed observations in the Farm model.

Analysis of Variance						
Model		Sum of Squares	df	Mean Square	F	Sig.
1	Regression	0.040	1	0.040	5.105	0.031
	Residual	0.235	30	0.008		
	Total	0.275	31			
Dependent variable: residuals squared						
Independent variable: (Constant), $\ln\text{NH}_4$						

7.3.2.2 Weighted least square model

Weighted least square regression was used after most assumption tests for MLR were violated. Transformed values were maintained. Weights for the observations based on outputs of the predicted dependent variable, $\ln\text{TP}$ in the first model run were created. The dependent variable ($\ln\text{TP}$) was regressed against the independent variables to generate unstandardised residuals and corresponding predicted TP values. Then, absolute values of the residuals were computed to serve as estimators of standard deviation. These absolute values of the residuals were regressed against the predicted TP values from the first model, and the output saved as predicted standard deviations of the original TP predictions. The weight was the reciprocal of the squared value of the standard deviation of the predicted TP values. Using these weights, the MLR model was setup and run again with the transformed values of both dependent ($\ln\text{TP}$) and log transformed independent variables and their weights included. Again, the stepwise method was employed to retain only statistically significant variables. Model performance (Table 7.7) retained $\ln\text{Temp}$ and $\ln\text{TSS}$ as predictor variables.

In the final model, two variables were retained: temperature and total suspended solids. This meant that temperature and total suspended solids can significantly predict TP at the River farm, $F(2,19) = 6.64$; $p = 0.007$.

This shows that the model fits the data well when temperature and TSS are included than what the intercept alone would predict. Model performance gave an R^2 of 0.411 (Table 7.8)

inferring that temperature and total suspended solids accounted for 41.1% of changes in TP at the site with higher water flow rates.

Table 7.7: Performance of River model using weighted observations.

Analysis of Variance						
	Model	Sum of Squares	df	Mean Square	F	Sig.
1	Regression	23.071	2	11.536	6.639	0.007^d
	Residual	33.015	19	1.738		
	Total	56.086	21			
Dependent Variable: lnTP						
Independent variable: (Constant), lnTemp, lnTSS						
Weighted least squares regression - weighted by weight						

Table 7.8: River model performance and retained variables of Temp and TSS

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	0.641 ^b	0.411	0.349	1.318	0.85
Independent variable: (Constant), lnTemp, lnTSS					
Dependent variable: lnTP					
Weighted least squares regression - weighted by weight					

Model coefficients showed a large negative intercept of the model equation, with a coefficient (magnitude) value of 1.694 and (–)0.063 for temperature and TSS (Table 7.9) showing a negative relationship with TSS.

Table 7.9: Coefficients for the River Model and the corresponding significance scores for the retained variables

		Unstandardized Coefficients		Sig.	95.0% Confidence Interval for B	
Model		B	Std. Error		Lower Bound	Upper Bound
1	(Constant)	-7.775	2.279	0.003	-12.55	-3.005
	lnTemp	1.694	0.700	0.026	0.229	3.159
	lnTSS	-0.063	0.028	0.035	-0.121	-0.005

The final model equation using the unstandardized coefficients was denoted by Equation 7.9

$$y_2 = -7.775 + 1.694 \ln x_3 - 0.063 \ln x_4 \quad \text{Equation 7.9}$$

Where y_2 is the concentration of Total Phosphorus (TP) (mg/l), x_3 is the temperature (°C) and x_4 is Total Suspended solids (mg/l), all values in natural log. This translates into *Equation 7.8*

$$\widetilde{TP} = -7.775 + 1.694\ln Temp\ (^{\circ}C) - 0.063\ln TSS \quad \text{Equation 7.8}$$

From the model, there is a positive relationship between TP and temperature and a negative relation between TP and TSS. For every unit increase in temperature, the concentration in TP increases by 1.694mg/l when other factors are kept constant while TP declines by 0.063mg/l for every unit increase TSS (mg/l) when other factors are kept constant at the river mouth.

7.3.2.3 Model evaluation and Interpretation

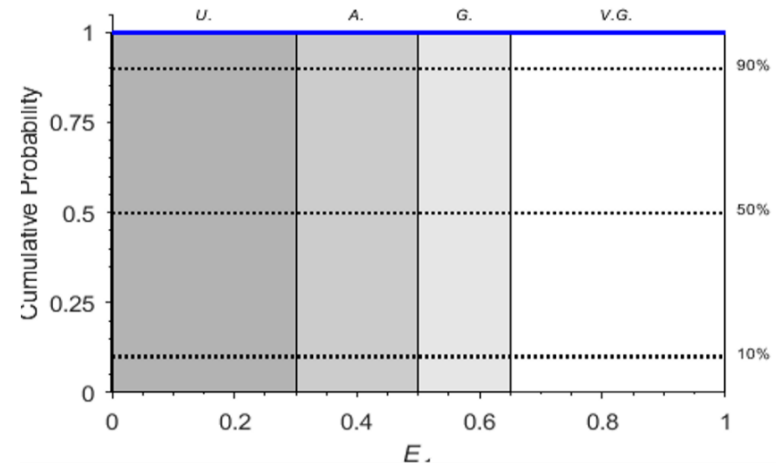
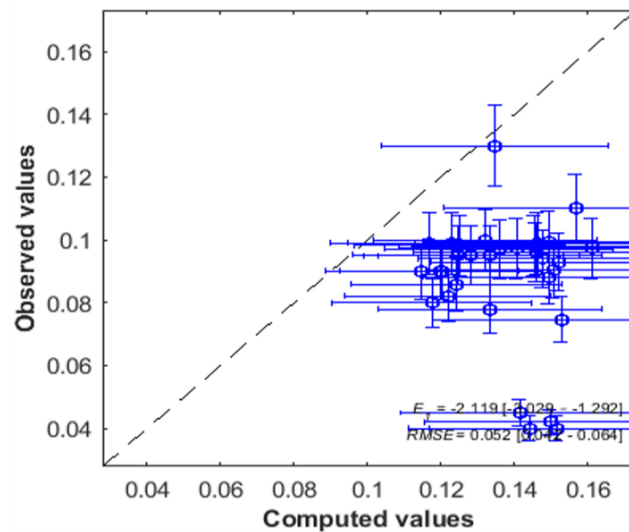
Using the FITEVAL tool (Ritter and Muñoz-Carpena, 2013), the bay model was first evaluated using data collected from within the bay (22% of the dataset randomly sampled from within and saved for model validation). Then the same model was tested using data collected at river farm to assess its performance at different hydrodynamic conditions.

For the Bay model tested with data river data, a poor model performance (Table 7.10), of an R^2 of 3.9% and an F statistic $F(2;29) = 0.589$ $p = 0.561$ was yielded, concluding that the model developed for use at the bay could not effectively be used at the sites with higher water flow rates

Table 7.10: Performance of Bay model tested with out-of-sample.

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Durbin-Watson
1	0.198 ^a	0.039	-0.027	0.0204	1.785
Independent variable: (constant), DO, SRP Dependent variable: TP					

Further assessment of the results that combined the RMSE, model and data uncertainty and the NSE range (Figure 7.5) confirmed this poor model performance with a Nash and Sutcliffe statistic NSE of $100\% < 0.3$, making it an unsatisfactory model output. The model overpredicts change in TP in water by 56.4% over the mean value if used at a site with different hydrodynamic conditions.



=====GOODNESS-OF-FIT EVALUATION=====

Evaluation of E_I : **UNSATISFACTORY**

Probability of fit being:

- Very good ($E_I = 0.650 - 1.000$): 0.0%
- Good ($E_I = 0.500 - 0.649$): 0.0%
- Acceptable ($E_I = 0.300 - 0.499$): 0.0%
- **Unsatisfactory ($E_I < 0.300$): 100.0% (p-value: 1.000)**

NRMSE=100*RMSE/SD=277%; KGE=-0.338 [-0.639 to -0.045]; N=32

Presence of outliers (Q-test): NO

Model bias: Overprediction by 56.4% of the mean
(E_I may be influenced by model bias)

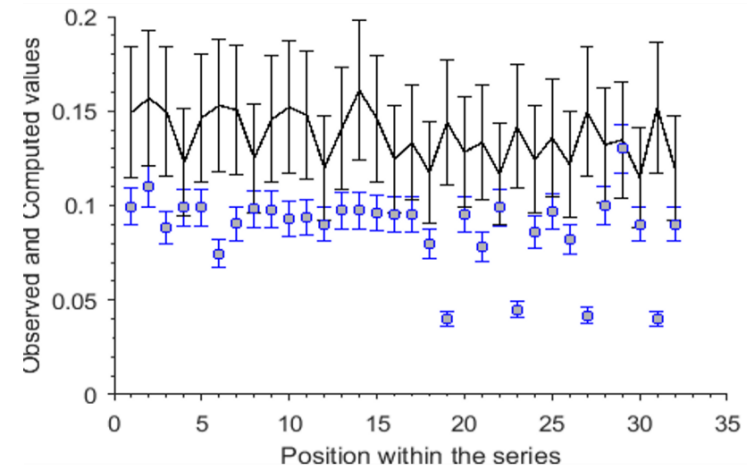


Figure 7.5: Performance of the Bay model tested with river data (out-of-sample testing), analysed on criteria for goodness-of-fit revised from Ritter and Muñoz-Carpena, (2013). A model bias of 3%, calculated coefficient of variation of 9.83% and 22.96% for observed and predicted values respectively.

The bay model was then tested with data collected from the same site. The R^2 value of this model was 67.2%. Model performance (Figure 7.6) showed 19.9% of predicted values being within acceptable range ($NSE = 0.3-0.5$), 46.3% in the good category ($0.5-0.65$), 30.7% within the very good category ($0.6-1.0$) while 3.1% were unsatisfactory. Model performance improved from 100% unsatisfactory when river data was used to 3.1% of the predictions as under unsatisfactory category. The model was found to overpredict TP concentration by 12.4% of the mean.

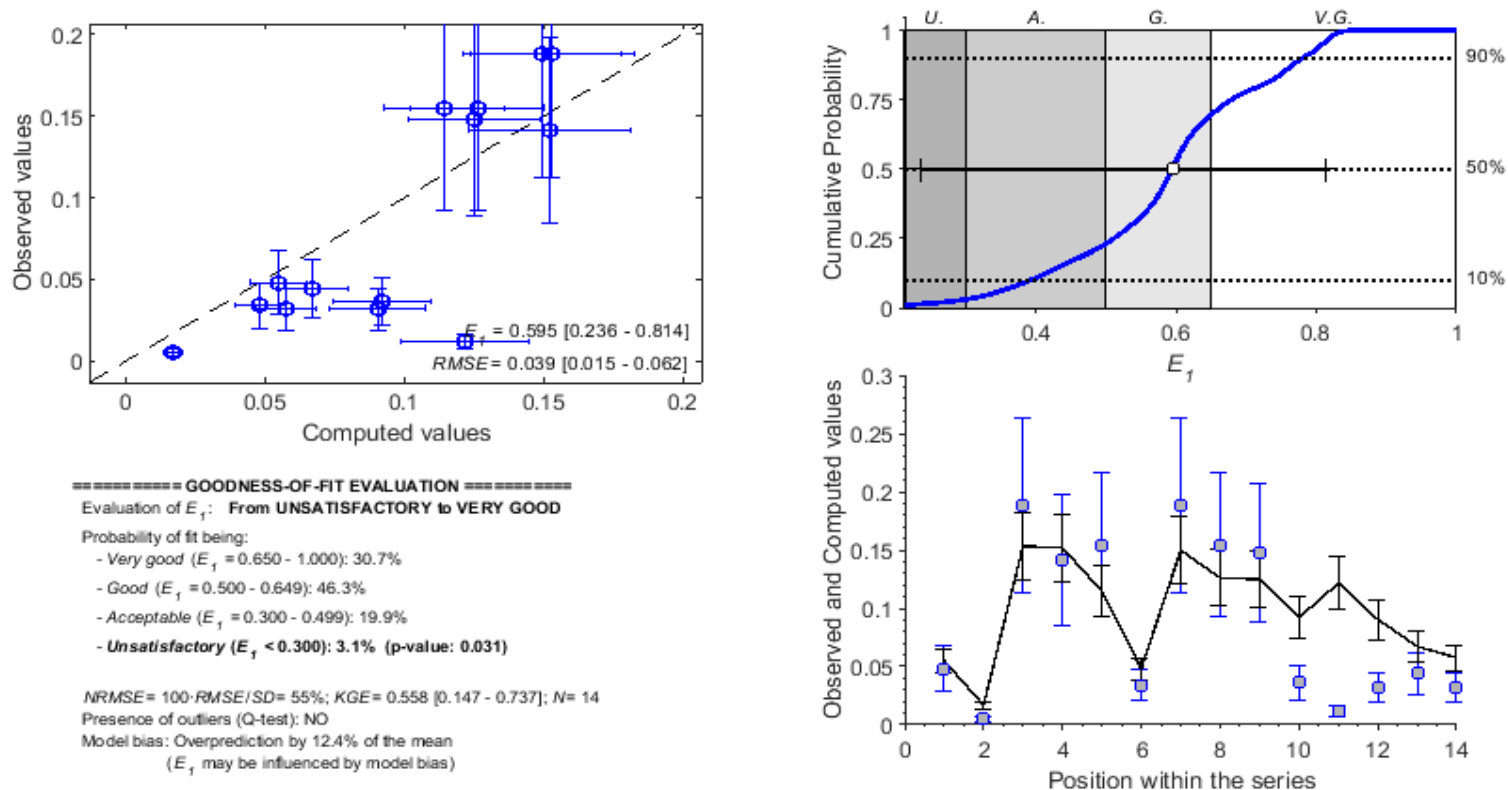


Figure 7.6: Performance of the Bay model, validated with data from the same site. Uncertainty in observed and predicted data, calculated as the coefficient of variation (%) was set at 18.99% and 40.2% for predicted and observed uncertainties respectively.

Finally, the river model was tested with data collected at the river farm which was not used in model training (Table 7.11). The regression model showed an R^2 of 30.2%. The NSE scores (Figure 7.6) indicated poorer performance with 97.9% of the predicted values under the unsatisfactory category and only 2% acceptable probability of fit. The model was found to overpredict TP concentration by over 43.1% of the mean.

Table 7.11: Model summary on testing the river model with validation data from same site.

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate
1	0.550 ^a	0.302	0.070	0.362

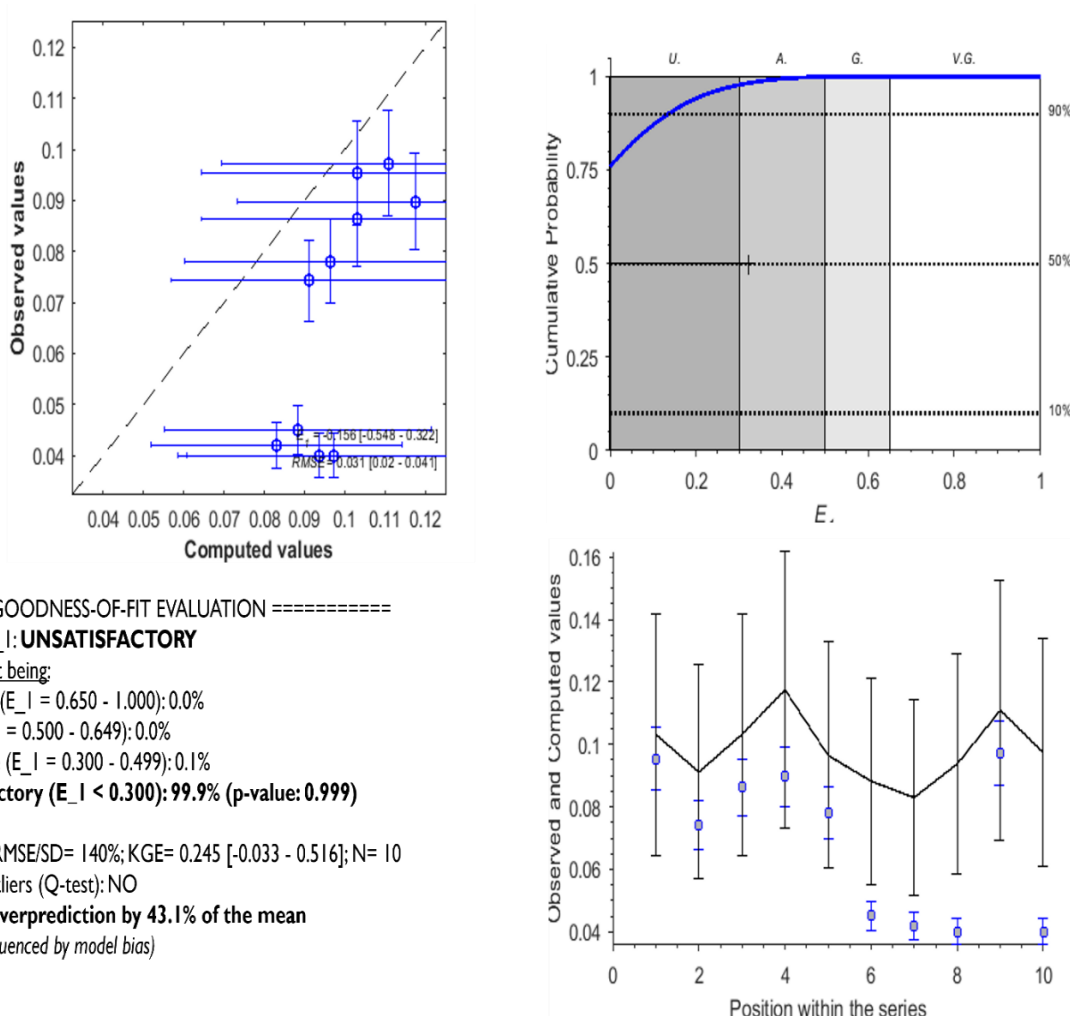


Figure 7.7: Performance of the River model, validated with dataset from same location Observed and predicted uncertainty, expressed as a coefficient of variation was 10.65% and 35.57% respectively and a model bias of 3%

7.4 Discussion

To address the challenge of organic waste enrichment at cage sites, monitoring farm activities and related changes in water is crucial. However, the shortage of testing laboratories and skilled technical personnel has crippled monitoring efforts in the African Great Lakes region for decades (Odada *et al.*, 2004; Plisnier *et al.*, 2018). Farmers and authorities will likely be deterred from monitoring environmental changes due to the deterrent technical and financial costs. As a result, detrimental effects will manifest. Simple regression models can be used by farmers to conduct their own rapid checks at low cost. In this study, the possibility of reducing the number of monitored parameters was assessed. The objective was to retain significant ones that could be used to predict the concentration of Total Phosphorus in the water column. The aim was to generate simple regression models that can be applied to rapidly assess a farm's possible impact on the environment.

Generated for fast and slow water flow rates using data from sites at the river mouth and in a bay on Lake Victoria respectively, regression statistical modelling was applied to identify which of the physico-chemical indicators of changes in the water column were statistically significant and could be most crucial for predicting TP concentrations at cage farms. TP was used as the response variable because it is a key indicator of phosphorus loading commonly observed in monitoring organic enrichment of freshwater systems (Hellawell, 2012). DO, TP, SRP, Ammonium, conductivity, TSS, Temperature, TN, Secchi depth, and pH were used as predictor variables. Using the stepwise regression method only statistically significant variables were retained (Schuenemeyer and Drew, 2010). Chang *et al.* (2015), Varol (2000) have applied similar reduction techniques to generate models that can explain changes in water quality. The reduction in the number of monitoring variables aims to lower the financial burden (Valentini, dos Santos, and Muller Vieira, 2021), providing farms with tools for monitoring. Evaluation of model performance was approached using qualitative and quantitative measures (Harmel *et al.*, 2014). A combined interpretation of the coefficient of determination (R^2), the root mean square error and Nash and Sutcliffe coefficient provided in the FITEVAL tool (Ritter and Muñoz-Carpena, 2013) aided a better comprehension and comparison than relying on the coefficient of determination- R^2 alone where the statistical power is affected by sample size (Cohen, 1992).

Model results indicated that temperature and total suspended solids were significant parameters at the river mouth at high water flow rates. In contrast, dissolved oxygen and

soluble reactive phosphorus were significant at slow water flow rates. At high water flow rates, the regression model showed that 41.1% of the variation in TP at the site could be explained by changes in temperature and TSS. At slow water flow rates, 67.2% of the variability in TP could be explained by changes in DO and SRP, with other factors kept constant. Model fitting analysis is crucial for performance evaluation in predictive modelling (Giancristofaro and Salmaso, 2003; Tredennick *et al.*, 2021). The two models were tested for performance using out-of-sample data. First, for both sites, 22% of the data was randomly extracted for model validation and not used in model training. For out-of-sample testing, the data at the river farm was run in the Bay model and vice versa. This was done to test model performance under similar and different hydrodynamic conditions.

Validating the Bay model yielded a result where 19.9% of the predicted observations were within the acceptable NSE range while only 3.1% were in the unacceptable category. Using data from a site with different hydrodynamic conditions (River) led to a coefficient of determination R^2 of 3.9%. This poor performance was substantiated by the model's NSE range being 100% unsatisfactory (<0.3). This confirms the need for developing models tailored to specific hydrodynamic conditions. The parameter being monitored, TP is affected by several internal and external drivers (Oliveira and Machado, 2013). Hydrodynamic conditions like water currents play a key role in the physico-chemical conditions and the fate of nutrients in the water column (Cioffi and Gallerano, 2000; Zhang *et al.*, 2019). High water movement means fast flushing rates (Beveridge, 2004) and variability in environmental conditions, making it difficult to detect changes in the water column.

Temperature and TSS were retained in the river model. Indeed, temperature would be more stable or constant (less variable than other parameters) along the depth profile especially in shallow sites less affected by thermal stratification. Due to the agitation of lake sediment caused by fast currents, sediment would be suspended into the water column, causing TSS to remain relatively constant. In lakes, suspended solids are influenced by current velocity (Håkanson, 2006). In addition to that originating from external sources, more suspended solids are added to the water column as fish faeces and uneaten feed in the proximity of farms (Schumann and Brinker, 2020). As a result, TSS increases uncertainty if used to explain organic enrichment from cages at a river mouth with high flow rates.

Temperature and TSS would theoretically be of minimal significance when monitoring organic enrichment in cages. Theoretically, explaining the change in TP due to

temperature and TSS changes may be difficult. The statistically significant results obtained by the river model do not have a strong theoretical basis, which illustrates the necessity of applying logic when using regression model outputs. On the contrary, in slow moving waters, changes in TP in the water column can be assessed by SRP or DO (Nowlin *et al.*, 2005). The regression model results showed that 67.2% of variability in TP (mg/l) in a bay could be explained by changes in DO and SRP, with other factors kept constant. Furthermore, a score of 46.3% in the good, 30.7% within the very good category on the NSE scale yielding only 12.4% predicted TP (mg/l) being over the observed mean shows the bay model could be validated with more data and tested for use in waters with slow currents at cage sites.

7.5 Conclusion

Reducing the number of variables monitored will ease the technical and financial burden and provide an incentive for cage impact monitoring. It would be possible to use data collected over time on a given farm site to develop simple regression models that farmers could use to rapidly assess TP levels at the site in a timely manner and without having to collect additional data. Results indicate that statistical regression for parameter reduction can yield better models in sites with slow water currents than in faster water currents. However, the theoretical relevance of the retained regression variables is crucial. Research in the future should focus on validating the models generated in the study with additional independent observations to assess their generalizability. Reducing the burden of parameters that must be monitored on cage farms is one way of promoting the practice of on-farm monitoring. Farmers must be trained and equipped with tools for performing environmental assessments. Keeping track of the impact of changes in water quality will enable them to evaluate their practices. Farmers and regulators need rapid assessment tools given challenges with technical capacity. This will contribute to effective monitoring to match the accelerated exploitation of the African Great Lakes for fish farming.

CHAPTER 8:

GENERAL DISCUSSION AND POLICY RECOMMENDATIONS

8.1 Background

Sub-Saharan Africa is endowed with vast freshwater systems that could be exploited to improve the livelihoods of the population. Due to declining fisheries in Lake Victoria, Uganda, policy directions are aimed at promoting the use of the lake for Tilapia cage aquaculture (MAAIF, 2017b; NPA, 2020). This research aims to contribute to evidence-based decision support for the development of Tilapia cage culture in Lake Victoria, Uganda. An assessment of the status of production is performed, as well as a review of the existing regulatory framework. In addition, stakeholders' perspectives on cage culture development are sought, to identify key focus areas for this research, and managers. The suitability of the sites for tilapia culture in small and large-medium cages is assessed, and because the system is used by multiple stakeholders, an approach to spatial prioritization and scenario analysis is used to analyze the tradeoffs and synergies that must be made for cage culture to develop alongside other sectors. Environmental monitoring techniques for use in licensing and for on-farm rapid assessment are explored. Results and recommendations of this research will be applicable to other tropical lake systems, most notably those within the African Great Lakes region. This chapter provides a summary of the key findings, an interpretation of the findings, implications for aquaculture in tropical lakes, and limitations of the findings.

8.2 Context and Prospects of Cage Culture in Lake Victoria

8.2.1 Stakeholder Perspectives

Exploring the views of those working in fisheries and aquaculture on Lake Victoria on the growth of cage culture, this study employed a specialised approach called Q methodology. Subjective views of those with knowledge of a topic were sought (Watts and Stenner, 2005). By observing patterns of similarity in how they sorted a list of statements, a rich description was gathered that may have been missed if predetermined categories were used (Addams and Proops, 2000; Ramlo and Newman, 2011). This methodology was a combination of both a qualitative (data collection) and a

quantitative approach (factor analysis and principal component analysis). To this end, Ramlo and Newman (2011) referred to it as a mixed method. Similar studies on environmental issues that used Q methodology to obtain views on socially debated topics include Armatas, Venn and Watson, (2014); Howard *et al.*, (2016); Chikudza *et al.*, (2020); Ciftcioglu, (2020). According to this study, stakeholders were optimistic about cage farming's prospects in the region but were concerned about institutional gaps and the environmental consequences of aquaculture. Similar concerns about gaps in regulation and environmental consequences were raised by Njiru, Aura and Okechi, (2018); Cowx and Ogutu-Owhayo, (2019). A general lack of concern about aquaculture's potential societal implications suggests that current levels of growth have not led to any significant conflicts among cage farmers or other users, or that there is a lack of knowledge or information about ongoing community interactions. An assessment of the social license to operate (Rooney, *et al.*, 2014) would provide an understanding of how communities and local stakeholders view aquaculture farms. Aquaculture development in the north focuses on sustainability (Belton, Reardon, and Zilberman, 2020), while in the south, the focus is on food and nutrition security, employment, and income generation. Saguin (2016) illustrates how industrialization characterized by capitalistic investment in fish farming destroyed fishers' livelihoods and ecological conditions on Lake Laguna. National regulatory frameworks should include provisions that protect the ecosystem and the population if the market does not force producers to adhere to ethical principles.

8.2.2 Farm management practices

Tilapia cage farming is expanding but some farmers are installing cages in shallow parts of the lake. The advantage is the ease with which the shoreline is accessed making routine activities and overall operational costs minimal (Morro *et al.*, 2022). However, shallow sites have limited waste dispersion that facilitates organic waste build-up (Cardia, *et al.*, 2017). Similarly, areas within 200m from the lake shoreline are prohibited from use by the environment management authorities (NEMA, 2000). Results indicated that considering bathymetry alone, only 5.6% of the study area is between moderate to high suitability when using small-sized cages. Also, shallow lake parts and enclosed or semi-enclosed bays with depths below 20m were reported to have oxygen concentrations between 0.8-9.2 mg/l (Deirmendjian *et al.*, 2021). While Tilapia can survive at DO levels as low as 0.1mg/l (Lim, 1989b; Abd El-Hack *et al.*, 2022), DO levels in tropical cage culture systems are recommended to be above 4mg/l (Devi *et al.*, 2017).

Poor water quality is directly linked to increased stress in cultured species and is worsened by high stocking densities (Conte, 2004).

One farmer who was part of an association of cage producers stated that the stocking densities were dependent on input availability and that in periods of shortage, trash feeds were used (Majjidu, pers comm.). Njiru, Aura and Okechi, 2018 reported a stocking density of 350 fish/m³ at small cage sites in shallow waters on the Kenyan side of Lake Victoria. Assuming comparable characteristics among Kenyan and Ugandan smallholders, high stocking densities combined with low flushing rates will deoxygenate the water column and promote kills. In Kenya, massive fish kills were reported in cages first in 2016 (Njiru, Aura and Okechi, 2018) and again in 2022 (Alando 2022) causing losses of US \$ 4,300 and US \$11.4 million respectively. Proper site selection will be a major intervention that could improve survivability of smallholders in cage production.

Due to low operating capital, weak competitiveness, limited access to finance and insurance, and increased risk, smallholder producers are more vulnerable to shocks worldwide (Harvey *et al.*, 2014). Similar challenges have been reported for smallholders in aquaculture by Adeleke *et al.*, (2021); and Odende *et al.*, (2022). The feed challenge has contributed to sector stagnation in sub-Saharan Africa for decades (Halwart and Moehl, 2006). Despite the Government of Uganda's efforts to subsidize the importation of feed (Owani, 2013), smallholder farmers have limited access to the same due to capital constraints (Odende *et al.*, 2022).

In contrast, larger farms used mostly pelleted floating feeds whose proximate analyses indicated most parameters were within suitable Tilapia culture limits. For instance, 5mm feed (with fish meal as the protein source) meant for big fish (>200gm) had an average of 29.9% protein suitable for bigger fish (Abd El-Hack *et al.*, 2022) while crude fiber was up to 8.77%, which is ideal for Tilapia (FAO, 2023). Most of these large farmers stated that they imported their feeds which was not an option very accessible to smaller cage farmers. It also proved inaccessible for small players to obtain multiple licenses and permits for cage farming from different institutions, despite struggling to produce at low working capital, and yet sell in the same markets as the bigger players. Netherlands Enterprise Agency, (2022) praised Uganda as having a good business environment in its assessment of the aquaculture roadmap for Uganda. In addition, it was stated that large-scale investors had more straightforward access to business structures than smaller investors, and several small cage farmers had exited the business following flooding in the lake basin in 2019. Government strategic planning needs to devise mechanisms that target levels of production; small, medium, and large, to ensure

levelled grounds for improved competitiveness, retaining of local skills, employment, increased sustainability of businesses and better livelihoods for all.

8.2.3 Regulatory framework; EIA and its limitations

In Chapter 2, it is revealed that institutions and regulatory framework were available albeit with weaknesses on key aspects. Environmental Impact Assessment (EIA) was a common tool suggested for monitoring and regulating cage farming on the lake. Given the state the lake is in i.e., interconnected, eutrophic with some species being extinct and stocks of others dwindling (Downing *et al.*, 2014), failed efforts to control catchment activities that contribute the bulk of nutrient loading, present a major challenge that has persisted for decades (Keys *et al.*, 2019; Njagi *et al.*, 2022). This implies that available tools to ensure sustainability should be used effectively. EIA needs to contribute to sustainable growth of cage culture rather than being a tick-box exercise (Alberts *et al.*, 2020). Among other challenges reported elsewhere are the political context and its influence on the final decision (Ho, Nor-Hisham and Zhao, 2020) and political interference in the EIA process. This disregard of scientific recommendations (Christopher, 2007) preempts the goal of the EIA, which is to improve the quality of political decisions (Ortolano and Shepherd, 1995). Important to recognize is that some of the impacts aquaculture might have on the lake are cumulative (Beveridge, 2008) and the EIA scoping process should be strengthened with predictive modelling (SEPA, 2019). Reducing the voluminous technical information for decision support (Christopher, 2007) would be necessary if this process is to contribute to the sustainable development of cage culture in Lake Victoria.

8.3 Spatial planning and optimization

As part of the EAA, the 3rd principle focuses on integrating aquaculture with other policies and goals, while the 1st principle acknowledges that ecosystem services should not be compromised in the process (Ross *et al.*, 2013). A significant challenge in freshwater systems is the lack of clear integrated frameworks for planning and managing aquaculture in conjunction with other ecosystem services and goals (Langhans *et al.*, 2019). Chapter 5 presented spatial prioritization simulation done with the Marxan with Zones (MarZones) software (Ball, Possingham and Watts, 2009; Watts *et al.*, 2009),

used to evaluate the tradeoffs and synergies that needed to be made between cage culture and other provisioning ecosystem services provided in the lake. A form of strategic planning, the results inform the process of comparing management alternatives (Getz *et al.*, 2018).

Results revealed sections of the lakes that were more prone to conflict and the potential outcomes when different resource use scenarios were employed. For the first time, results that objectively inform licensing decisions for multiple activities in the lake were generated. This tool could significantly contribute to the establishment of aquaculture parks planned by authorities (MAAIF, 2017b) and the implementation of strategies like the polluter pays principle (NEMA, 2019). Factors considered would be relevant attributes for or against cage farming in addition to holistically evaluating implications on other ecosystem services that could spatially conflict with park space. A priori planning can prevent conflicts and mitigate irreversible environmental changes since ecosystem health can be factored into the costing process. The results of Chapter 5 are significant for planning multiple uses on African Great Lake systems that are heavily degraded, yet they serve large populations.

Spatial prioritization tools that aim to plan multiple uses in one space according to priority, with the least amount of tradeoffs and maximizing synergies have primarily been studied in the terrestrial and coastal realms (Hull *et al.*, 2011; Delavenne *et al.*, 2012; Snäll *et al.*, 2016). Considering the constraints of water quality in the lake, the highest tradeoff was associated with recreational services and space allocated for fish breeding grounds in all scenarios of cage culture growth. The means that space for these two services needs to be demarcated and excluded when licensing other activities that pose potential risk. It was revealed that cage farming could spatially expand with limited impact on fishing, space for ports and navigation on the lake. However, to achieve this, space optimization is required. Specifically, fishing needs to be allowed within cage farming spaces, while recreation services could be allowed within fish breeding areas if targets are to be met. Results indicated the availability of space for fish breeding in the open parts of the lake. As a result of this approach, licensing activities on the lake can be objectively evaluated in terms of costs (tradeoffs) and benefits (synergies), ensuring efficient resource use and equitable distribution. Despite being highly valued in research and academia (Janßen *et al.* 2019), the use of such strategic planning simulation software for Lake Victoria decision-making is a much-needed strategy. Natugonza *et al.*, (2020) utilised similar strategic simulation approaches to identify trade-offs between conservation and other socio-economic objectives in Lake Victoria and emphasized the

necessity of utilization of strategic modelling in informing policy directions. By utilizing such strategic models, decision makers can better examine costs and benefits without being influenced by economic rationality and political bias.

A limitation to the spatial optimization done in Chapter 5 was the failure to integrate connectivity in flows of the ES (functional or ecological) (Lieberknecht *et al.*, 2010). Environmental processes are highly dynamic and complex to exhaustively incorporate in spatial modelling which presents an enormous challenge in evaluating forecasts and uncertainties in the results. However, model adequacy is driven by data availability, and the urgency with which environmental decisions need to be made (Getz *et al.*, 2018). Obtaining quality data of enough resolution for modelling in Lake Victoria was a challenge. However, the use of proxy data, scenarios or model outputs provided an alternative in this study.

Contrary to holistic decision support that considered multiple objectives at once, the site suitability assessment done in Chapter 4 considered only attributes that were relevant to cage farming alone. In other words, a multi-criterial evaluation as opposed to the multi-objective evaluation done in Chapter 5. Results revealed that from the expanse of the lake in the Ugandan sector, only 9.7% was moderately to highly suitable for Tilapia fish farming in medium-large cages while the largest part, 86.4% had low to very low suitability. Similarly, only 3% was between moderately to highly suitable for small cages, while the largest portion of the lake, 93.1% had low to very low suitability. As aquaculture expands, suitable sites in the inshore parts of the lake will keep reducing and potentially become riskier. This is due to an observed trend of new farms setting up close to existing ones. The reason for the crowding was not well established but could be linked to ease of obtaining farmer-to-farmer assistance (Aura *et al.*, 2018) or an assumption that existing farms were situated in suitable sites.

The final suitability models would be strengthened when wind speed data is included. To make lake modelling better, the mechanistic details of the processes that drive the state of the environment need to be incorporated. Modeling the dynamics of Lake Victoria for planning aquaculture or fisheries will depend on the availability of data with a suitable resolution and in a format that is suitable for analysis. Musinguzi, Natugonza, and Ogutu-Ohwayo, (2017) provide an explicit explanation of the challenges involved with applying ecosystem models within the African Great Lakes region highlighting a lack of data and technical know-how as key.

The trade-off analysis scenarios in Chapter 5 showed it was less costly to achieve targets on ecosystem services that were suitable in the deeper open waters of the lake. Due to the increased competition for space as well as the pollution coming from the catchment (Njagi *et al.*, 2022) into the enclosed and semi-enclosed bay, it is necessary to explore development in deeper locations. Murchison Bay, for example, has been reported to have periods of DO as low as 0.0mg/l (Akurut, Niwagaba and Willems, 2017), extremely unsuitable conditions for sustaining cultured Tilapia. Based on the findings of the suitability assessment and the trade-off analysis, it is evident that, in the event cage farming needs to expand without adversely impacting other ecosystem services or ecosystem health, deeper sites will have to be considered. To operate effectively, it is imperative to strike a balance between the production carrying capacity of the site and the cost of doing business. As advancements in cage production like improved cage facilities and remote feeding technologies (Morro *et al.*, 2022) become more accessible, the feasibility of offshore cage farming in Africa's Great Lake systems becomes a foreseeable option for expanding the industry. A cost-benefit analysis of near and offshore production sites is necessary. However, more research into the dynamics of phosphorus internal loading at offshore sites especially during periods of deep thermal stratification would be needed. Under natural conditions, Tilapia is known to do well at 0-20m depth (Njiru *et al.*, 2008), further research into its performance in deeper site cages (e.g., 20-40m depth) will be required.

8.4 On-farm monitoring criteria

In Chapters 6 and 7, focus is drawn on the environmental aspects of cage farming in the Lake. These two chapters contribute to identifying techniques suitable for impact assessment with a goal of licensing by setting on-farm production limits (Chapter 6) and on-farm rapid assessment (Chapter 7). In Chapter 6, an evaluation of three techniques of impact assessment i.e., water quality assessment, nutrient mass balance and benthic invertebrate assemblage analysis is done. Best-technique identification has been used in the past to establish the most ideal method of evaluating response to human interaction (Poikane *et al.*, 2016). Using results from a commercial cage site, the technique identified as suitable for on-farm monitoring to set production limits in Lake Victoria was nutrient mass balance modelling together with the analysis of the benthic invertebrate assemblages. It was revealed that regardless of evidence of organic enrichment at cage farms, establishing the impact of farm production based on

traditional water quality assessment alone (physico-chemical and nutrients) did not show significant differences in conditions between aquaculture and non-aquaculture sites in eutrophic tropical lakes like Lake Victoria. Regardless of an increase in feeding, similar results were obtained by Mwebaza-Ndawula *et al.*, (2013); Kashindye *et al.*, (2015); Musa *et al.*, (2022); and Okechi *et al.*, (2022) who compared physico-chemical and nutrient (TN, TP) conditions in water at cage and non-cage sites in Lake Victoria. Similarly, Gondwe, Guildford and Hecky, (2011) found comparable results studying cages in the southern part of Lake Malawi, another tropical lake system of major importance. Similar results of no significant difference in water quality and nutrient condition were obtained by Nabirye *et al.*, (2016). This situation was not unique to Lake Victoria or other African tropical lakes but had been reported in other lakes associated with cage farming like Lake Huron (Gondwe, 2009).

In cage farms, the fate of organic waste was influenced by biotic and abiotic factors, such as physico-chemical conditions at the sediment-water interface and mixing in the water column (Otu, Bureau, and Podemski, 2017). Solid waste from the cages could be quickly eaten by wild fish or the dissolved portion assimilated into other components of the food web in the vicinity of the cages. In Lake Victoria, hydrodynamic conditions vary throughout the day, diurnally, seasonally, and annually (Ssebuggwawo *et al.*, 2000), making physico-chemical parameters and nutrient concentrations highly variable. As a result, organic enrichment from cages was difficult to correlate with water quality conditions due to its dependence on prevailing environmental conditions. In addition, allochthonous sources of waste such as catchment runoff and atmospheric deposition (Scheren, Zanting and Lemmens, 2000; Tamatamah, Hecky and Duthie, 2005) massively contribute to the nutrient enrichment of the lake. Other methods of analysis, such as isotope analyses as in Gondwe, Guildford and Hecky, 2012; Yuan *et al.*, 2019, are required to quantitatively isolate organic loading from cages with that from other sources. However, the technical expertise to perform such analyses within the region is limited. This is not to assume impacts where there are not, but to confirm that in tropical lakes not limited in nutrients, using water quality assessment (physico-chemical and nutrient conditions) does not provide an accurate indication of on-farm organic loading making it complex to conclusively link farm activities to ecological change in the system. This reduces the precision of water quality assessment in setting on-farm production limits.

The nutrient mass balance model produced comparable results i.e., average of 14.11kg TP, 274.12kg Carbon and 86.97kg TN/ton to similar models done for Tilapia cage culture

tropical lakes. Gondwe, Guildford and Hecky, (2011a), estimated 10.75kg TP, 26.5kg TN, and 25kg Carbon/ton of fish while Montanhini Neto and Ostrensky, (2015) found 14.26 kg TP, 44.95kg TN and 1040.63kg of Carbon/ton of fish. To inform licensing decisions, mass balance models quantify waste in terms of load, but do not indicate the actual impact on the environment. When applied in isolation, both water quality assessment and mass balance modelling techniques were found to be insufficient for decision support when setting on-farm production limits. This is not to mean water quality assessments should not be done, because the monitored parameters are critical for fish growth, health, and welfare. Similarly, nutrient balance models could be explored for rapid assessment of on-farm performance in terms of feed usage i.e., uptake of feed or wastage. For licensing purposes, more precision is required in predicting changes in the environment due to organic loading from the cages.

From assessing benthic conditions between cage and control sites, the results in Chapter 6 indicated differences in species richness, diversity (H') and the Heip's Evenness Index (HE). These differences in site condition were not established using water quality or the nutrients in the lake sediment assessment. Benthic macro-invertebrates are reliable ecological proxies of water quality. Benthic macro-invertebrates' assessments were easy to implement, less costly and required limited laboratory analysis (Mandaville S, 2002; Odountan *et al.*, 2019) compared to water quality assessments. The ASC Benthic Technical Working Group, (2022) recommends a system that classifies the farm impact based on the benthic quality. A similar assessment was used by Sekiranda, Ndawula and Gandhi, (2004) to establish the impact of land use on the water quality in three bays on the Ugandan sector of Lake Victoria. An ecological quality standard generated from analysis of performance against several sediment quality indices (ASC Benthic Technical Working Group, 2022), specifically set for freshwater environments could be a robust tool for monitoring farm production and how it impacts on the ecology of the lake.

Recommended is a tiered system for monitoring and licensing such as ASC Benthic Technical Working Group, (2022) that is based on site specific hydrodynamic and limnological characteristics as well as farm management practices. A combination of nutrient balance models, which indicate the amount of nutrients added to the environment, and biomonitoring for benthic quality would be ideal. Monitoring frequency and timing should be planned to capture seasonal dynamics (Mandaville S, 2002). Based on the results of the nutrient balance model, sampling criteria could be applied depending on the farm's level of waste generation; from rapid semi-quantitative

(presence or absence of indicator species), to detailed biomonitoring that leads to species identification (Mandaville S, 2002; ASC Benthic Technical Working Group, 2022). Accordingly, the results could range from relying on a single biodiversity index, to calculating a range of indices, whose compliance thresholds are determined based on research. A multi-metric index approach as used by ASC Benthic Technical Working Group, (2022) and suggested by Odountan *et al.*, (2019) is recommended. However, species tolerance ranges specific to tropical conditions will have to be devised (Ochieng, Okot-Okumu and Odong, 2019). In this research, relying on species richness, diversity and evenness improved the ability to conclude on conditions in the cage and non-cage sites. In order to ensure that farms are allocated production limits that are not biased by the activities that happen in the background on the site, it is critical to understand better criteria for licensing. Furthermore, the strategy could be used to inform interventions such as fallowing, which ultimately improves how aquaculture affects the environment and how the environment affects aquaculture. Prior to further expansion, data on baseline indices of environmental change should be collected to facilitate on-farm monitoring in the future.

Chapter 7 focused on regression modelling to obtain simple models for farm rapid assessments. Data from two cage sites was used to depict different hydrodynamic conditions in the lake. Using stepwise regression, the number of physico-chemical parameters tested was reduced to retain only those that were statistically significant. Stepwise regression is criticized for dropping predictor variables that are not statistically significant but are of importance in predicting the regressor (Smith, 2018). However, when the objective is to retain only those parameters that significantly explain variations in the regressor variable, the stepwise approach is commonly applied (Bachmair *et al.*, 2018; Franklin *et al.*, 2020; Rao and Yan, 2020). As data becomes more available, such statistical models could contribute to the monitoring challenges that have engulfed the African Great Lakes region for decades (Plisnier *et al.*, 2022). In this study, two models were generated to investigate how water quality variables collected at a cage site could explain changes in TP in the water column. Better performance was observed with the model generated for low water flow rates where DO and SRP were retained as predictor variables. Further, the results of this study revealed that in addition to examining the performance indicators of the model, consideration must be given to the theoretical meanings of the variables retained as predictors. Secondly, it was revealed that regression models for use in Lake Victoria would have to be site specific.

The models generated in this study would require more data for validation. It is pertinent to note that farms can use these tools to quickly evaluate the quality of water at their facilities. Regression models of this nature could be used to circumvent the prohibitive costs and technical burden of sampling and analysis. However, such models are highly dependent on the availability of substantial datasets, which was a major limitation of this study. More data needs to be collected in other enclosed and semi-enclosed bays to test the regression models generated in this research. Phosphorus internal loading is a key aspect of modelling in lake systems. Understanding the loss of particulate P to the lakebed and the fate of phosphorus generated from internal loading will improve the ability to predict the effect of cage culture on the lake if dynamic models are used.

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