ASSESSMENT OF ANGUILLID EELS' BIO-ECOLOGY AND ECOLOGICAL STATUS IN ATHI-GALANA-SABAKI AND RAMISI RIVERS

KARIUKI CATHERINE WANJIRU

A Thesis Submitted to the
Graduate School in Partial Fulfilment of the Requirements for the Doctor of Philosophy Degree in Environmental Science of Egerton University

EGERTON UNIVERSITY

JUNE, 2025

DECLARATION AND RECOMMENDATION

Declaration

This thesis is my original work and has not been presented in this university or any other for the award of a degree

Signature: Date: 12th June, 2025

Catherine W. Kariuki

ND12/16555/18

Recommendation

This thesis has been submitted with our approval as University supervisors

Signature________________________________Date: 19th June, 2025

Dr. Charles M. Kihia

Egerton University

Kenya

Signature: ________________________ Date: 19th June, 2025

Prof. Gordon O'Brien

Charles Sturt University

South Africa

Signature: _______________________ Date: 1st June, 2025

Dr. Oscar Donde

Egerton University

Kenya

COPYRIGHT

DEDICATION

I would like to dedicate this work to my family; my husband Mac Githinji, my daughter Ray Wambui and my son Ian Gitonga for the love and support they gave me during the study.

ACKNOWLEDGEMENTS

The author would like to thank the West Indian Ocean Marine Science Association (WIOMSA) for funding this project. I thank Egerton University through the staff development program for allowing me to study and waiving my tuition fees. My regards go to the Kenya Marine and Fisheries Research Institute for providing space and fieldwork equipment during the study. I would like to most sincerely thank my supervisors, Dr. Charles Kihia (Ph.D.), Prof Gordon O'Brien (Ph.D.), and Dr. Oscar Donde (Ph.D.) for their tireless guidance that added quality to this work. My gratitude goes to Mr. Boniface Kavisi, Suleiman Mtawazo, Ali Mfunga, Jira Mbolu, and Roman Mwashenga for their assistance in the field. Special thanks to Dr. Celine Hanzen (Ph.D.), Ms. Annelize van der Merwe, and Ms. Angelica Kaiser of South Africa for training me in eel sampling methods and the use of various equipment relevant to eel research.

ABSTRACT

The conservation of anguillid eels in Kenya's Athi-Galana-Sabaki (AGS) and Ramisi Rivers is crucial due to their ecological and economic significance and the increasing threats from habitat degradation. This study aimed to: analyse the population structure of anguillid eels, explore their ecological niches, and determine the ecological risks they face in these rivers. Fieldwork was conducted over 12 months, during which 393 eels were sampled using fyke nets. Population health was assessed through length-weight relationships and condition factors, while ecological niche dynamics were determined via stomach content analysis and habitat characterization. Bayesian models were applied to evaluate current and future ecological risks. The results revealed differences in population structure and ecological dynamics between the rivers. The Ramisi River supported a larger eel population (mean density 1.1 individuals/net/day) than the AGS River (0.8 individuals/net/day). *Anguilla bengalensis* was the most abundant species (65%), and yellow eels in the AGS River exhibited a higher mean length (48.9±13.7 cm) compared to those in the Ramisi River (34.4±12.4 cm). Silver eels also demonstrated greater lengths in the AGS River (88.0±2.3 cm) than in the Ramisi River (66.8±8.4 cm). Ecological niche analysis revealed the utilization of nine dietary items, with the highest dietary overlap observed between *A. bengalensis* and *A. bicolor* (Schoener's index: 0.82). Anthropogenic activities, including water abstraction and riparian encroachment, were more pronounced in the AGS River, leading to greater habitat degradation (habitat integrity score of 56.6% compared to 92.3% in the Ramisi River). Risk analysis identified the yellow eel stage as the most vulnerable under future scenarios, underscoring the need for proactive conservation. This study concludes that anguillid eels in Kenya's East-flowing rivers recruit across diverse size classes and habitats but face significant threats from human activities. It recommends urgent habitat restoration, regulation of water extraction, and community-driven conservation initiatives to safeguard these ecologically and economically vital species.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS AND ACRONYMS

AGS	Athi – Galana - Sabaki
ANOVA	Analysis of Variance
CPUE	Catch Per Unit Effort
ECU	European Currency Unit
EIFAC	European Inland Fisheries Advisory Commission
ICES	International Council for the Exploration of the Sea.
FAO	Food and Agriculture Organization
GoK	Government of Kenya
GHGs	Greenhouse Gases
IUCN	International Union for Conservation of Nature
mm	Millimetres
RRM	Relative Risk Model
U.S.A.	United States of America
UNESCO	United Nations Educational, Scientific, and Cultural Organization
WIO	Western Indian Ocean
WIOMSA	Western Indian Ocean Marine Science Association

CHAPTER ONE
INTRODUCTION

1.1 Background to the study

The taxonomic group of freshwater Anguillidae, known as the genus *Anguilla*, encompasses 19 species worldwide (Arai, 2016). These species inhabit freshwater, brackish estuaries, and coastal waters across more than 150 countries (Jacoby & Gollock, 2014). Due to spending more than 90% of their lives in freshwater environments, they are primarily considered freshwater fish (Béguer-Pon *et al*., 2017). Eels of the *Anguilla* genus are highly migratory catadromous species with three freshwater stages (glass, yellow, and silver eels) and two marine stages (egg and larvae). Eggs hatch into Leptocephali (Larvae), which drift along ocean currents to the continental shelf. Leptocephali then metamorphose into glass eels and migrate upstream as elvers (Aarestrup *et al*., 2009; Kuroki, 2023). Elvers further metamorphose into yellow eels (immature stage) within freshwater habitats. Yellow eels then mature upstream into adult silver eels, which migrate downstream to ocean spawning areas where they spawn and perish.

Anguilla eels are exploited globally at all life stages, with glass eels particularly sought after for food and aquaculture (Drouineau *et al*., 2018). East Asia has been the primary destination market for eels, with China and Taiwan accounting for nearly 75% of eel exports by volume, primarily to Japan (FAO, 2015). Along the east coast of Africa in the West Indian Ocean, five species have been documented, although *Anguilla obscura* Günther 1872 was considered an incidental occurrence (Jubb, 1964), and little information exists about it. Among the four main species found in East African rivers, *Anguilla bicolor* McClelland (1844), *Anguilla marmorata* Quoy Gaimard (1824), *Anguilla mossambica* Peters (1852), and *Anguilla bengalensis* Gray (1831), *A. bicolor* is less abundant (Jubb, 1964). Some Western Indian Ocean countries like Madagascar, Mauritius, and South Africa have entered global trade and could play a growing role in the future (Crook, 2014).

Declining eel populations have led to various species being listed on the IUCN Red List of endangered species. The European eel (*Anguilla anguilla*) is classified as "critically endangered" (Jacoby & Gollock, 2014), while American and Japanese eels are categorized as "endangered" (Jacoby & Gollock, 2014; Jacoby *et al*., 2015). *Anguilla reinhardtii* and *Anguilla marmorata* are listed as "least concern" (Pike *et al*., 2020). Consequently, the European Union (EU) has prohibited all imports and exports of *A. anguilla* to and from its member states to curb further decline and conserve dwindling populations. With dwindling temperate eel supplies, attention has shifted to less studied tropical eels, such as the increasing demand for *A. mossambica* glass eels for stocking ponds in East Asia (Fan *et al*., 2008). Despite fewer data available for tropical *Anguilla* species, conservation concerns also exist for many, including *Anguilla bicolor*, *Anguilla mossambica*, and *Anguilla bengalensis*, classified as "near threatened" by the IUCN (Pike *et al*., 2019).

Human activities have altered aquatic ecosystems, inducing stressors affecting eel assemblages, such as pollution, river modifications, overharvesting, introduction of alien species, and climate change (Kuehne *et al*., 2023). These stressors may impact eels' physiology, population, and ecosystem functioning, directly or by facilitating opportunistic organism growth. For example, chemical or mechanical activities leading to lesions may promote opportunistic microorganism growth, causing diseases. The stressors result in unhealthy eel assemblages, loss of diversity, and ecological dysfunction (Arai, 2016). Thus, assessing the potential impacts of the stressors on eel assemblages for resource management and conservation measures is essential.

Eels' effective habitat utilization stems from successful migration from marine to freshwater habitats without barriers. Such effective habitat use enhances feeding habits, growth, and maturity, essential for a successful eel life (Alexandre *et al*., 2016). The construction of barriers like dams and weirs, along with associated changes in river hydrology and hydraulics, poses significant threats to eel habitats (Koster *et al*., 2020). These impacts affect individual or group eels, ultimately influencing species assemblages, including species richness, relative abundance, and other ecological characteristics. An unpublished study by Kihia *et al*. (2022), sheds light on anguillid eels in Kenya, indicating that they have been overlooked as a resource and have not been thoroughly studied in terms of their biological or ecological status. In conclusion, this study will assess the bio-ecology and ecological status of anguillid eels in the Athi-Galana-Sabaki and Ramisi Rivers, shedding light on their population dynamics, habitat utilization, and potential threats within these riverine ecosystems.

1.2 Statement of the problem

Despite the East African anguillid species namely; *Anguilla bicolor*, *Anguilla bengalensis*, *Anguilla marmorata* and *Anguilla mossambica* being listed in different categories of the IUCN red list of threatened species, detailed information about their current status and the threats they face in Kenya remains scarce. Various anthropogenic activities, including damming, agriculture, industry, and river modification, have been shown to affect the health of anguillid populations. In Kenya, east-flowing rivers originating from the central highlands are increasingly exploited for agriculture, industrial activities, channelization projects, and water abstraction. These alterations degrade habitat quality and disrupt anguillid migration, impacting both upstream and downstream movements and threatening the survival of local eel populations. Anguillids' unique catadromous life cycle, which requires them to migrate between freshwater and marine environments, is particularly vulnerable to barriers that obstruct their movement, such as dams and weirs. These obstructions can prevent eels from completing their life cycle, leading to reduced maturity and reproduction rates, ultimately contributing to population declines. Additionally, pollution from agriculture and industrial sources threatens the biology of eels by impairing vital physiological and reproductive functions. The Athi-Galana-Sabaki River, which flows through densely populated agricultural and industrial areas, has become one of Kenya's most polluted rivers, subjecting its aquatic ecosystems to multiple stressors. To protect the *Anguilla* eel species from further decline, it is essential to understand their population characteristics, ecological niches, and risks impacting their lifecycle.

1.3 Objectives

1.3.1 Broad objective

The broad objective was to contribute to eel conservation efforts by assessing the population structure, ecological niche, and conservation status of anguillids in rivers that flow eastward in Kenya.

1.3.2 Specific objectives

i. To analyse the population structure of anguillids in the Athi-Galana-Sabaki and Ramisi rivers.
ii. To explore the ecological niches of anguillids in the Athi-Galana-Sabaki and Ramisi rivers.
iii. To determine the ecological risks against anguillids in the Athi-Galana-Sabaki and Ramisi rivers.

1.4 Hypotheses

i. There is no difference in the population structure of anguillids of River AGS and River Ramisi.
ii. There is no difference in ecological niches of anguillids of River AGS and River Ramisi.

iii. There is no difference in ecological risks against the anguillid eels of the Athi-Galana-Sabaki and Ramisi Rivers.

1.5 Justification

Monitoring the population characteristics of anguillid eels over time reveals trends that indicate whether their populations are thriving, declining, or stable. These trends can guide conservation efforts and assess the success of existing measures. anguillid eels hold both ecological and economic significance: they play an essential role in their ecosystems by contributing to nutrient cycling, habitat formation, and supporting biodiversity, while also providing a livelihood resource for local communities. Their status as indicators of ecosystem health allows changes in their populations to reflect broader environmental issues like pollution, climate change, and habitat degradation, which may impact other species as well. Given that anguillid eels are listed as vulnerable by the IUCN, understanding their population dynamics is critical for ensuring their survival and the continued ecological and economic benefits they provide. This study establishes a scientific basis for protecting anguillid eels and the ecosystems they inhabit, guiding policy and promoting sustainable management practices. The research also supports Kenya's Vision 2030 'Environmental Pillar' by identifying ecological threats in the Athi-Galana-Sabaki (AGS) and Ramisi Rivers, such as, pollution, habitat destruction, and water extraction; thereby contributing to strategies for sustainable riverine ecosystem management. Additionally, it aligns with Sustainable Development Goal 15, which emphasizes sustainable ecosystem management and the protection of biodiversity. Anguillid eels, as a component of Kenya's freshwater biodiversity, serve as indicators of ecosystem health, making their study critical for highlighting and addressing the need to protect these habitats from anthropogenic pressures such as pollution and habitat fragmentation.

1.6 Scope /Assumptions /Limitations

Scope

The scope of this study includes assessing the population structure, ecological niche, and ecological risks facing anguillid eels in the Athi-Galana-Sabaki and Ramisi Rivers. This will involve collecting data on anguillid populations, their habitat preferences, and the current and future threats each life stage encounters within these river systems. Field surveys, data analysis, and interpretation will be conducted to gain insights into anguillid eels' bio-ecology and overall ecological status in these rivers.

Assumption

Anthropogenic activities (e.g., pollution, dam construction, and water extraction) are assumed to influence eel population health and migration patterns significantly. Thus, any changes in population characteristics could be attributed to these activities, at least in part.

Limitations

i. The study duration did not capture seasonal variations or long-term trends in anguillid populations and their ecological niche assessments, because the study was time-bound.

ii. There is a scarcity of literature on tropical anguillids and none in the East African region by the time of this study

1.7 Definition of terms

Alien species - Species that occur outside their natural range and dispersal potential. Alien species are spread by intended or unintended human activity to new areas.

Catadromous – Fish or other organisms growing in fresh water and breeding in the Ocean

Ecological health - Ecological health of a species refers to the overall well-being and functioning of a species within its ecosystem. It encompasses various factors such as population size, genetic diversity, habitat quality, interactions with other species, and overall resilience to environmental changes (Schaeffer *et al*., 1988)

Ecological risk assessment - (EcoRA) Involves the assessment of the risks posed by anthropogenic threats.

Glass eel - refers to young, unpigmented eel, recruiting from the sea into continental waters

Habitat Fragmentation happens when parts of a habitat are destroyed, leaving behind smaller unconnected areas.

Habitat integrity - "The intactness of ecosystems and associated ecological processes, as measured by indicators that capture a) the extent, quality and function of ecosystem components (including biotic and abiotic factors), and/or b) anthropogenic pressures as a proxy for ecosystem degradation and loss." (Watson *et al*., 2018)

Habitat loss –The elimination or alteration of the conditions necessary for animals and plants to survive

Habitat preference - Habitat preference is the habitat most likely to be chosen by a species given the opportunity or the habitat the species is best suited for.

Habitat use -It i**s** the way an animal uses the physical and biological resources in a habitat**.** Habitat may be used for foraging, cover, nesting, escape, denning, or other life history traits

Human disturbance- ‘any human activity that impact the quality and/or quality of air, land and water

Nodes - Nodes in ecological risk assessments represent different components such as stressors, impacts, species, habitats, or other relevant factors that interact within the ecosystem. These nodes are interconnected through probabilistic relationships that help model the complex interactions and dependencies between various elements in the ecosystem. They play a crucial role in ecological risk assessments as they allow for the representation of uncertainties, dependencies, and interactions among different variables within the system. By defining nodes and their relationships, risks associated with specific stressors or changes in the environment can be quantified and potential impacts and ecological endpoints evaluated (Appendix 7).

The North Atlantic Oscillation (NAO) is a climatic phenomenon characterized by a change in atmospheric pressure patterns over the North Atlantic Ocean. It is a large-scale seesaw in atmospheric mass between the subtropical high-pressure system near the Azores and the subpolar low-pressure system near Iceland. The NAO has a significant impact on weather changes in the North Atlantic region, influencing temperature, precipitation, and storm tracks.

Recruitment refers to the number of fish that survive to enter a fishery. These fish have to pass through several life history stages (e.g., egg, larva, juvenile, etc.).

Risk is the probability or likelihood of a prescribed undesired effect occurring and impacting an environment (Suter, 1993).

Silver eel refers to the seaward migratory eel life phase following the yellow eel phase (ICES 2012).

Silver eel Escapement is the downstream migration of Silver eels into the Ocean to breed.

A stressor is any physical, chemical, or biological entity that can induce an adverse response to the structure and function of an ecosystem (United States Environment Protection Agency (USEPA), 1998).

Yellow eel refers to the life-stage resident in continental waters

Yellow eel maintenance is the ability of the ecosystem to keep the yellow eels in good condition

CHAPTER TWO
LITERATURE REVIEW

2.1 Life cycle and growth stages of anguillid eels

Anguilla eels have six life growth stages, namely: egg, Leptocephali (Larva), Glass eel, elver, yellow, and silver eel stages. The duration taken to change from one developmental stage to the other is well documented for temperate eels, but similar information for tropical eels is scarce. The leaf-like eel leptocephali traverse tropical Oceans growing to 75–90 mm within one to three years, before reaching the continental slope of coastal nations where they transform into transparent elongated glass eels that recruit into river estuaries (Hatakeyama *et al*., 2022). Glass eels typically refer to an intermediary stage in the eel's complex life history between the leptocephalus stage and the juvenile (elver) stage. Glass eels are defined as "all developmental stages from completion of leptocephalus metamorphosis until full pigmentation", (Hatakeyama *et al.*, 2022). Once they recruit to coastal areas and river mouths, they migrate up rivers and streams, overcoming various challenges of climbing over natural and man-made obstacles. In freshwater, they acquire pigmentation, often called elvers (young eels), that feed on insects~~,~~ small crustaceans, and other invertebrates (Dörner & Berg, 2016).

The yellow eel growth duration varies according to environmental parameters such as food availability, water quality and temperature, and species (Arai, 2016; Bernotas *et al*., 2020). The yellow eel stage is characterized by golden pigmentation on their ventral surface, after which they mature into silver eels and migrate to spawning locations in the deep sea. During migration, the silver eels do not feed but rely on stored energy (Couillard, 2014). The external features undergo dramatic changes, such as; enlarged eyes, (for optimal vision in the dim -blue -clear -ocean light) and their ventral surfaces turn silvery, to create a counter-shading pattern that shields them from oceanic predators such as sharks (Couillard, 2014).

2.2 Distribution and population characteristics of anguillid eels in the West Indian Ocean

Freshwater eels are divided into temperate, subtropical, and tropical eels based on their distribution with tropical species being more specious (Arai, 2016). The four species that have been confirmed to occur within the West Indian Ocean (WIO) (Froese & Pauly, 2016; Jacoby & Gollock, 2014) have the widest distribution in the world. The tropical eel *Anguilla marmorata* (Quoy & Gaimard, 1824) is one of the largest anguillid eel reaching two metres in length and weight of up to 21 kg (Arai & Chino, 2018). This species also has the widest geographic distribution of the genus *Anguilla* (Tsukamoto *et al*., 2020) occurring from the east coast of Africa to the Marquesas Islands in the South Pacific, as far as southern Japan in the north and as far south as southern Africa (Arai & Abdul, 2017). A*nguilla bicolor* (McClelland, 1844) is widespread in the Indo-western Pacific, being described from streams in the Kimberley regions of northern Western Australia but also in Africa, where it is widespread but relatively uncommon along east African coast, Madagascar, and Mozambique in the lower Zambezi River (Froese & Pauly, 2016). *Anguilla bengalensis* (McClelland, 1844) is common in East African water systems, an occasional vagrant to the Arabian Peninsula, and across the Indian Ocean to mainland India, Pakistan, and Bangladesh. *Anguilla mossambica* (Peters, 1852) is endemic to the South Western Indian Ocean (SWIO) covering the coastlines of several countries and territories, such as Madagascar, Mauritius, Seychelles, Comoros, Réunion (a French overseas department), and the coastal regions of Mozambique, Tanzania, and Kenya (Jacoby & Gollock, 2014). For all the WIO eels, spawning is hypothesized to occur over the Mascarene Plateau (Schabetsberger *et al*., 2016), between 10°S and 20°S, and 60°E and 65°E (Figure 1). All anguillid eels in Kenya occur in the east-flowing Rivers that empty into the Indian Ocean.

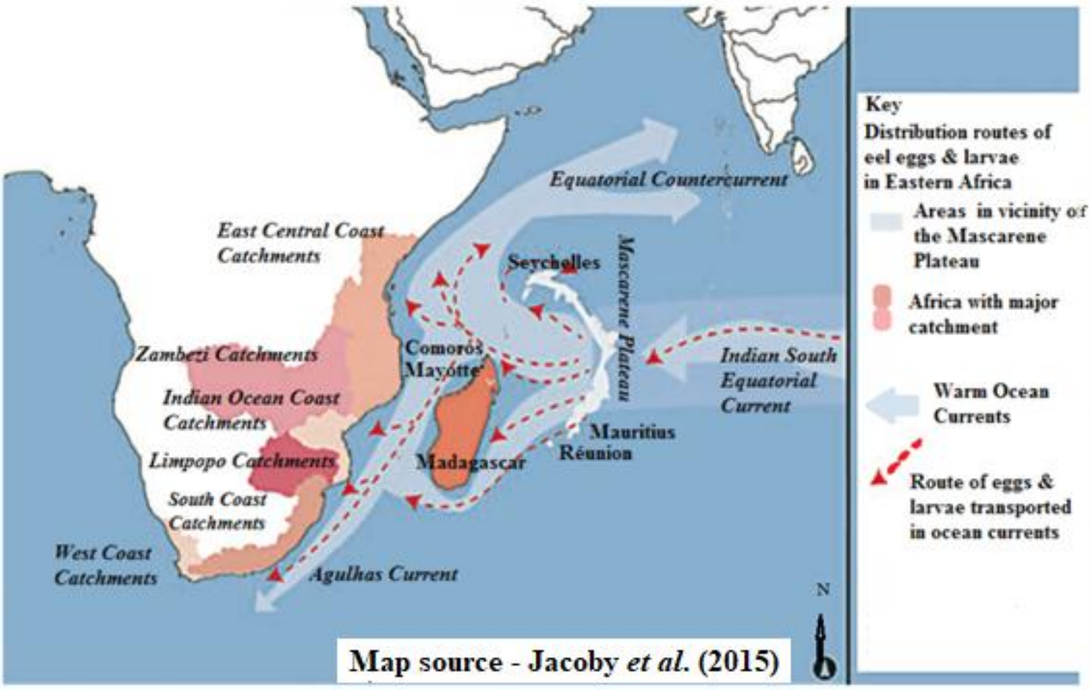


Figure 1: Map Showing the Spawning Area of WIO Tropical Eels

2.3 Recruitment dynamics, migration strategies, and challenges of Glass eels in estuarine and freshwater habitats

Glass eel recruitment patterns are characterized by pulses of high recruitment followed by periods of low recruitment (Shuai *et al*., 2023). These pulses can be driven by a variety of factors, including changes in ocean currents, temperature, and food availability. Recruitment occurs in specific seasons; for example, European eels' recruitment occurs in the spring, while American eels recruit in the fall (Miller *et al*., 2016). Studies in tropical Central and South America, recruitment occurs from February to May, whereas in West Africa it is from March to June, and May to August in Southeast Asia (Aoyama *et al*., 2018). However, the timing of recruitment can change significantly between years and locations.

Recruitment of the temperate *Anguilla anguilla* glass eels has declined by over 5 % level pre-1980; for the Baltic Sea and 10% for the North Sea (ICES, 2012); Recruitment in tributaries of the River Thames, UK indicated declines of over 99 % (Gollock *et al*., 2011). The Bristol Channel and Severn Estuary system are the largest and most outstanding contributors to the UK glass eel fishery recruitment decline of approximately 70 % from their peak in the late 1970s (Bark *et al*., 2007). Other recruitment declines (97 %) have occurred in American eel (*Anguilla rostrata)* since the 1980s (Benchetrit & McCleave, 2016; MacGregor *et al*., 2008; Richkus & Whalen, 2000), Japanese eel (*A. japonica*) (Knights, 2003) over a similar timespan. Doole (2005) and Jellyman (2007) reported declines of up to 75 % in the subtropical New Zealand longfin eel (*Anguilla dieffenbachii*) since commercial fishing commenced in the 1970s. Specific statistics on the decline of recruitment of anguillid eels in the West Indian Ocean (WIO) region are limited. Most research and statistical data on eel recruitment decline focus on temperate species.

There are several anthropogenic impacts potentially influencing the recruitment level and survival of anguillid eels. Larval marine migration and survival are influenced by oceanic and climatic variables that are beyond the control of single countries or states (Als *et al*., 2011; Côté *et al*., 2013; Gagnaire *et al*., 2012; Popova *et al*., 2019). One of the anthropogenic pressures affecting eel stocks in many temperate regions is commercial exploitation (fishing) for the glass eel life stage (Aschonitis *et al*., 2017; Guo *et al*., 2024). As a result of the temperate-wide decline in glass eel recruitment, the value of glass eels has grown a hundred-fold since the 1970s (USD 7.50 per Kg) to 2008 (USD 750 per Kg) (Briand *et al*., 2009). However, the glass eel trade was effectively outlawed in 2009 when European eel listing on the Convention on International Trade in Endangered Species (CITES) was effected (CITES

2007). The majority (87%) of European glass eel fishing occurs in estuaries, where a significant number of 'civelliers' (boats equipped with push nets) apply intense fishing pressure on glass eels as they migrate upstream during the flooding tide (Briand *et al*., 2009).

In Japan, glass eels are a delicacy that is appreciated for their taste and nutritional value. The Japanese eel is listed as "Endangered" on the IUCN Red List, and the Japanese government controls fishing for this species, but illegal fishing and poaching threaten the species' recovery (Kaifu, 2019).

Glass eels enter estuaries depending on latitude and the varying oceanic factors. For example, in Southern Spain, Arribas *et al*. (2012) found out that changes in the density of glass eels were partially influenced by local environmental factors, such as turbidity, rainfall, and temperature in the short term (monthly), whereas long term (inter-annual) changes were associated with primary production at the spawning area and the North Atlantic Oscillation (NAO) index (Arribas *et al*., 2012). Due to their cylindrical shape and small size, glass eels have relatively restricted locomotory capabilities; for instance, *A. rostrata* has a critical swimming speed of 11.7–13.3 cms^{-1}, over a temperature range of 14–24.5 ^{O}C (Wuenschel and Able, 2008). It has also been reported that glass eels of *A. rostrata* and *A. anguilla* cannot swim continuously for long distances against currents beyond 30 cms^{-1} (Adam *et al*., 2008). The power of glass eels to actively migrate upstream, manoeuvring through an estuary system against the river flow is, in this case, limited.

Passive upstream migration using 'Selective Tidal Stream Transport' (STST) is a technique by which fish with low swimming potential move upstream throughout an estuary system using tidal currents (Benson *et al*., 2021; Bice *et al*., 2023; Merk *et al*., 2023; Verhelst *et al*., 2018). Selective Tidal Stream Transport' occurs when a fish remains on or near the bottom channel during receding tides but moves into the water column on flood tides. The flood tide pushes the fish towards freshwater with relatively little energy costs. Using this technique, juvenile fish save up to 90 % of the energy that would otherwise be consumed by active swimming the same distance (Reeve *et al*., 2022). Although STST is the fundamental technique facilitating the migratory passage of glass eels through estuaries, tidal effects tend to be weaker in upper estuarine zones, a trait that makes the glass eels shift to active swimming for further dispersion upstream (Guo *et al*., 2024). At the freshwater interface or definitely from the point where the glass eels accumulate (Harrison *et al*., 2014; Tesch, 2003) they change their behaviour (Podgorniak *et al*., 2016) and actively swim against the current. Such active swimming is displayed by characteristic ‘crawling’ behaviour that glass eels exhibit while scaling surfaces (Harrison *et al*., 2014; Tesch, 2003; Van Wichelen *et al*., 2021).

Water temperature of between 10 and 15 °C is considered a major stimulant to the onset of this active swimming for both *A. anguilla* (Briand, 2009; Cresci, 2022) and *A. rostrata* glass eels (Overton & Rulifson, 2009). In the UK, temperatures between 10 and 11ºC are critical for stimulating elvers to scale weirs or sluice barriers (Monteiro *et al*., 2023). At a salinity of 10 ppt, the transformation from passive to active migration takes about 50 days at 8^{O}C but reduces to about 14 days at 12 °C (Briand, 2009). However, this will be particular to the haline and thermal fluctuations of the estuarine system, which will be determined by estuary length, tidal magnitude, spatial complexity, and freshwater discharge.

Glass eel dispersion is also determined by endocrine control, with thyroid hormones being indicated to invigorate migratory behaviour (Cresci *et al*., 2017). Thyroid hormones play a fundamental role in the modulation of physiological and behavioural adjustments leading to the upstream migration and colonization of freshwater habitats by glass eels (Edeline *et al*., 2009; Liu *et al*., 2019). Internal drivers such as body condition and endocrine control are indicated to play a fundamental role in the tendency of glass eels to remain within estuaries rather than proceed upstream to freshwater habitats (Cooke *et al*., 2022; Edeline *et al*., 2009; Harrison *et al*., 2014). *Ex situ* studies have shown that glass eels with a lower body condition preferred saline over freshwater (Cresci, 2020; Edeline *et al*., 2009) and adaptation is hypothesized to limit mortality due to exhaustion from swimming. Therefore, to settle within an estuary or to continue migration into freshwater is influenced by an individual’s energy content on estuarine arrival (Edeline *et al*., 2009).

2.3.1 Migration patterns and environmental cues influencing glass Eels’ recruitment and behaviour

Glass eels are distributed throughout the water column during the flood tide and passively migrate upstream using the flood tide current (Steendam *et al*., 2020). They move to the margins and swim upstream briefly before the current speed exceeds their sustained swimming potential, during the slack and early ebb tide. They move to the river bed during the full ebb tide and remain on or in the substrate to avoid being swept back by the ebb tide and river flow (Bru *et al*., 2009; Steendam *et al*., 2020; Trancart *et al*., 2013). River discharge has been displayed as an attracting cue that plays a fundamental role in the abundance of glass eels at the estuaries (Arribas *et al*., 2012; Crivelli *et al*., 2008). During high river flows, glass eels assemble close to the banks of the estuary, exposing them to potential predation and exploitation risk as they scale upstream (Adam *et al*., 2008, Bru *et al*., 2009; Chen *et al*., 2023; Zumbrägel, 2024). Besides the rheotactic influence of river flow on upstream migration, olfactory cues related to freshwater are also important in the regulation of glass eel migratory behaviour (Briand *et al*., 2012; Edeline *et al*., 2009; Houde *et al*., 2022; Porceddu *et al*., 2022; Tesch *et al*., 2003). Eels have a well-developed sense of smell, with the olfactory loop and nasal cavity in glass eels being similar to those manifested by adults of other fish species (Naisbett-Jones & Lohmann, 2022).

The smell of freshwater acts as both a behavioural 'cue' (e.g., prompting intense swimming activity during STST) and a directional 'clue' (e.g., showing the direction of inland waters) (Edeline *et al*., 2009). Earthy-odorous substances, such as geosmin, related to inland waters are important as attractants towards freshwater (Houde *et al*., 2022). Moreover, glass eels are also stimulated by odour associated with the presence of adult eels, which prompts orientation toward suitable habitats (Edeline *et al*., 2009; Huntingford *et al*., 2012).

2.4 Environmental and ecological factors influencing anguillid eels' distribution, habitat preference, and feeding dynamics

The riverine distribution of eels is affected by environmental factors in microhabitats and biotopes such as depth, velocity, sediment, aquatic vegetation, and riverbank conditions, among other factors (Kume *et al*., 2019; Laffaille *et al*., 2003). Nguyen (2018) and Hsu (2019) reported that *A. marmorata* tends to live in freshwater areas rather than in brackish and marine habitats. However, other studies show the species inhabit a broad range of habitats, ranging from brackish estuaries to upland headwaters (Arai & Chino, 2018; Kumai *et al*., 2020; Wakiya *et al*., 2019). Arai et al. (2020) found that *A. bicolor* was widely distributed along rivers in the northwestern part of Peninsular Malaysia, with higher abundance in the downstream and midstream reaches, and only occasional occurrences in upstream areas. *A. bengalensis*, however, was found to prefer mainly upstream reaches and rarely occurred in midstream areas. Other studies indicated that the African mottled eel (*A. bengalensis)* mainly inhabited areas far inland by surmounting formidable barriers in their upstream migration (Bell-cross & Minshull, 1988; Okeyo, 1998). *A. mossambica*, a species of plain skin coloration, is an inter-habitat migrant, moving between freshwater and seawater habitats (Lin *et al*., 2013). In other studies, *A. mossambica* was found to inhabit coastal and estuarine areas without migrating upstream into inland freshwater reaches (Lin *et al*., 2016).

In the river system, the substrates of the upper reaches are mainly bedrock, boulders, cobbles, and gravel, quite different from the substrates of the downstream, which are sandier and muddy because of eroded materials carried from the upper reaches (Chilton & Spotila, 2022). Itakura and Wakiya (2020) studied the preference of biotopes by eels and reported that the density of eels was consistently lower among riverbanks consisting of concrete and sand, while it was highest in vegetation-infested riverbanks and the main channel. Densities increased in mud and boulder substrates, and a combination of habitat factors. Kumai *et al*. (2021) studied the habitat preference of *A. marmorata*. They found the species preferred habitats with larger substrate materials, fewer fallen leaves and less leaf detritus, higher current velocity, and lower turbidity than the temperate species *A. japonica*. Both species co-occur in the same river systems, although the distributions of the two anguillid eels were segregated. *A. marmorata* preferred clear-flowing mainstreams and tributaries, whereas *A. japonica* preferred stagnant muddy estuaries, backwater areas, irrigation channels, and reservoirs. Hsu *et al*. (2020) divided tropical and subtropical eels into two broad groups according to skin coloration (marbled or plain) to study biotope preferences. The study found out that, in tropical/subtropical areas, marbled and plain eels coexisted, but marbled eels tended to inhabit the middle to upper sections of the rivers while plain eels preferred the middle reaches to estuaries. Therefore, they concluded that the plain eels' countershading may have helped them to remain hidden in the mud and sand substrates at the lower reaches of the Rivers, providing a refuge from being detected by predators/prey. On the contrary marbled eels can blend in with substrates such as cobble, gravel, and fallen leaves in the upper reaches (Chilton & Spotila, 2022).

The feeding ecology of a species is greatly linked to its population dynamics and contributes to the knowledge of resource partitioning, prey selection, predation, habitat preferences, competition, evolution and energy transfer within and between ecosystems (Braga *et al*., 2012). Such ecological knowledge is of significant value when developing conservation plans and therefore, it is a fundamental element in the protection of species and ecosystems (Braga *et al*., 2012). Anguillids are trophically valuable because adult eels often represent large piscivorous fish species in aquatic systems, feeding on crabs, frogs, shrimps and fish (Wakiya & Mochioka, 2021). Hartanto *et al*. (2016) based on stomach content analysis of *A. bicolor* and *A. marmorata*, identified three dominant types of foods, namely; crabs, shrimps, and earthworms. The study showed that crabs were predominantly based on temporal and spatial studies. However, statistical analysis showed no significant differences in food variation during the three months of sampling. Other studies by Rupasinghe and Attygalle (2006) on *A. bicolor* in the Bolgoda estuary, Bandaragama, Sri Lanka showed ontogenetic dietary shifts in eels where smaller individuals primarily fed on invertebrates, whereas larger individuals consumed both invertebrates and additional prey such as fish and worms.

2.5 Spawning locations and oceanic dynamics of anguillid eels in relation to recruitment declines

Eel spawning locations are difficult to identify because monitoring actual spawning has been difficult and expensive as it occurs in the first 1000m in the open ocean, making surveys hard to carry out (Miller *et al*., 2015). Consequently, the location and sizes of leptocephali are frequently used to identify possible breeding sites (Kimura & Tsukamoto, 2006; Miller *et al*., 2022; Quatrain *et al*., 2019). Additionally, silver migrating eels have rarely been caught at the spawning location but a few incidental catches of silver eels have been obtained along continental margins (Chaput *et al*., 2014; Haro, 2014). Therefore, it has been deduced that the presence of the youngest eel larvae suggested the probable spawning areas (Friedland *et al*., 2007; Leone *et al*., 2016; Schabetsberger *et al*., 2016; Tsukamoto, 2006).

Temperate American (*A. rostrata*), European (*A. anguilla*), and Japanese (*A. japonica*) silver eels make long migrations (approximately 5,000 km) to spawn in the subtropical gyres of the North or North Atlantic Pacific oceans (Miller & Tsukamoto, 2016; Schabetsberger *et al*., 2021). In the case of *Anguilla anguilla* and *A. rostrata* the surface temperature of the 22.5°C found near the frontal zone in the Sargasso Sea, forms the northern zone of the spawning area (Hsiung *et al*., 2022). The temperature fronts (22.5°C) in the Sargasso Sea act as cues that aid adult eels to detect the spawning area (Friedland *et al*., 2007). The Sargasso Sea has a relatively stable temperature range, typically between 22-28°C (72-82°F), high salinity range, between 36-37 parts per thousand (ppt), relatively low light levels due to its depth (>200m), complex current systems, (including the Gulf Stream, the North Atlantic Current, and the North Equatorial Current), which transport Anguilla eel larvae from the breeding sites towards continental shelf. The presence of Sargassum, a floating brown seaweed provides shelter, food, and protection for *Anguilla* eel larvae during their long oceanic journey (Godínez-Ortega *et al*., 2021). A rapid gush of water affiliated with these fronts drives a variety of leptocephali species eastwards (Kuroki *et al*., 2014). Therefore, variation in the latitude or strength of these fronts (and the affiliated rapid water gushes) affect the spawning locality and the subsequent transport (south of the Mariana Islands) of the leptocephali to continental habitats, and partially explain the decline in eel recruitment documented since the early 1980s (Correia *et al*., 2018; Westerberg *et al*., 2018). The spawning location of the Japanese eel corresponds to a salinity front (Leone *et al*., 2016; Miller & Tsukamoto, 2016) as in Sargasso Sea. The decrease in salinity in the frontal zone acts as a cue that triggers the cessation of migration and the commencement of spawning. Changes in the location of the salinity front impact on successful spawning, thus a decline in eel recruitment as observed in the recent decades (Correia *et al*., 2018; Westerberg *et al*., 2018).

The earliest attempt at locating WIO eel spawning was made by Jespersen (1942), after observing small leptocephali distribution in a region to the East of Madagascar, and suggested that the probable spawning site is at the Mascarene plateau. As established in the temperate Atlantic and Pacific Oceans, the eel spawning area is likely to be marked by thermal and/or salinity fronts. In the tropical SWIO, there is no clear thermal front in the upper ocean; however, the South Eastern Current (SEC) acts as a barrier, separating water masses of southern and northern origin with different salinities (Aubone *et al*., 2021; Fiedler & Lavín, 2017; New *et al*., 2007).

2.6 Anthropogenic impacts on freshwater ecosystems and the vulnerability of anguillid eels in Kenyan Rivers

As the human population increases, the destruction of natural habitats to satisfy the demand for goods and services has increased dramatically, especially in developing countries, precipitating environmental disturbances that strongly affect freshwater ecosystems (Ligeiro *et al*., 2013). The peripheries of watercourses have historically been preferred for human settlements (Plieninger *et al*., 2018) but also for fisheries, agriculture, power generation, and industry. Modern industrial development and urbanization have concentrated human beings in densely populated cities located along rivers and waterways. As a result, there is increased utilization of hydrological resources culminating in water quality and quantity modification, habitat loss, and fragmentation (Collier *et al.,* 2016), leading to habitat degradation and biodiversity loss (Lucy *et al.,* 2016). Fish community structure is a key component of freshwater ecosystems, and species community composition is susceptible to the effects of human disturbance on lotic systems (Kalogianni *et al.,* 2017).

Habitat modification, such as instream dredging, bank stabilization, dams, and hydropower construction, directly impact community composition and cause the progressive dominance of alien species capable of enduring degraded environments (Boston *et al.,* 2016).

Habitat fragmentation and reduced connectivity due to an increase in hydroelectric power plants are remarkable drivers of diversity loss and altered species community composition (Gracey & Verones, 2016). For example, species of salmon and trout have been significantly impacted by habitat fragmentation and modification caused by dams and other human activities in the Pacific Northwest of the United States. Dams disconnect migration routes, disrupt spawning grounds, and fragment populations, leading to decreased genetic diversity and resilience to environmental changes (Tamario *et al*., 2019). The current study seeks to examine the ecological and biological traits of anguillids in Kenya and the risks associated with their existence in Rivers AGS and Ramisi.

2.6.1 Habitat loss, fragmentation, and impact on eel bio-ecology

Rivers provide multiple goods and services to society (Elliott & Whitfield, 2011). This has led to river channelization, hydro-morphological modifications (dams and weirs), drying out of the lateral wetland, wetland drainage, water extraction, and modification of land use in the floodplain that leads to higher erosion and sedimentation (Basset *et al.,* 2013; Elliott & Hemingway, 2002; Postel & Richter, 2003). These human activities lead to fragmentation and loss of eel habitats, through physical destruction of eels' hide-outs and blockage of their waterways. For example, in Europe, 50–90% of habitats were lost by the end of the twentieth century (Feunteun, 2002), while approximately 75% were lost in Japan, Korea, Taiwan, and China between 1970 and 2010 (Chen *et al.,* 2014). The subsequent results are visible impacts on the ecology of eels with impacts on their phenology (Chevillot *et al.,* 2017; Menzel *et al.,* 2006) and modification of distribution patterns (Cheung *et al.,* 2010). Itakura *et al*. (2015), reported that the increased river modifications and connectivity reduced the quantity and quality of appropriate habitat for not only eels but also a range of invertebrates and other fishes. Reduction in population and diversity of the invertebrates, many of which could provide food for the eels affected eel growth and abundance (Itakura *et al*., 2015). The rate of growth of eels in disturbed river systems was slower by 10% than in an undisturbed environment (Itakura *et al.,* 2015). Similarly, yellow eels' populations decreased with a decrease in the diversity of benthic invertebrates in areas with River modifications than in areas with an unmodified river system (Kanazawa & Miyake, 2006). A study by Onikura *et al*. (2007) reported lower eel abundances in creeks with river modifications than in comparable creeks with no modifications.

The construction of a dam on a river causes many problems for down and upstream migrating eels. These include: the delay, or even the total prevention of migration in still water zones of the impoundments; damage to eels when passing over spillways or through turbines; mortalities as a result of predation by birds either in the impoundment or else at the outlet of turbines; mortalities due to changes in water quality (deficit in oxygen content in the impoundment and super saturation of the atmospheric gases downstream of turbines or spillways), (Drouineau *et al.,* 2015); mortalities due to mechanical injuries by turbines ranged from 15% to 100% depending on the type of turbine (Heisey *et al*., 2019). Obstruction of eels migrating upstream leads to a higher concentration of eels downstream of obstacles. This increases intraspecific competition for food and space, resulting in lower survival and growth weight (Righton *et al.,* 2021). Aggregation of eel leads to increased susceptibility to predation (Drouineau *et al*., 2015), overfishing (Briand *et al.,* 2012; Dekker *et al*., 2003), and possible modification to the sex ratio, as sex determination is density-dependent (Davey and Jellyman, 2005). Obstacles to upstream migration act as a permanent selection pressure for the fittest to survive by modifying demographic traits, which act as selection pressure (Podgorniak *et al.,* 2016). Côté *et al*. (2014) demonstrated that by impairing migration within catchments, obstacles not only decreased the fitness of eels to reach suitable habitats but also led the eels to suffer damage during their migration (Mateo *et al.,* 2017).

2.6.2 Impacts of pollution on anguillid eels

Ecologically, eels are important top predators in freshwater rivers, that govern ecosystem health by regulating the population of other smaller aquatic organisms like smaller fish, molluscs, crustaceans, insects, worms, carrion, and eggs of other fishes (Denis *et al*., 2022). The *Anguilla* eels are useful bio indicators of stressful environments due to; their worldwide distribution and abundance, size, and long life (Aarestrup *et al*., 2009). They migrate extensively; collecting stressor signatures that can be correlated with the vast areas they cover (Patey *et al*., 2017). In polluted environments, smaller organisms consist of small amounts of pollutant compounds, and when fed upon by eels they accumulate in eel tissues over time.

Yellow eels stay in fresh water for between 10 to 25 years (Jellyman, 2022), during which they continuously accumulate xenobiotic compounds through gills, skin, and contaminated food (Belpaire *et al*., 2016). During the yellow stage, they accumulate high levels of fat reaching up to 27–29% of body weight at the onset of migration at the silver eel stage

(Van Ginneken *et al*., 2009), accumulating lipophilic compounds such as polychlorinated biphenyls (PCBs) from industrial waste and organochlorine pesticides (OCPs). They also become vulnerable to heavy metal pollutants such as Manganese (Mn), Zinc (Zn), Lead (Pb), Cadmium (Cd), Nickel (Ni), Cromium (Cr), Mercury (Pb) and Arsenic (As) from industrial and agricultural processes (Le *et al*., 2009). These heavy metals accumulate in tissues of eels in higher concentrations than in water because of biomagnification up the food chain.

In circumstances where high doses of these metals accumulate in the eel, physiological damages to organs occur, leading to impacts on individual growth rates, reproduction, and mortalities (Rajeshkumar & Li, 2018). The situation is worsened synergistically by parasitic infections leading to immunosuppression of the eels (Sures & Knopf, 2004**),** which paves the way for colonization by opportunistic bacteria and fungi (Rajeshkumar & Li, 2018) and subsequent mortalities. Eel diseases, which have been reported to be associated with pollution of the aquatic environment, include epidermal papilloma (Korkea-Aho, 2006**),** fin and tail rot (Kar, 2015), gill disease/hyperplasia (Kirk & Lewis, 1993), liver disease (Kar, 2015), neoplasia (Lombardini *et al*., 2014), parasitic infections, skin disease/ulceration (Tamam, 2014) and viral diseases (Van Beurden, 2012). The trigger for these diseases has been blamed on contaminated habitats (Landsberg, 2005). Of these, fin/tail rot, gill disease/ hyperplasia, and ulceration are linked to bacterial involvement (Kar, 2015).

Organic waste causes eutrophication, which often comes along with oxygen depletion, leading to hypoxia (Teichert *et al*., 2016) and thus catastrophic events in eels, as intolerance to low oxygen concentration is a widespread species attribute. A study by Galib *et al*. (2018) analysed changes in fish abundance, species richness, and water quality parameters over time and space due to municipal wastewater disposed of in the Barnoi River in Bangladesh. The study found that fish abundance (measured as CPUE) and fish species richness were reduced by 40 and 50%, respectively, at the effluent-receiving downstream sites. Oxygen-demanding organic municipal wastewater has the potential to cause changes in biodiversity in freshwaters (Dudgeon, 2006).

2.6.3 Invasive alien species and their impact on anguillid eels

Invasive alien species (IAS) have resulted in major impacts on biodiversity at a global scale, where at least 39% of the species extinctions during the past 400 years have been attributed to Invasive Alien Species (Rattan *et al*., 2003). Human disturbance intentionally or unintentionally introduces alien species into the environment (Halpern *et al*., 2008). These invasions are considered to be permanent forms of pollution and negatively affect stressed ecosystems susceptible to invasions (Kiruba-Sankar, 2018). Hundreds of freshwater species have been moved outside of their native ranges by vectors such as ballast water, canals, deliberate introductions, and releases from aquaria, gardens, and bait buckets (Shalovenkov, 2019). As a result, many freshwater systems now harbour dozens of alien species. In turn, alien species can strongly alter the hydrology, biogeochemical cycling, and biotic composition of invaded ecosystems, and thus modulate the effects of other stressors (Kiruba-Sankar, 2018). This happens when the introduced species compete with native fishes for food, space, and survival, and also act as carriers for accompanying microscopic organisms, including pathogens and parasites. As a result, native species are confronted with new environmental conditions they had not encountered previously, further increasing stress and the need to adapt to the new circumstances. Deliberate or accidental introduction of fish species is considered as most insidious threat to fish conservation around the world.

The eel trade has seen live glass eels being moved from one continent to another for aquaculture purposes. The introduction of European eel into Japan to enhance stocks for trade has the potential risks of competition for food and of the eventual possibility of, interbreeding between European and Asian eel species (Yuan *et al*., 2022). These introduced species move into the vicinity of the spawning grounds of the Japanese Eel, which could result in interspecific hybridizations and the collapse of the species (Pujolar & Maes, 2016).

Anguillicola crassus is the most documented and has a wider impact on eels than all the other alien species (Becerra-Jurado *et al*., 2014). It is a natural parasite of Japanese eel, which was introduced into Europe in the mid-1970s through the aquaculture trade. This parasite causes inflammation of the swim bladder, leading to multiple bacterial infections, stress, and loss of appetite (Kirk, 2003; Lefebvre *et al*., 2013). A study by Pratama *et al*. (2019), on parasites of the swim-bladder of tropical *A. bicolor* found two nematodes, *Anguillicola* and *Spirocamallanus* with prevalence rate and intensity of 3.3% - 6.7%, 0.03 – 0.06 and 13.3%, 0.13 respectively. Molecular identification of the nematodes demonstrated that they were closely related to *Anguillicola crassus* (95.40%) and *Spirocamallanus philppinensis* (97.93%). In Kenya, Seegers *et al*. (2003) listed the following fishes as alien species in the Athi-Galana-Sabaki River; *Oncorhynchus mykiss* (Rainbow trout), *Salmo trutta* (Brown trout), *Poecilia latipinna* (Sailfin molly), *Oreochromis mossambicus* (Mozambique Tilapia), *Ctenopharyngodon idella* (Grass carp), *Gambusia holbrooki* (Eastern mosquito fish) and other fishes of the family Cichlidae. They were introduced at different times and for different purposes, for example; *Ctenopharyngodon idella* was introduced in 1969 from Japan into Kenya for aquaculture and weed control; *Gambusia holbrooki* was introduced for mosquito control in Kenya. Tembo *et al*. (2023) reported that introduced *Clarias gariepinus* in AGS and Ramisi competed for food with *Anguilla bengalensis*, *Anguilla bicolor* and *Anguilla mossambica*.

2.6.4 Over-harvesting of anguillid eels

Glass eel fisheries take place in estuaries and at the mouths of Rivers and dams, where the natural concentration of glass eels (as they enter and move upstream) can more easily be exploited (Ringuet *et al*., 2002). Hand-held or ship-based nets are used, which are moved manually or fixed and include trawls, stow nets, and fyke nets (Yeh, 2002). The fishery for yellow and silver eels involves the use of cheap and fairly simple gear - baited traps, fyke nets, baited long lines, spears, or shore seines. Of these, the first three methods are the most commercially viable, and the choice of a method is governed by what is permitted in a country. The volume of global live eel exports (all species and all biological stages) reached 25,794 tonnes in 1997 (compared to about 5000 tonnes before 1983) and was valued at USD385 million (an average price of USD15/kg) (Anon., 2000). Spain is the largest consumer of eels in Europe, importing 20 tonnes of glass eels from Asia in 1997 for domestic consumption and they were purchased by consumers at an average price of USD125/kg (Frost, 2001).

According to FAO (2015), eel farming is responsible for 90% of total eel production compared to wild-caught eels. Glass eels are caught in the wild because eels do not reproduce in captivity. Japan consumes 70% of total freshwater eel production (Shiraishi and Crook, 2015). China has supplied two-thirds of the world's production of eels - 130,000 tonnes each year since 1998 (FIS, 2000), worth over USD1.3 billion. Eel farming can be quite lucrative; For example, on one hectare of farmland, the net income from eel production reaches 1.2 to 1.35 million Yuan (USD145 000 to USD163 000) per harvest in China (Guangwei & Shishan, 1999). In Taiwan, the eel is the most important farmed fish, with an annual production that fluctuates between 26 000 and 56 000 tonnes, and worth more than USD400 million (Yeh 2002). This high demand for eels in Eastern Asia food markets has led to the development of highly competitive aquaculture farms (Lee *et al*., 2003). These farms depend on wild-caught glass eels which increased demand for glass eels, completely transforming the global industry. The shortage of Japanese glass eels since the early 1970s enhanced the import of European and American glass eels (Haro *et al*., 2000; Lee *et al*., 2003; Ringuet *et al*., 2002) into Japan. Also, fishing efforts increased in Europe and Asia (Briand *et al*., 2009) leading to the development of a large fishery targeting glass eels from North America (Meister & Flagg, 1997).

In contrast, commercial eel exploitation from Africa is more recent. Since first record of the tropical eel trade from Africa started in 1998 with regular exports from Madagascar and intermittent live exports from Mozambique, Tanzania, and South Africa (Hoyle *et al*., 2016). Additionally, 387 tonnes of wild-caught glass eels and 10 tonnes of aquaculture were recorded in Madagascar, all of which were exported (Hoyle *et al*., 2016). In Kenya, *Anguilla* eels are caught for domestic consumption by riverine communities.

Conservation measures have been employed in Europe for eel exports. Exports have been restricted after European eel inclusion in Appendix II of the Convention on Trade of Endangered Species (CITES) in 2009 and a ban on all imports and exports from and to the European Union implemented in 2010 (Nijman, 2015). In Japan, China, and Taiwan, glass eel fisheries are forbidden, and a special license is required to capture seed for aquaculture and research. Also, the catch of silver eels has decreased throughout the world by as much as 95% in the last 25 years (Cairns *et al*., 2014; Tatsukawa, 2003; Tsukamoto, 2009) because of a reduction in the abundance of the stock. Silver eel fisheries have, for example, completely disappeared in Taiwan (Tzeng, 2016) and are restricted in 11 prefectures in Japan (Jacoby & Gollock, 2014). In Europe, the decline in the silver eel has preceded the decline in the recruitment of glass eels from the ocean to freshwater ecosystems (Dekker *et al*., 2003).

There are five principles of conservation measures in place for glass eel and elver fisheries practiced in Europe and Asia (Ringuet *et al*., 2002); a ban on commercial fishing, a requirement for elver passes against barriers, fishing gear type regulations, closed fishing seasons and licenses for eel fishing. Seven conservation measures have been drawn up for yellow and silver eel fisheries in Europe: gear controls in all countries (controls on net mesh sizes, mesh size differ in different countries), closed fishing, and licenses for fishing/dealing in eels and eel products which limits on the size of eels caught. Free gaps in weirs have been introduced to allow downstream migration, quota fishers allocations have been established and a complete banning of eel fishing for some time has been done, as it happened in Portugal, during the 2001-2002 season (Antunes, 2008). Riverine communities do the exploitation of Kenyan eels for food and subsistence, however, there are no policies in place for eel fishery management and conservation (Kihia *et al*., 2022).

2.6.5 Impacts of climate change on anguillid eels

There are correlations between glass eel recruitment and different oceanic indicators, such as sea surface temperatures (Drouineau *et al*., 2017) Three main impacts of global warming on eels are: a limitation in trophic conditions (change in primary production) (Bonhommeau *et al*., 2008; Bonhommeau *et al*., 2009;), changes in oceanic currents modifying larval transport (Friedland *et al*., 2007; Zenimoto *et al*., 2009), and/or spatial oscillations of a salinity front used by adult eels to detect the spawning grounds which then lead to oscillations in the success of larval transport (Kimura & Tsukamoto, 2006). Bonhommeau *et al*. (2008) while analysing the decline of three *Anguilla* species, concluded that due to climate, an increase in sea surface temperature led to a higher stratification of the eel spawning grounds and consequently led to a lower primary production, which resulted in lower food availability for the leptocephali. Climate change interferes with the hydrological cycle, which indirectly influences river discharge (Milly *et al*., 2005) through modifications of precipitation regimes. Rainfall and river discharge are important triggers (indirect and direct, respectively) of silver eel migration (Drouineau *et al*., 2018; Trancart *et al*., 2013). Higher river discharge increases migration speed (Stein *et al*., 2016), while reduced discharge delays migration. Recruitment and escapement delay until environmental conditions are favourable (Drouineau *et al*., 2017). Moreover, reduced discharge leads to higher numbers of eels going through turbines, since a higher proportion of water is guided through the turbines during low flows, leading to physical injuries and higher mortalities (Jansen *et al*., 2007).

2.7 The decline of anguillid eels: Human-induced threats and conservation concerns for the Kenyan anguillid eel species

Like many other fish species, anguillid eel populations have seen progressive declines in recent decades because of several factors, including overexploitation, habitat modification and fragmentation, climate change, the introduction of alien species, and pollution (Cairns *et al*., 2022; Miller & Casselman, 2014; Simeoni *et al*., 2023; Tsukamoto *et al*., 2009). Human-induced multiple threats have been linked to an upsurge in the listing of eels in the IUCN Red List of species. For instance, *Anguilla anguilla* (European eel) is listed as "critically endangered" (Jacoby & Gollock, 2014), while American and Japanese eels have been classified as "endangered" (Jacoby & Gollock, 2014; Jacoby *et al.,* 2015; Pike *et al.,* 2020). Consequently, due to declining eel stocks, in 2010, the European Union (EU) banned all imports and exports of *A. anguilla* to and from the EU member states as a measure to curb further decline and spur the conservation of the dwindling species. In this regard, diminishing global supplies of temperate and subtropical eels have shifted the exploitation focus to the less studied tropical eels. This is exemplified by the recent upsurge in export of *A. mossambica* glass eels from Mozambique for stocking ponds in E. Asia (Fan *et al.*, 2008). Although there is less data available for tropical *Anguilla* spp., conservation concerns exist for many of these species, including *Anguilla bicolor, Anguilla mossambica,* and *Anguilla bengalensis,* which are listed as "near threatened" by IUCN (Pike *et al.,* 2020). This study will come up with scientific information about the risk status of Kenyan eels that will inform conservationists to draw informed conclusions on the eels under study.

2.8 Assessing ecological integrity and risks to anguillid eels' populations in the face of anthropogenic threats

Reduction in the quantity and quality of freshwater, estuarine, and coastal habitats compromises ecological integrity, and the resultant loss of biodiversity has become a cause for global concern (Davidson, 2014). Ecological integrity has been defined as "the ability to support and maintain a balanced, integrated adaptive assemblage of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region" (Karr & Dudley 1981). The intact integrity of ecosystems should exhibit all the potential niche diversity associated with a given complexity operating at appropriate spatial and temporal scales (Karr & Dudley, 1981). Ecological features, either biotic or abiotic components, have been identified as the major factors in the distribution and abundance of fishes (Arunachalam, 2000). Fish species in an assemblage are stratified based on the suitability of the various macro-habitat and micro-habitat features and also of diet resources. Moreover, loss of or changes in fish assemblages may quantify habitat degradation that has occurred in the ecosystem. Generally, ecological integrity declines with increasing anthropogenic modifications, impacting negatively on diversity and population structure of assemblages (Guo *et al*., 2017).

Diadromous fishes such as anguillids have broad distributions and are exposed to multiple threats at different stages of development (Dudgeon *et al*., 2006). There is a growing global concern for their population abundance and escapement trends because of ecological pressures that include habitat loss/modification, migration barriers, pollution, exploitation, parasitism and oceanic conditions that have synergistic and regionally variable impacts (Lotze *et al*., 2006). Landis and Wiegers (1997), and Wiegers *et al*. (1998) adopted regional risk assessment using a relative risk model (RRM) that incorporates multiple stressors, historical events (literature justification), spatial structure (regions/ landscapes), and multiple end points (impacts). Ecological risk assessment calculates the probability of an impact on a specified set of endpoints that exhibit a spatial and temporal distribution (Landis & Wiegers, 2005). The impacts considered are mortality, chronic physiological impacts, and reproductive effects. Relative risk assessment using RRM identifies the sources of stressors, ranks their importance, and combines this information to predict relative levels of risks. The Relative Risk Model (RRM) is a semi-quantitative approach that uses numerical ranks and weighting factors to evaluate and compare multiple ecological risks, especially when risks originate from different sources or types (Landis & Wiegers, 2005).

Despite significant global research on the distribution, life cycle, and ecological functions of anguillid eels, critical knowledge gaps persist in Kenyan river systems, particularly within the Athi-Galana-Sabaki and Ramisi rivers. Existing studies have largely focused on temperate regions, with limited attention to tropical freshwater ecosystems, where environmental dynamics and species interactions differ considerably. Key knowledge gaps include the lack of detailed analyses of population structure, such as size distribution, age classes, and species diversity in these rivers. Furthermore, there is insufficient information on the ecological niches of anguillids, including habitat preferences, trophic interactions, and their role within riverine ecosystems. Conservation studies are lacking, with minimal evaluation of region-specific threats such as habitat degradation, migration barriers, and anthropogenic pressures.

Addressing these gaps is critical for developing tailored conservation strategies to mitigate threats and ensure the sustainable management of anguillid populations in Kenyan rivers. Future research should prioritize long-term monitoring of anguillid populations to understand temporal trends in population dynamics. Studies should also focus on the genetic diversity and connectivity of populations to uncover migration patterns and recruitment processes. Additionally, experimental research on the impacts of specific anthropogenic activities, such as water extraction and riparian degradation, is necessary to quantify their effects on anguillid habitats. Collaboration between local communities, policymakers, and conservation organizations will be essential to implement and evaluate adaptive management strategies that support the conservation of anguillid eels and their ecosystems.

2.9 Conceptual Framework

Independent variables are considered the multiple anthropogenic activities (agricultural, urbanization, industrialization, alien species, loss and fragmentation of habitats, climate change) that produce the stressors (Intervening variables) that impact the health (dependent variable) of the anguillids or pose risks (dependent variable) to the life stages of the eels.

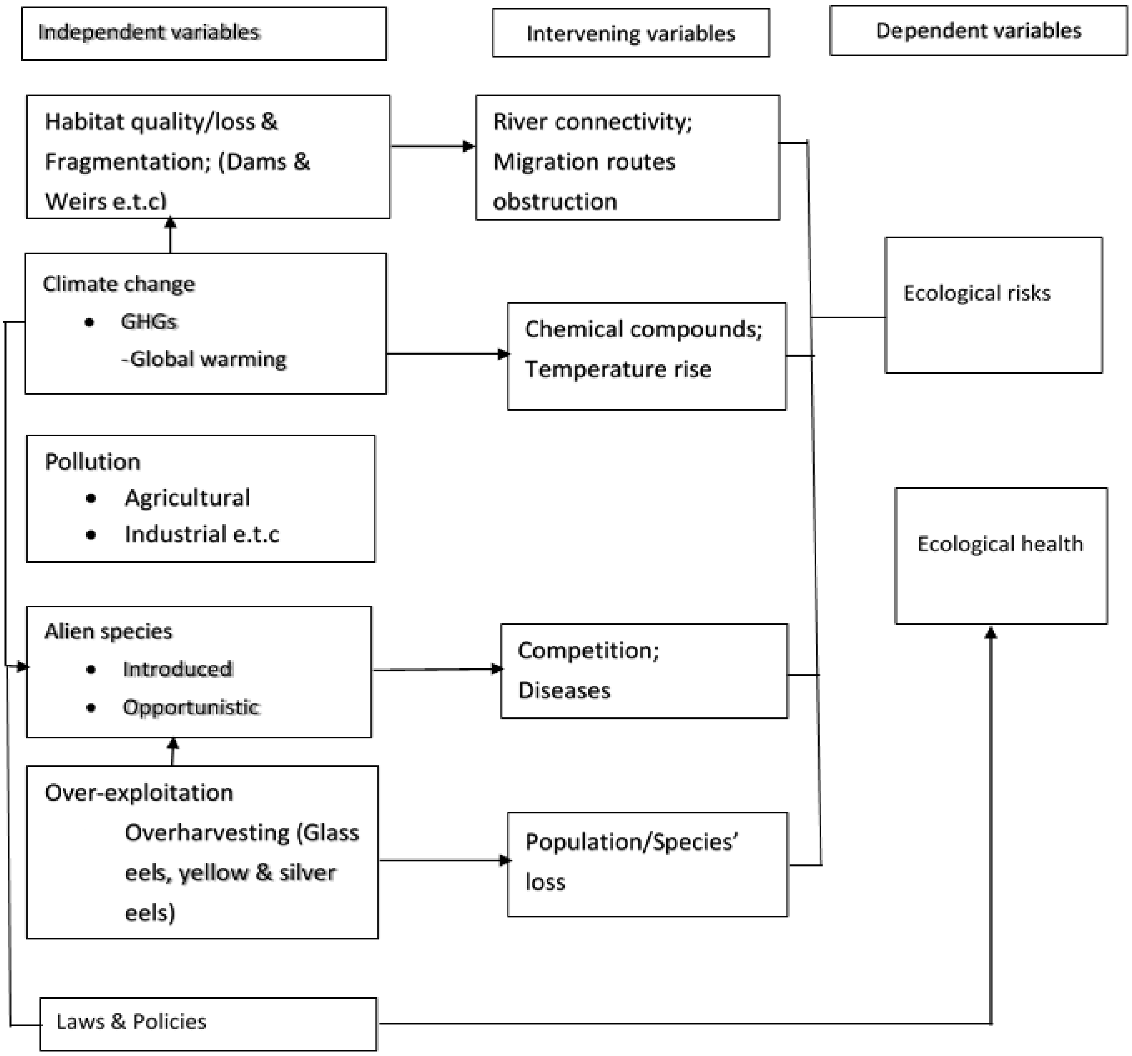


Figure 2: Conceptual Framework Showing Relationships Between Variables

CHAPTER THREE

POPULATION STRUCTURE OF ANGUILLID ASSEMBLAGES AT RIVERS ATHI-GALANA-SABAKI AND RAMISI

Abstract

Understanding the population dynamics of anguillid eels is crucial for effective conservation and sustainable management. However, this knowledge gap persists, particularly concerning anguillid eels in Kenya, emphasizing the importance of this study. From May 2021 to June 2022, 393 anguillid eels were captured, and their biological measurements were recorded on-site. The collected data underwent thorough analysis, employing both inferential and descriptive statistical methods. Results unveiled significant disparities between anguillid eel populations in the River Athi-Galana-Sabaki (AGS) and the River Ramisi. Notably, longer eels were predominantly found in River AGS ($t = 4.8$; $P < 0.01$), while upper River Ramisi exhibited higher biomass compared to upper River AGS ($F=3$, $p=0.04$). Specifically, *Anguilla bengalensis* showed the highest biomass in Upper River Ramisi. Observable differences in relative densities were noted between the upper and lower reaches of the rivers. By categorizing eels into seven size classes based on length (cm), the study identified significant variations in species abundance between the rivers ($p<0.05$), with the highest relative abundance occurring in the 50-60 cm size class (36%). Moreover, the study revealed low diversity in both rivers, with River Ramisi displaying dominance at 0.82 ± 0.06 and River AGS at 0.85 ± 0.1. Overall, the study confirmed the presence of four eel species in Kenya with varying size class abundances, biomass, and low diversity. By elucidating the population structure of anguillid eels, this study provides crucial insights to inform targeted conservation strategies, ultimately contributing to protecting vulnerable populations.

3.1 Introduction

Fish communities often exhibit distinct and non-random population structures (Plass-Johnson *et al*., 2016), influenced by a mix of abiotic (such as temperature, water depth, currents, bottom substrates, and oxygen levels) and biotic factors (including alien species and predators) (Erős, 2017). Species distribution and abundance are determined by their tolerance to environmental conditions and interactions with other organisms (Gebrekiros, 2016). Larger water bodies offer more diverse habitats, supporting a wider range of species and age classes (Gebrekiros, 2016), while moderating biotic interactions like competition. This habitat diversity significantly shapes the composition and structure of stream fish communities (Tóth *et al*., 2019).

In the West Indian Ocean, including the Kenyan coast, four anguillid species thrive: *Anguilla bengalensis*, *A. bicolor*, *A. marmorata*, and *Anguilla mossambica* (Jacoby *et al*., 2015). Variables like distance from the sea, river depth, and flow velocity influence these anguillids' population structure and spatial distribution during their growth stages (Haro, 2014). Physiological traits like euryhalinity and stress tolerance enable them to adapt to fluctuating environmental conditions (Kültz, 2015), making them a significant component of fish communities.

Anguillid eels show a size increase with latitude and distance from spawning areas (Durif *et al*., 2022), indicating their ability to tolerate fluctuating habitats. Understanding the fish-size continuum aids in determining trophic levels' production status and predicting human activities' impacts on ecosystems (Guo *et al*., 2017). However, the inability to breed anguillid eels in captivity has led to overharvesting wild juveniles for aquaculture and trade, resulting in a global decline (Aoyama *et al*., 2012). This decline significantly affects juvenile recruitment into freshwater ecosystems, thus impacting population structure consistency.

While temperate eels have been extensively studied and listed on the IUCN Red List, tropical eels remain relatively overlooked (Jacoby *et al*., 2015; Nijman & Stein, 2022). Given the demand for eels, proper management and conservation policies are essential to prevent tropical eels from facing the same threats as their temperate counterparts (Arai 2014; Jacoby *et al*., 2015). Cooperation between countries is crucial for global conservation efforts due to the catadromy nature of anguillids (Jacoby *et al*., 2015).

In Kenya, anguillid eels are harvested by artisanal fishers for food, but they receive little attention in conservation and management efforts (Kihia *et al*., 2022). Information on the population structure of wild stocks is essential for developing appropriate policies (Marini *et* *al*., 2021). Currently, Kenya lacks specific conservation and management policies for anguillid eels. This study provides critical information on the population structure of anguillid eels in Kenya, laying the foundation for conservation and management strategies and policies.

3.2 Materials and Methods

3.2.1 Study area

This study was carried out at two east-flowing rivers; Athi-Galana-Sabaki (AGS) River is the second largest River in Kenya and rises at 1° 42' S. Athi River, midway, the name changes to Galana and enters the Indian Ocean as Sabaki River (Figure 3). The River stretches 600 Km and drains a catchment of 66,837 km^2, with upper reaches flowing NE across South-east of Nairobi; and then turns direction to the southeast at the north of the Ol donyo Sabuk National Park, where it shares a catchment area with the Tana River Catchment, to pour into the Indian Ocean to the northern of Malindi. The upper zone stands at 1500-2600 m above mean sea level (amsl), the middle zone at 500-1,500m amsl, and the coastal zone below 500 0 m amsl. Daily temperatures in the AGS catchment range between 10°C at the Upper reaches to 30°C in the downstream Coastal reach. The primary sources of the Athi-Galana-Sabaki River are the Ondiri Springs, the Tigoni Falls, the Kikuyu escarpment, Kabete, and Karura Forests. The major towns in this AGS catchment are: Nairobi, Thika, Kiambu, Machakos, Kajiado, and Malindi.

Based on the 2009 Census, the population of the upper reaches is estimated at 9.79 million, or 25.4% of the total population of Kenya, with a density of 167 persons.km^{-2}, (Kenya National Bureau of Statistics, 2010). The per capita renewable water resource of the Athi Catchment Area is calculated at 464 m^3 per year, while the annual water demand is estimated for the year 2010 as 1,145 Million Cubic Meters (MCM), and is projected to increase to 4,586 MCM per year for 2030 in Athi Catchment (Koei, 2013).

The upper zone of 2,600-1,500M (amsl) covers the Aberdares, Kikuyu Escarpment and Ngong Hills. This zone is predominantly volcanic, well-drained compared to the middle and lower reaches, and has relatively good aquifers of considerable value for domestic, community, and commercial water use (GoK, 2017). The implication of this is that the upper areas are well endowed with water resources such as Koma, Ruaka, Ruiru, Mathare, Mutoine, Mbagathi, Ngong, and Nairobi tributaries. Land use in this area is characterized by intensive agriculture of dairy farming, subsistence crop farming such as maize, beans, bananas and horticultural crops; providing a livelihood to about 70% of the area population (Koei, 2013). Coffee and tea are the main cash crops in the area.

The middle zone of 1,500-500M (amsl) the catchment covers the Taita Hills, Tsavo East National Park and Arabuko Sokoke forest reserves, which are important water towers for the region. The river flows through semi-arid terrain before reaching the Galana River Cliffs. Several tributaries namely Rivers Tsavo, Voi and Mbololo contribute to its flow and ecosystem sustainability, providing water for wildlife, vegetation, and local communities (domestic and fisheries). Rangelands in the Middle Athi Catchment Area (ACA) and Nolturesh Lumi area support pastoralism and the Galana Kulalu Irrigation scheme. Unfortunately, these rangelands are increasingly being converted to agro-pastoralism and urbanization with consequent loss of the protective vegetation cover (Koei, 2013). Their fragile volcanic soil, once disturbed by cultivation, becomes susceptible to soil erosion during rainy seasons.

Downstream, the river drains large areas characterized by arid and semi-arid conditions at the Coastal zone of 500-0M (amsl). It is made up of the coastal region of Kilifi and finally Malindi, where the AGS River enters the Indian Ocean at Sabaki Estuary. The region is prone to seawater intrusion, worsening with proximity to the ocean. Water abstraction here is limited due to salinity, which renders it unfit for domestic and agricultural activities (Koei, 2013). The region is of economic importance to tourism, coral limestone, and sand harvesting. Cashew nuts, coconut, and mangoes are the dominant crops grown in the region and are the most important cash crops.

Athi-Galana-Sabaki River basin experiences environmental pressures from catchment degradation (e.g., clearing of vegetation/forests for agriculture and industries) leading, to encroachment and cultivation of wetlands, pollution, especially at the major cities (Nairobi and Malindi), and rapidly growing towns such as Thika, Kiambu, Machakos, and Athi River. These urban centres constitute major sources of pollution from raw/partially treated domestic waste, industrial discharges, and poor solid waste disposal. Overgrazing causes accelerated erosion and subsequent siltation of the Rivers (GoK, 2017). Sampling at the upstream was carried out in Machakos County, Kabaa sub location at Kiaoni Bridge, while at the downstream sampling was done at Kilifi County at Sabaki Bridge.

Ramisi River is in Kwale County in the south-eastern tip of the country. It lies between latitudes 3° 3’ and 4° 45’ South and Longitudes 38° 31’ and 39° 31’ East. The River stretches 200km starting at Shimba Hills (420 msl) to emptying into the Indian Ocean at Funzi Bay. The dominant rainy season is from March to June, while the short rains are from November to December (GoK, 2017). Average rainfall ranges from 500 mm in the rangelands to 1,200 mm in the high rainfall areas of the coastal Strip. The upper part of the catchment supports subsistence agriculture of mainly Maize, Sorghum, Sweet potatoes, Sunflower, Soya beans, cowpeas, Simsim, Vegetables, Cassava, Bananas, Coconuts, Mangoes, Pawpaws, Avocadoes and Curcuma. The population lives in small scattered communities with population decreasing inland away from the coast (Mumma *et al*., 2011). The mid catchment is semi-arid with most of the local economy based on small-scale agriculture and pastoralism, but two other major activities are industrial agriculture (sugar-producing company KISCOL) and mineral exploitation (mining company "Base Titanium") (Ferrer, 2019). The coastal areas host urban communities including Ukunda, Diani, and Msambweni. Ramisi water quality is rated as "good and less polluted" (GoK, 2017) but saline towards the Indian Ocean. The relatively high salinity is due to the inflow of brackish geothermal waters from the Mwananyamala hot springs and the influx of seawater during Oceanic high tides. Sampling was carried out at the Eshu bridge, upstream and at the Taliani site, downstream.

The map of Study Area

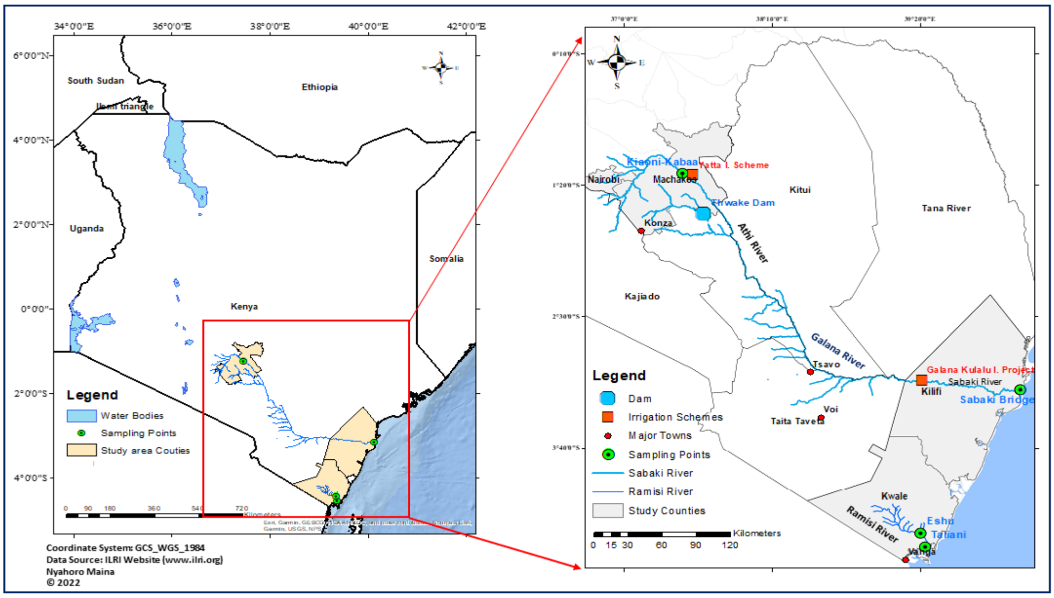


Figure 3: Map of Study Area Showing the AGS and Ramisi Rivers' Sampling Sites

3.2.2 Sampling and research design

Sampling was done between May 2021 and June 2022 where a total of 308 yellow and silver eels (Appendix 3) were landed. Each study river was stratified into upper and lower River reaches. Sampling at the upper River AGS was done at the 'Kiaoni' site and at 'Sabaki' on the lower River AGS. Six fyke nets per site were laid per sampling session, each approximately 30 metres apart. In River Ramisi, sampling was done at 'Eshu' (upper River reach) and Taliani (Lower River reach) (Appendix 6). Three fyke nets were laid per site in each sampling session.

For 10 days in a month, commercial fyke nets of 3 and 10 mm mesh (Appendix 4) were set overnight for twelve months. Eel species landed were identified using morphological features as described by Ege (1939) and Durif *et al*. (2009) (Appendix 2).

The total length of the eels was measured from the tip of the snout to the tip of the caudal fin using a graduated pipe (Appendix 1). Weight measurements were taken to the nearest 0.01g using the "Denver Instrument" weighing balance XL–6100. The eels were grouped into size classes of between <20>80 cm. The size classification was determined through different life stage lengths as indicated by Froese (2006). Relative abundances were calculated as follows;

$$RA\% = {}^{I_{si}}\!/\!{\textstyle\sum N_{si}} \times 100 \quad \ldots\ldots 1$$

Where, Isi = Total Number of individual spp; ∑ Nsi = Total Number of species population.

Diversity indices were calculated as Shannon-Wiener Index (H') (Shannon, 1948);

$$H' = \sum[\left(\frac{ni}{N}\right) X \ln\left(\frac{ni}{N}\right)] \quad \ldots\ldots 2$$

Where n_i = number of individuals or amount (e.g., biomass) of each species (the i^{th} species); N = total number of individuals (or amount) for the site, and ln = the natural log of the number. Shannon diversity (H): 0 = lowest diversity; 3 = highest diversity.

Species dominance (D) was calculated using the formula by Berge and Parker (1970);

$$D = \frac{\sum n(n-1)}{N(N-1)} \quad \ldots\ldots 3$$

where n = the total number of individuals of each species, and N = the total number of organisms of all species. Dominance: 0 = lowest dominance (habitat shared by the maximum number of species); 1 = highest dominance (only one species dominating the habitat).

Evenness was calculated as follows

$$E = \frac{H}{H_{Max}} \quad \ldots\ldots 4$$

Where E is evenness; H is the Shannon diversity index; H_{max} is the maximum possible value of the Shannon diversity index given the number of species present

The index ranges from 0 to 1, with values closer to 1 indicating greater evenness.

3.2.4 Biomass and Density Measurements

Biomass (grams per net per day) ($gnet^{-1}d^{-1}$) was computed by taking the weights of the eels per the number of nets employed at the study sites, per the number of days the nets were set. Density (CPUE) was measured as individuals per net per day by counting the total number of eels per the number of nets set at each site, per the number of days sampling was done.

3.2.5 Data analyses

Bartlett's test was used to evaluate the homogeneity of variances in groups. Kruskal-Wallis test was carried out where assumptions of parametric tests were not met. All statistical analyses were considered significant at $p \leq 0.05$. A T-test was used to compare statistical differences in eel lengths and weights between the rivers. ANOVA was used to analyse significant differences among eel weights and lengths of different size classes and species in the Rivers, and CPUE (Density) of the eels' life stages. Descriptive statistics were used to analyse density, biomass, and relative abundances of size classes between Rivers and among eel species across the Rivers. The influence of water quality parameters on the abundance of eels across the rivers was analysed by Principal Component Analysis (PCA). Eel species abundance and distribution in different biotopes were analysed using Canonical Correspondence Analysis (CCA). Ecological risk assessment was analysed using the Relative Risk Model.

3.3 Results

3.3.1 Relative lengths of eel species between the Rivers

Generally, yellow eels of River AGS were the longest, with *A. marmorata* leading with a mean of 56.6 cm ± 10.6, whereas in River Ramisi, *A. bengalensis* was the longest at 40.2 ± 14.5 cm (Table 1). The longest silver eels occurred in River AGS, led by *A. marmorata* at 104.0 cm (n=1), while *A. bicolor* was the longest silver eel in the River Ramisi at 89.0 cm (n=1). The difference was statistically significant at (t-test t= 4.8; $p \leq 0.05$) between the rivers (Table 1).

Table 1: Mean Lengths of Yellow and Silver Eels of the AGS and Ramisi rivers

	Yellow eels (mean length) ± SE cm		**Silver eels (mean length) cm**	
Species	**AGS**	**Ramisi**	**AGS**	**Ramisi**
A. bengalensis	55.31 ± 15.1[a]	40.2 ± 14.5[a]	72.0 ± 4.6	49.8 ± 5.3
A. marmorata	56.6 ± 10.6[a]	34.4 ± 12.5[b]	104.0 (n=1)	48.6 ± 16.8
A. mossambica	47.2 ± 15.2[a]	28.5 ± 10.3[b]	-	48.6 ± 16.8
A. bicolor	41.1 ± 16.8[a]	29.5 ± 8.5[b]	-	89.0 (n=1)

3.3.2 Relative abundance of Yellow and Silver eels of Rivers AGS and Ramisi

A. bengalensis had the highest abundance of yellow and silver eels in both the Rivers Ramisi and AGS. Of all the yellow eels caught in the River Ramisi, 86.9% were *A. bengalensis*; 5.1% were *A. bicolor*, 5.7% were *A. mossambica*, and 2.3% were *A. marmorata*. In the River AGS, 50.1% were *A. bengalensis*, *A. bicolor* were 27.9%, whereas *A. mossambica* and *A. marmorata* trailed with 15.7% and 6.3%, respectively (Figure 4). In the River Ramisi, silver eels of *A. bengalensis* were 46.87%, 40% for *A. bicolor*, 6.3 % for *A. mossambica*, and 6.3% for *A. marmorata*. In River AGS, all the silver eel catches were *A. bengalensis* (Figure 4).

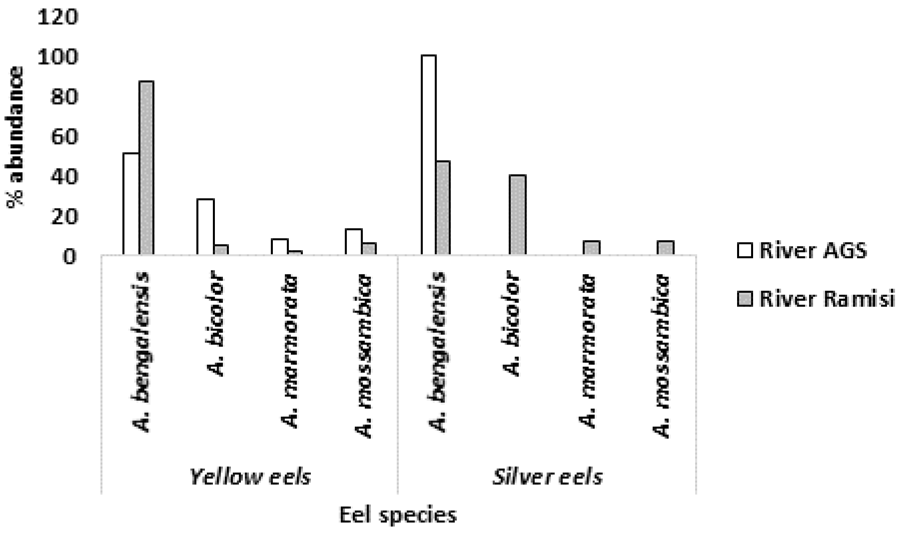


Figure 4: Relative Abundance of Yellow and Silver Eels of the Rivers AGS and Ramisi

3.3.3 Diversity of anguillids at Rivers AGS and Ramisi

Generally, eel diversity was not statistically different between the rivers (t-test; t=0.48, p=0.63). However, diversity between the downstream locations of AGS (0.67±0.13) and Ramisi (0.42±0.21), and subsequent upstream locations (0.24±0.13) and (0.35±0.12) showed significant differences (Table 2). Species richness differed statistically between up and downstream reaches of the rivers (Table 2). The diversity at upper and lower AGS was comparable, unlike that at the Ramisi, which showed a statistical difference (Table 2). Eel dominance was comparable among the upstream locations (0.85±0.1) at River AGS and (0.82±0.06) at River Ramisi. The highest species evenness (0.96±0.11) was recorded at the downstream location of River Ramisi, which was statistically significant among the other locations.

Table 2: Diversity Indices of Eel Communities Along Rivers AGS and Ramisi

Rivers	**Reaches**	**Richness (S)**	**Frequency (n)**	**Dominance_D**	**Evenness_e^H/S**	**Richness index (D)**	**Diversity (H)**
AGS	**Upper**		29	0.85± 0.1ab	0.94±0.07a	0.46±0.19a	0.24±0.13ab
	Lower	4	41	0.58±0.1a	0.91±0.07a	0.75±0.16b	0.67±0.13b
Ramisi	**Upper**	4	222	0.82±0.06b	0.84±0.06a	0.41±0.15a	0.35±0.12a
	Lower	3	12	0.71±0.11a	0.96±0.11ab	0.72±0.25b	0.42±0.21b

* Variables with different letters (a, b) in the same columns indicate significant differences

3.3.4 Eel Biomass

Total eel biomass per river reaches

The highest total eel biomass was at the upper River Ramisi (165.00±46.27) and was 1.6 fold higher than at the lower Ramisi (97.60±71.5), whereas at the upper River AGS (110.90±56.9), biomass was 3.5 fold higher than at the lower AGS (32.20±10.0). Eel biomass differed statistically (ANOVA; F=3.00, p=0.04) among the river reaches (Figure 5)

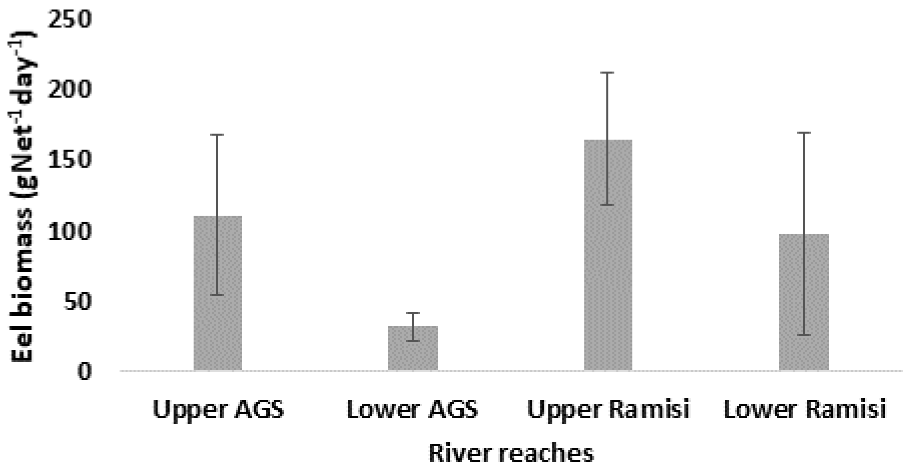


Figure 5: Total Eel Biomass at Upper and Lower River Reaches

Individual species biomass at the upper and lower River AGS

Individual species biomass at River AGS showed *A. marmorata* having the highest biomass (222±187 gnet^{-1}d^{-1}), followed by *A. bengalensis* (72.4 ±17.7 gnet^{-1}d^{-1}) at the upper reaches of River AGS (Figure 6). *A. bicolor* was absent at the upper reaches of AGS.

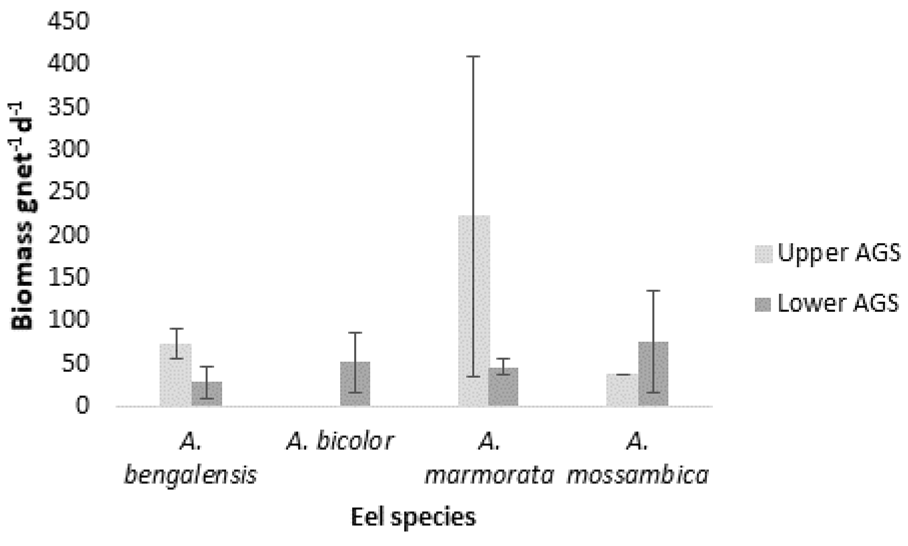


Figure 6: Species Biomass at the Upper and Lower Reaches of AGS

Individual species biomass at the upper and lower River Ramisi

At the River Ramisi, the highest biomass occurred at the upper reach with *A. bengalensis* leading (288±84.5 gnet^{-1}d^{-1}), followed by *A. bicolor* (51.1±38.8 gnet^{-1}d^{-1}), whereas *A. marmorata* had the least biomass (10.30 ±0.89 gnet^{-1}d^{-1}). At the lower Ramisi *A. mossambica* had the highest biomass (48.75±18 $gnet^{-1}d^{-1}$), followed by *A. bicolor* wth (5.83±10 $gnet^{-1}d^{-1}$), whereas *A. marmorata* and *A. bengalensis* were absent at the lower reach (Figure 7)

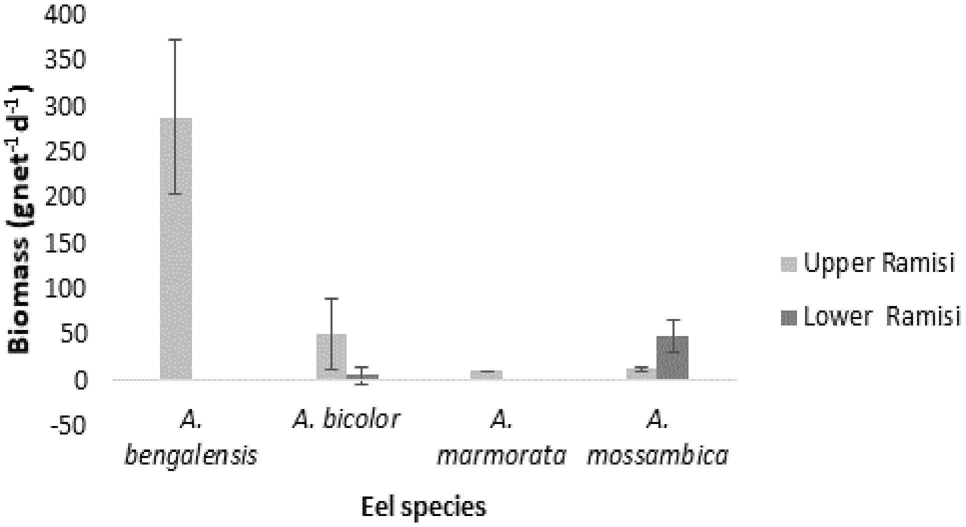


Figure 7: Species Biomass at the Upper and Lower Reaches of the River Ramisi

3.3.5 Densities (Individuals.$net^{-1}day^{-1}$) of the eel species and life stages between the Rivers

Species densities per River

Generally, the River Ramisi ecosystem had higher eel densities (Catch Per Unit Effort) ($Individuals^{-1}net^{-1}day^{-1}$) than River AGS. *A. bengalensis* of the River Ramisi had the highest CPUE (1.7 $Ind.^{-1}net^{-1}d^{-1}$), followed by *A. mossambica* (Ramisi) (1.3 $Ind.^{-1}net^{-1}d^{-1}$), whereas *A. marmorata* of the River Ramisi had the least (0.64 $Ind.^{-1}net^{-1}d^{-1}$) CPUE (Figure 8). *A. marmorata* was the only species that had a higher CPUE in the River AGS than in the River Ramisi.

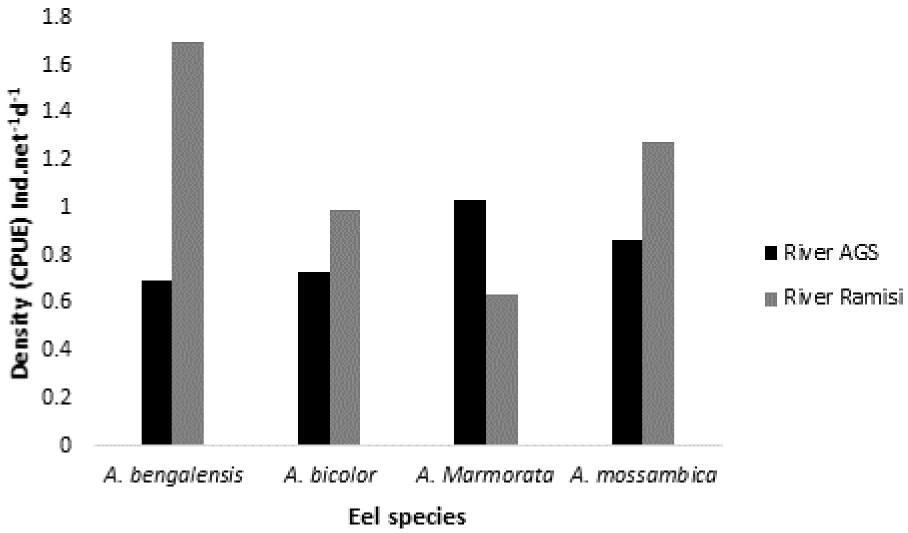


Figure 8: CPUE of the Eel Species Between the Rivers

3.3.6 Size class abundance of the eel species occurring in River AGS and River Ramisi

Size class abundance of *Anguilla bengalensis*

The size class abundance graph for *A. bengalensis* took a near pyramid shape in both Rivers with peaks at size class 50-60; 42.4% in the River AGS and 37.3% in the River Ramisi (Figure 9). Smaller eels, in the size classes of less than 20 cm to 30–40 cm, were more abundant in the River Ramisi compared to the River AGS. In contrast, larger eels over 60 cm were more common in the River AGS (33.3%) than in the River Ramisi (5.1%) (Figure 9).

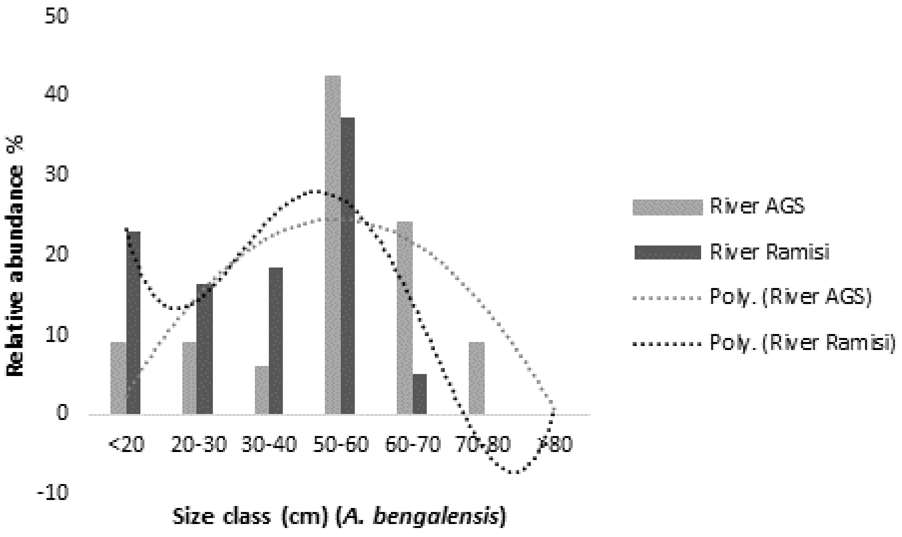


Figure 9: Size Class Abundance of *A. bengalensis*

Size class abundance of *Anguilla bicolor*

Size class abundances graph for *A. bicolor* in both Rivers took a near dome-shape, though skewed more to the left (Figure 10). The peak for relative abundance in River AGS occurred at size class 20-30 (36.8%), whereas at River Ramisi the peak was at size class 30-40 (29.4%). Abundance in both Rivers reduced drastically from size class 60-70 to >80 (ranged between 5.9% and 0% respectively) (Figure 10)

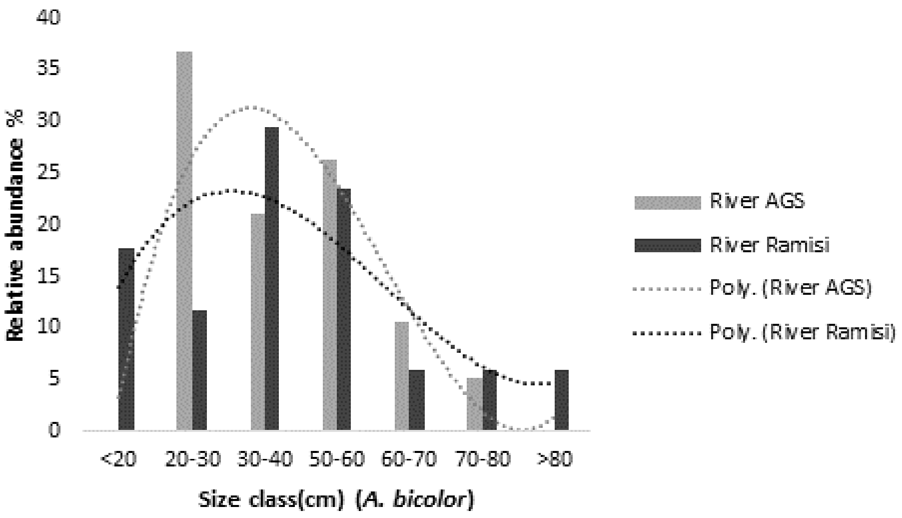


Figure 10: Size Class Abundance of *A. bicolor*

Size class abundance of *Anguilla marmorata*

At the River Ramisi, *A. marmorata* had higher abundance at smaller size classes with a peak at size class 30-40 (50%). All size classes >60 were absent in Ramisi, skewing the graph towards the left (Figure 11). On the contrary, smaller size classes <40 were absent in the AGS, but had a peak at size class 50-60 (66.7%). All size classes >60 in River AGS were present, with the longest size class of > 80 having a relative abundance of 11%

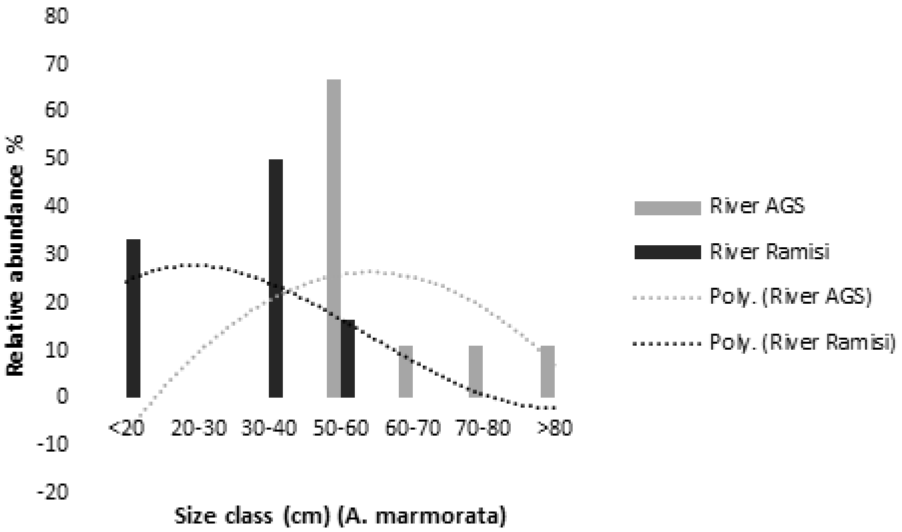


Figure 11: Size Class Abundance of *A. marmorata*

Size class abundance of *Anguilla mossambica*

At the River Ramisi, *A. mossambica* had concentrated abundance at size classes <60, with a peak at size class 20-30 (43.8%) (Figure 12). Size classes >60 recorded no eels, skewing the graph towards the left. At the River AGS, the graph took a near dome shape with a peak at size class 60-60 (44.4%), whereas size class 70-80 recorded 11% relative abundance (Figure 12).

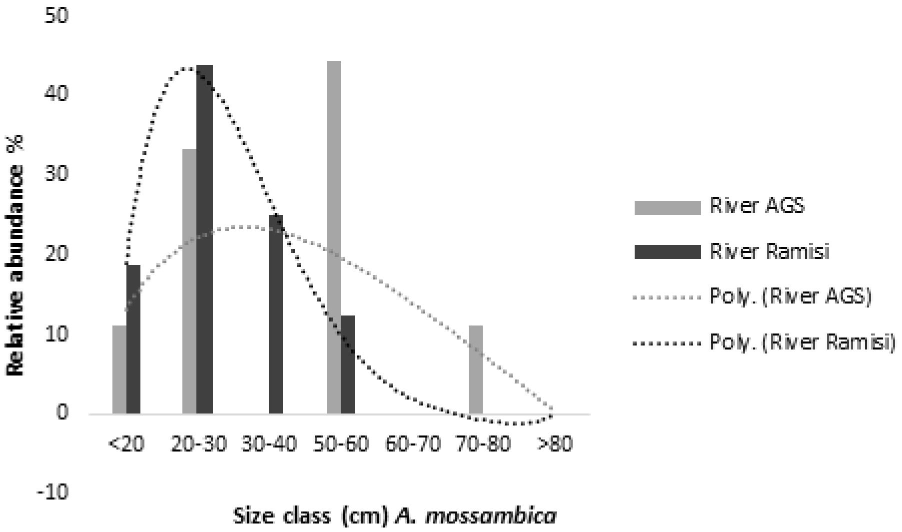


Figure 12: Size Class Abundance of *A. mossambica*

3.4 Discussion

In the present study, the Yellow and Silver eels of the River AGS exhibited greater length compared to those found in the River Ramisi. However, despite this length disparity between the rivers, the mean length ranges of Yellow eels in both rivers (28.5 – 56.6 cm) fell within the mean range (20.8 – 60.9 cm) reported in a study by Guimaraes *et al*. (2009). Woolnough *et al*. (2009) noted that apart from environmental and biological conditions of aquatic ecosystems influencing fish growth, the size of the water body also plays a pivotal role in determining fish size. This is because larger water bodies provide more diverse habitats for predator evasion, increased productivity, and varying temperature zones conducive to eel growth. According to Helfman *et al*. (1987), eels tend to attain larger sizes with increased latitude and distance from spawning areas. The River AGS, being larger (600 km long; average width of 75 meters) than Ramisi (200 km long; average width of 15 meters), and extending further away from the spawning sites (Mascarene Islands), may have contributed to the larger eels in the River AGS compared to River Ramisi, as reported by Helfman *et al*. (1987).

The relatively higher CPUE of all eel life stages in the River Ramisi compared to the River AGS could be attributed to the suitability of the ecosystem for eel recruitment and maintenance, governed by the integrity of biotic and abiotic factors conducive to anguillid growth and survival. The highest eel densities occurring in Upper River Ramisi over other river reaches confirm the suitability of the upper River Ramisi for eel growth. Itakura and Wakiya

(2020) reported that eel densities increase with a combination of habitat factors that are more likely to occur in ecosystems providing sufficient biotic and abiotic resources.

Anguilla bengalensis yellow and silver eel stages were the most abundant in both rivers, followed by *Anguilla bicolor*. *Anguilla marmorata* and *Anguilla Mossambica* trailed in abundance in both rivers. River Ramisi portrayed a more successful ecosystem for yellow eel maintenance and silver eel escapement, in all eel species, compared to River AGS, as indicated by the abundance of yellow and silver eels.

Generally, the population structure of the eels was characterized by an expansive population of young and growing individuals, indicating high recruitment and growth. High silver eel migration was evidenced by the low numbers of larger eels landed. The abundance of different size classes of eels in the current study was consistent with results from other studies determining eel population dynamics and growth (Lambert *et al*., 2006; Panfili *et al*., 1994). Variations in size class abundance could be related to the number of glass eels recruited into the rivers at different seasons. Subsequently, the recruiting glass eels would develop into yellow eels of different sizes and abundances. A study by Sugeha (2010) on the recruitment of tropical eels (*Anguilla celebesensis* and *Anguilla marmorata*) in the Poso estuary, Indonesia, showed that recruitment was season and species-specific. Recruitment occurred in the dry season (May to September) and the rainy season (October to April). In the study, *Anguilla celebesensis* and *Anguilla marmorata* appeared as dry-season recruiting species, while *Anguilla interioris* and *Anguilla bicolor* were identified as rainy-season species.

Glass eels and riverine yellow eels of *Anguilla bicolor*, *A. bengalensis*, *A. marmorata*, and *A. mossambica* were sampled in estuaries and rivers of La Réunion Island, Mauritius Island, Mayotte Island, and on the eastern coast of Madagascar between 2000 and 2006, suggesting that the composition of eel assemblages seems contrasted between the different islands of this region (Robinet *et al*., 2008). This contrasted distribution may be due to diverse migratory paths followed by the eel larvae before their estuarine recruitment (Réveillac *et al*., 2008; Robinet *et al*., 2007; Robinet *et al*., 2003), varying spawning localities or seasons among species (Robinet *et al*., 2003), or different developmental traits or innate metabolism (Réveillac *et al*., 2008; Robinet *et al*., 2003; Robinet *et al*., 2008). These findings can be related to the current study, where different size classes had differing abundances because of differences in the recruiting number of individual species and seasonal recruitment variations. Species-specific recruitment governed by seasons (Sugeha, 2010) could lead to differing abundances of the four eel species as observed in the current study.

In the current study, eel biomass differed among river reaches, a scenario that could be attributed to the influence of habitat conditions on eel growth and distribution, as reported in a study by Glova *et al*. (1998) on short and longfin eels in three New Zealand lowland streams. The study assessed the relationship between eel biomass and several instream variables and found that biomass of small shortfin eels was negatively correlated with distance from the ocean, mean water velocity, and median substrate size, whereas that of longfins was positively related to instream debris. Similar to the current study, Glova *et al*. (1998) noted considerable variance in eel biomass between streams for both species. Moreover, in the current study, similar eel species portrayed different biomass values at different river reaches, further confirming similarities with the findings of Glova *et al*. (1998).

The diversity of eels was not statistically different between river habitats, generally portraying low diversity, indicating relatively few species present in the ecosystem, and those species are not evenly distributed in terms of abundance. Reduced diversity leads to ecological simplification, altering local interactions and controlling biodiversity and ecosystem function. Dominance was high with values towards one, meaning minimal variation or diversity. In this case, one specific species or trait is overwhelmingly prevalent compared to others in the same habitat. These findings are consistent with the findings of declining eel species as reported by Jacoby *et al*. (2015).

3.5 Conclusion

In conclusion, the population structure analysis of anguillid eels in Rivers AGS and Ramisi revealed significant ecological insights, providing critical data for understanding species distribution and habitat utilization. The identification of four distinct anguillid eel species highlights the biodiversity of these rivers and their role in supporting freshwater eel populations. However, variations in population structure across species and between different river sections indicate that ecological and environmental factors influence eel distribution and abundance.

Contrary to the null hypothesis that there was no difference in the population structure of anguillid eels between River AGS and River Ramisi, the findings suggest habitat-specific variations. The predominance of longer eels in River AGS, alongside higher biomass and Catch Per Unit Effort (CPUE) in River Ramisi, underscores differences in habitat conditions, resource availability, and recruitment success. The dominance of the yellow eel life stage across both rivers further emphasizes the importance of these freshwater ecosystems for the growth and development of anguillid eels. Additionally, the higher relative abundance of mid-length size classes suggests ongoing recruitment and growth patterns that require further investigation.

Understanding these population dynamics is essential for fisheries management and conservation. These findings provide a foundation for assessing the health and sustainability of anguillid eel populations in Kenya, informing strategies to mitigate anthropogenic threats and ensure long-term species viability. Further research is necessary to investigate the underlying drivers of population structure variations and monitor changes over time, facilitating evidence-based conservation and management initiatives.

CHAPTER FOUR

EVALUATION OF THE ECOLOGICAL NICHES OF ANGUILLID EELS IN THE ATHI-GALANA-SABAKI AND RAMISI RIVERS

Abstract

This study assessed of the ecological niches occupied by anguillid eels within the complex river systems of the Athi-Galana-Sabaki (AGS) and the Ramisi. Focusing on dietary patterns and habitat preferences, our research explored the dynamic interaction between these eels and their surrounding river environments. Utilising humane techniques, 68 anguillid eels were carefully examined, with stomach contents analysed to identify nine distinct diet items, revealing fish as the predominant dietary component. Niche breadth varied between 0.27 and 3.4, while trophic levels ranged from 3.1±0.4 to 3.7±0.1. High niche overlap (0.65±0.5) was observed between *Anguilla bengalensis* and *Anguilla bicolor*. Relative condition factors (Kn) ranged from 0.15±1.08 to 0.86±0.42 across species. Length-weight relationship analysis yielded b values between 2.67 and 3.55, with *A. bengalensis* exhibiting the lowest and *A. mossambica* the highest. Notably, *A. mossambica* elvers displayed the highest b value (4.8), while *A. marmorata* silver stage exhibited the lowest (0.3). Habitat preferences varied among species, with *A. bengalensis* and *A. marmorata* favouring the upstream reaches, while *A. mossambica* and *A. bicolor* dominated the lower reaches. Biotope selection was river-specific, with *A. bengalensis* and *A. marmorata* preferring vegetated pools in the Ramisi, whereas vegetation presence was not significant in the AGS. In the River Ramisi, *A. mossambica* and *A. bicolor* showed a preference for fine substrate. Physical-chemical parameters positively influenced *A. marmorata* in the upper AGS and *A. mossambica* in the lower Ramisi. In conclusion, this study provides valuable insights into the ecological dynamics of the anguillid eels in the AGS and Ramisi rivers, emphasizing the intricate relationships between eel populations and their aquatic habitats. These findings contribute to our understanding of freshwater ecosystems and informs conservation efforts aimed at preserving eels' biodiversity and ecosystem health.

4.1 Introduction

Anguilla eels are globally distributed catadromous fishes that spend most of their lives in freshwater, before migrating to the Ocean to spawn (Arai, 2016). In the ocean, eggs hatch into leptocephali (larvae), which metamorphose into glass eels before entering continental waters and grow into elvers, yellow eels (immature adults), and finally, into silver eels (mature adults) that undertake return oceanic spawning and migrate and die (Arai,2016). Their existence is threatened by over-exploitation, habitat loss, climate change, pollution, and the introduction of alien species (Collier *et al*., 2016). Most of them are listed in different categories in the International Union for Conservation of Nature (IUCN) red list of endangered species (Jacoby *et al*., 2015).

Anguilla spp. is a widely consumed fish of high economic value and an export commodity to markets in Europe and Asia (Shiraishi & Crook, 2015). The market demand for eels continues to rise, especially in Asian, American, and European countries which reach 600,000 tons per year (Pangerang, 2022). Due to the reduced supply of commercially important temperate eels, the poorly studied tropical species are being targeted as an alternative to satisfy the high demand for eels and eel products (Cadiz & Traifalgar, 2020). West Indian Ocean (WIO) where Kenyan coast lies is home to four anguillid species; *Anguilla bengalensis, A. bicolor, Anguilla marmorata* and *Anguilla mossambica* (Jacoby *et al*., 2015).

During the early larval stage in the open ocean, anguillid eel larvae (Leptocephalus) primarily feed on marine snow, which includes exudates from bacteria and phytoplankton (Tsukamoto & Miller 2021). The exudates release polysaccharides and simple sugars and provide the base to facilitate other particles to stick together into marine snow (Miller *et al*., 2020). These carbohydrates become an important food source for the leptocephali, in addition to nutrition obtained from the digestion of the microorganisms aggregated in marine snow. In the early eels' life, the transition from marine to freshwater is a critical stride preceded by metamorphosis from the marine larval to the continental glass eel stage, involving energy-taxing morphological, physiological, and behavioural shifts during which time they do not feed (Van Wichelen *et al*., 2022). On arrival at the estuary, glass eels feed on detritus and wormlike benthic invertebrates (Lagarde *et al*., 2022; Van Wichelen *et al*., 2022). Eventually, food uptake intensifies and their diet diversifies, including a wide range of planktonic and benthic organisms.

In freshwater, yellow eels become more predatory, preying on a wide range of marine, estuarine, and freshwater fauna. The principal food is invertebrates (especially molluscs and crustaceans) and fish (Dörner & Berg, 2016; Maitland & Creg, 2010; Nishimoto *et al*., 2023). Yellow eels also scavenge on dead fish (Coad, 2016). In some cases, they are reported to leave the water channel and enter marshy fields to feed on terrestrial fauna, such as slugs and worms (Coad, 2016). Silver eel stage experiences regression of the digestive tract and therefore, it does not feed but survives on the fat reserves accumulated during the yellow eel stage (Hayden *et al*., 2019; Rousseau *et al*., 2013; Trischitta *et al*., 2013). One of the most important life-history traits that affect individual characteristics influencing food acquisition is body size

Information on the diet of a species in nature is of the essence in the establishment of its nutritional requirements and the interaction with other organisms (Albertoni *et al*., 2003), and it is an integral tool for assessing the structure and functioning of ecosystems (Maestre *et al*., 2016). The feeding ecology of a species is greatly linked to population dynamics and contributes to the knowledge of resource partitioning, prey selection, predation, habitat preferences, competition, evolution, and energy transfer within and between ecosystems (Braga *et al*., 2012). Such ecological knowledge is of significant value when developing conservation plans; therefore, it is a fundamental element in protecting species and ecosystems (Braga *et al*., 2012).

Anguillids are ecologically valuable because adult eels are long-lived top predators in freshwater systems, feeding on crabs, frogs, shrimps, and fish (Denis *et al*., 2022). Hartanto *et al*. (2016) based on stomach content analysis of *A. bicolor* and *A. marmorata*, identified three dominant types of foods, namely, crabs, shrimps, and earthworms. However, statistical analysis showed no significant differences in food variation during the three months of sampling in that study. Other studies by Rupasinghe and Attygalle (2006) on *A. bicolor* in the Bolgoda Estuary, Bandaragama, Sri Lanka showed that the eel fed on invertebrates when they were at a smaller size and changed to piscivorous (crabs, shrimps, small fish, and worms) when they became large.

The condition factor is used to examine the overall condition, the fatness, or well-being of a fish. The reliance on the condition factor is based on the theory that the heavier fish of a given length is in a better condition (Koushlesh *et al*., 2018). Length-weight relationship is a prerequisite for assessing population characteristics and indicates the degrees of stabilization of taxonomic characters in species (Pervin & Mortuza, 2008). Such studies are essential for assessing population dynamics, development of stock assessment models, and estimation of the health condition of the fish, and for life history and morphological comparisons of populations from different regions (Le Cren, 1951). In addition, the growth pattern obtained from the slope parameter (*b*) of the length-weight relationship and condition factor (Kn) plays a vital role in understanding fish ecological adaptation (Abdoli *et al*., 2009; Dinh *et al*., 2016), explained by growth categories. The growth of fishes can be explained in three categories i.e., isometric, negative, or positive allometric growth. The cube law states that for isometric growth, the weight (or mass) of the organism is proportional to the cube of its length. Negative allometry occurs when the fish becomes lighter with length, which is attributed to insufficient diet and environmental changes. In positive allometry the fish increase in weight at a faster rate with its increase in length. Fulton's (1904) cube law, Froese (2006) stated that, in the juvenile stages of fish, their growth rate is faster in length than width or any other direction. Thus their length-weight relationship changes as the fish grows in length. According to the general cube law, the weight of the fish varies with the cube of the length.

The amount of energy present in a prey impacts weight increase, as well as the ratio of size and length (Cárcamo *et al*., 2019), and therefore, growth in length and weight is complementary and their rates of increase depend on the amount of prey energy (Wootton,2012). An energy-rich diet means that fish can have a high consumption rate, whereby costs of maintaining metabolism and synthesizing new tissue are covered (Wuenschel *et al*., 2006). In contrast, fish without an energy-rich prey intake rate will assign the energy budget to basic maintenance, affecting their rates of consumption and assimilation efficiency (Gallagher *et al*., 2022), thus compromising the making of new tissue.

The contrast in dietary niche breadth has been linked to various aspects of species distribution (Lanszki *et al*., 2022), variety of spatial distributions (Alberdi *et al*., 2020), and habitats occupied (Fargallo *et al*., 2022; Walsh & Tucker, 2020), and reactions to landscape change (Kellner *et al*., 2019). The contrast in dietary niche breadth fosters resource partitioning among coexisting species within a community (Lanszki *et al*., 2022). When different species have definite dietary preferences or consume different food resources, they coexist without direct competition for the same limited resources. This leads to more versatile and stable ecosystems. Species with a broader dietary niche breadth can utilize a wider range of habitats or environmental conditions (Alberdi *et al*., 2020). This versatility enhances the ability to colonize new habitats or adapt to changing conditions, leading to broader geographic distributions. Conversely, species with narrower dietary niches are more restricted to specific habitats that provide for their preferred food resources. The contrast in dietary niche breadth influences species interplay such as predation, competition, and mutualism (Fargallo *et al*., 2022). It contributes to biogeographic designs by controlling species distributions across differing spatial scales. For example, some dietary specialists may portray restricted distributions, confined to specific regions where their preferred food resources occur. In contrast, generalist species with wider dietary niche breadths may have broader geographic ranges, stretching across multiple habitats or biomes (Fargallo *et al*., 2022).

Considering the various variables that can determine dietary niche overlap, regional disparity could affect broad-scale patterns of niche overlap between species (Walsh & Tucker, 2020). The overlap is influenced by various factors, including similar niches, competition for resources, habitat requirements, physical barriers (such as dams and weirs that act as barriers to species dispersal), changes in climate (which can shift suitable habitats and resource availability), and human-induced factors like habitat destruction, pollution, introduction of invasive species, and climate change. These factors can significantly impact species overlap by altering habitats and resource availability.

The size of an organism can be a strong predictor of trophic position and the diversity of prey a species can consume. Jellyman (1989) found that the diet of each of the eel species he studied changed with size. The smallest eels of both species ate mainly amphiphods insect larvae and small molluscs incomplete sentence. Therefore, consumer trophic ecology is driven by a trade-off between environmental factors and biological traits (Hayden *et al*., 2019). For example, fishes exposed to benthic or pelagic habitats will be adapted to totally different prey communities; a predator's morphology, such as its size, mobility, and dentition will dictate what prey types it can capture; and the spatial range of a predator will also influence the breadth of prey it may consume (Stuart-Smith *et al*., 2013). The amount of energy present in a prey impacts weight increase, as well as the ratio of size and length (Cárcamo *et al*., 2019), therefore, growth in length and weight is complementary and their rates of increase depend on the amount of prey energy (Wootton, 2012). An energy-rich diet means that eel can have a high consumption rate, whereby costs of maintaining metabolism and synthesizing new tissue are covered (Wuenschel *et al*.,2006). In contrast, fish without an energy-rich prey intake rate will assign the energy budget to basic maintenance, affecting their rates of consumption and assimilation efficiency (Rosen & Worthy, 2018), thus compromising the making of new tissue.

The geomorphological and hydrological conditions of rivers are highly diverse and dynamic and provide various habitats for fish and other aquatic life (Langerhans, 2008). In rivers three main zones exist; the crenon (spring zone), the rhithron (middle course) characterized by turbulent - oxygenated waters, and the potamon (lower course) which is the downstream reach and characterized by slow and turbid water flows in the plains (Junk & Piedade, 2011). The temporal shifts between floods and low lotic flows create a large variety of habitats, of diverse durations, over time. Floodplain waters carry over nutrients from the decomposition of organic matter and vegetation, as well as animal manure (cattle, wild animals) into river channels. This leads to the rapid growth of algae, zooplankton, bacteria, and a rich fauna of aquatic invertebrates. After the floods, the water leaves the floodplains through a series of channels to return to the river's main watercourse creating pools that may remain throughout the year and serve as a refuge for aquatic fauna. At the same time, aquatic vegetation grows rapidly, adding to the characteristics of the biotopes.

In the river system, the upper and lower reaches consist of diverse biotopes of biotic and abiotic characteristics (Guareschi & Wood, 2019). The riverine distribution of eels can be affected by environmental factors in microhabitats and biotopes, such as depth, velocity, sediment, aquatic vegetation, and riverbank conditions (Kume *et al*., 2019; Laffaille *et al*., 2003). The influence of biotopes on eel abundance and diversity encompasses factors related to trophic interactions, habitat quality, food availability, and genetic adaptation (Bănăduc *et al*., 2022).

Itakura and Wakiya (2020) studied the preference of biotopes by eels and found that the density of eels was consistently lower among riverbanks consisting of concrete and sand, while it was highest when riverbanks and main channels consisted of vegetation. Individual eels were found in a broader range of habitats with diverse depths and discharge velocities. Densities increased in mud and boulder substrates and combination with habitat factors (Wakiya, 2020).

Kumai *et al*. (2021) studied the habitat preference of *Anguilla marmorata*. They found that the species preferred habitats with larger size substrate materials, fewer fallen leaves and leaf detritus, higher current velocity, and lower turbidity than *Anguilla japonica*. Both species co-occur in the same river systems, although the distributions of the two anguillid eels were segregated. *Anguilla marmorata* preferred clear-flowing mainstreams and tributaries, whereas *Anguilla japonica* preferred stagnant muddy estuaries, backwater areas, irrigation channels, and reservoirs. In another study, Nguyen (2018) and Hsu *et al*. (2019) reported that *Anguilla marmorata* tends to live in freshwater areas rather than in brackish and marine habitats, although other studies (Arai & Chino, 2018; Kumai *et al*., 2020; Wakiya *et al*., 2019) have shown that the species inhabit a broad range of habitats ranging from brackish estuaries to upland headwaters. Studies carried out by Arai *et al*. (2020) in the north-western part of Peninsular Malaysia showed that *Anguilla bicolor* was distributed widely from the upstream to the downstream areas of the rivers, although it was more abundant in the downstream and midstream reaches and rarely found in upstream areas. *Anguilla bengalensis,* however, was found to prefer mainly upstream reaches and rarely occurred in midstream areas. Other studies concurred with Arai *et al*. (2020) and indicated that the African mottled eel (*Anguilla bengalensis)* mainly inhabited areas far inland by conquering formidable barriers during their upstream migration (Craig, 2015; Davies *et al*., 2021). *Anguilla mossambica*, a species with plain skin colouration, is an inter-habitat migrant, moving between freshwater and seawater habitats (Lin *et al*., 2012). In other studies, *Anguilla mossambica* was found to inhabit coastal and estuarine areas without migrating upstream into inland freshwater reaches (Lin *et al*., 2015).

Hsu *et al*. (2020), in a study on biotope preferences, divided tropical and subtropical eels into two broad groups depending on skin colouration (marbled or plain). The study found that in tropical/subtropical areas, marbled and plain eels coexisted, but marbled eels tended to inhabit the middle to upper sections of the rivers, while plain eels preferred the middle reaches to estuaries. They concluded that the countershading of plain eels may be an adaptation that allows them to blend into the mud and sand substrates commonly found in the lower reaches of rivers, helping them avoid predators. In contrast, marbled eels appear to be adapted for camouflage among cobble, gravel, and fallen leaves typically found in the upper river reaches (Unmack, 2013).

This study aimed to determine the habitat use through diet studies and habitat preference of anguillid eel species occurring in the River Athi-Galana-Sabaki and River Ramisi of Kenya. Information on the habitat ecology of the anguillid eels in Kenyan Rivers is lacking, and this is the first study of this kind and the region. The information from this study will assist in understanding the ecological niche of anguillid eels in Kenya, for the conservation and management of eels and their ecosystem

4.2 Materials and methods

4.2.1 Dietary evaluation

A subsample of the total eels (68 eels) caught was selected for diet studies. Subsamples were considered per species, number of eels caught, and the sampling sites where they were landed. The subsamples were immobilized in an overdose of clove oil before being euthanized for the studies. The belly was cautiously split using a sharp blade, and the stomach contents were put on a petri dish containing 1% normal saline solution. The contents were examined under a dissecting microscope. They were identified to the lowest taxon possible using dichotomous keys (Thorp & Rogers, 2014). The eels that were not euthanized were released back into their habitat. *Anguilla marmorata* was not sacrificed for diet studies because of the low number of individuals caught (N = 6). The frequency of diet items was calculated as in formula 5:

$$\%F_{fi} = \frac{N_{fi}}{N_f} \times 100 \quad \ldots\ldots 5$$

Where, % $\boldsymbol{F_{f\ i}}$ is the frequency of occurrence of given item ***i***, $\boldsymbol{N_{f\ i}}$ is the number of stomachs in which the given item ***i*** occurs, and $\boldsymbol{N_f}$ is the total number of stomachs

The trophic (TL) level of each eel species was calculated using the formula 6 as proposed by Adams *et al*. (1983)

$$\text{Trophic level} = \sum_{i=1}^{n}(\text{Proportion of diet from prey}_\text{i} \times \text{Trophic level of prey}_\text{i}) + 1 \quad \ldots\ldots 6$$

Where *n* represents the number of different prey items in the species' diet.

The proportion of Diet from Prey$_i$ is the proportion of the species' diet that comes from a specific prey item$_i$.

The trophic level of Prey$_i$ is the trophic level of the prey item$_i$. The sum is calculated for each prey item in the diet, and 1 is added to the final result to represent the trophic level of the species itself.

Diet breadth was calculated using the following formula by Feinsinger and Poole (1981);

$$B_i = \frac{1}{(n-1)}\left\{\frac{1}{(\sum_j Pij^2}\right\} - 1 \quad \ldots\ldots 7$$

Where B*i* = standardized index of niche breadth, pij = proportion of the diet of predator i on prey j, and n = total number of items (resources), whereby a value of 1 in diet breadth indicates generalist taxa that utilize all available resources and 0 indicates specialist taxa that favour specific resources.

Niche overlap was calculated using a method described by Pianka (1981), as follows;

$$o_{jk} = \frac{\sum_i^n pij p_{ik}}{\sqrt{\sum_i^n pij^2 \sum_i^n pik^2}} \quad \ldots\ldots 8$$

Where O_{jk} = Pianka's index of niche overlap between species j and k, pij = the proportion of the ith resource in the diet of species j, pik = the proportion of the *ith* resource in the diet of species k, and n = the total number of items. Diet overlap of value >1 will indicate increased sharing of diet items while <1 is the reduction in diet sharing between species.

The length-weight relationship was estimated by the following formula by Le Cren (1951) as;

$W = aL^b$ ……………………………………………………………………………………9

This formula is expressed logarithmically as Log W = Log a + b Log L;

Where W = Body weight of the eel, L = Total length of the eel, 'a = a constant being the initial growth index, and 'b' = growth coefficient. Parameters 'a and 'b' were calculated by the least squares regression method. The result was interpreted as R^2 = Coefficient of determination (* b < 3 = negative allometric growth, b=3.0, isometric growth; b > 3, positive allometric growth).

Condition factor (K) was calculated using the formula by Le Cren (1951) as follows;

$Kn = \frac{W}{L^b}$ ……………………………………………………………………………………10

Where K is the condition factor, W is the weight of the fish, L is the length of the fish, and *b* is the allometric coefficient.

4.2.2 Measurement of water quality parameters

The study area is described in Chapter 3 (3.2.1). Channel discharge was measured by the 'float gauging' method using oranges of the same weight that were thrown into the River channel, and the velocity multiplied by the average cross-sectional area of the river. Conductivity, Total Dissolved Solids (TDS), temperature, salinity, and dissolved oxygen were measured in situ using a hand-held multi-parameter water quality meter (YSI 556 MPS model, USA).

Water and hydrological variables were taken at each site during the sampling session before retrieving the nets. River depth was measured using a graduated string fastened with a piece of iron bar sank into the deepest parts of the rivers, while river width was taken across the river with a graduated string. All measurements were taken at intervals of 10 metres and repeated in triplicates at each sampling site.

Twelve biotopes were identified according to lotic speed, substrate, and habitat cover. The substrate was categorized into four types using soil sieves (e.g. bedrock) as follows; silt (70% <0.05 mm), sandy silt (70% = 0.02mm to 1mm), sand (70% = 1mm to 2mm), pebbles (70% >2mm), or bedrock. The river covers were evaluated as under vegetation or no vegetation, boulders (>256mm), bank/riparian vegetation, hydrophytes, or "no cover". The biotopes were at the upper and lower reaches of the rivers. Fyke nets were set at each of these biotopes and the density (CPUE) of the eels was obtained.

4.2.3 Data analysis

Levene's Test was used to evaluate data homogeneity to test for normality of data. Where data was homogenous and followed a normal distribution curve (normality), Analysis of Variance (ANOVA) was used at $p \leq 0.05$. Where homogeneity was not achieved, Kruskal-Wallis tests and F statistics were applied. Descriptive statistics were applied to gain insights into patterns, trends, and relationships within the data. Least significant difference (LSD) statistical analysis was performed to show differences ($p \leq 0.05$) in environmental water parameters across upper and lower river reaches under study. The influence of water quality parameters and biotope types on the habitat preference of the eel species was determined by Canonical Correspondence Analysis (CCA).

4.3 Results

4.3.1 Trophic levels, niche breadths, and diet overlaps

Among the three eel species examined, *Anguilla bengalensis* had the highest trophic level (TL), (3.7±0.08), followed by *A. bicolor* (3.4±0.30) and *A. mossambica* (3.1±0.4). Analysis of variance (ANOVA) demonstrated no significant differences ($F=2.42$, $df = 67$, $p=0.097$) in trophic levels among the species evaluated. Of the three species evaluated, *A. bicolor* had the highest diet breadth (0.38), followed by *A. mossambica* (0.34) and *A. bengalensis* (0.27), respectively (Table 3)

Table 3: Trophic Levels and Diet Breadth of Anguillid Eels of Rivers AGS and Ramisi

Species	**N**	**Trophic level (TL)**	**Diet breadth Index**
A. bengalensis	48	3.7±0.1	0.27
A. bicolor	11	3.4±0.3	0.38
A. mossambica	9	3.1±0.4	0.34

The highest diet overlap occurred between *A. bengalensis* and *A. bicolor* (0.65±0.5), while *A. mossambica* and *A. bicolor* had the least overlap (0.40±0.1) (Table 4).

Table 4: Diet overlap of anguillids of the Rivers AGS and Ramisi

Eel species	**Diet overlap**		
A. bengalensis	*A. bengalensis*		
A. bicolor	0.65±0.5	*A. bicolor*	
A. mossambica	0.56±0.2	0.40±0.1	*A. mossambica*

*Diet overlap of value 1 indicates increased sharing of diet items while 0 is reduction in diet sharing between species.

4.3.2 Stomach Diet Composition of the Eel Species of the Rivers AGS and Ramisi

Fish was the diet item that occurred in the stomachs of all the eel species. Fishes constituted the highest diet content (59.5%) in the stomachs of *A. bengalensis* and *A. bicolor* (37.4%) (Figure 13). Earthworms and insects occurred only in the stomachs of *A. mossambica*, with insects constituting the highest (42.4%) stomach content among the diet items consumed by the species. Plant material was only encountered in the stomachs of *A. bicolor*, while Pond snails and Bird remains were only in the stomachs of *A. bengalensis* (Figure 13)

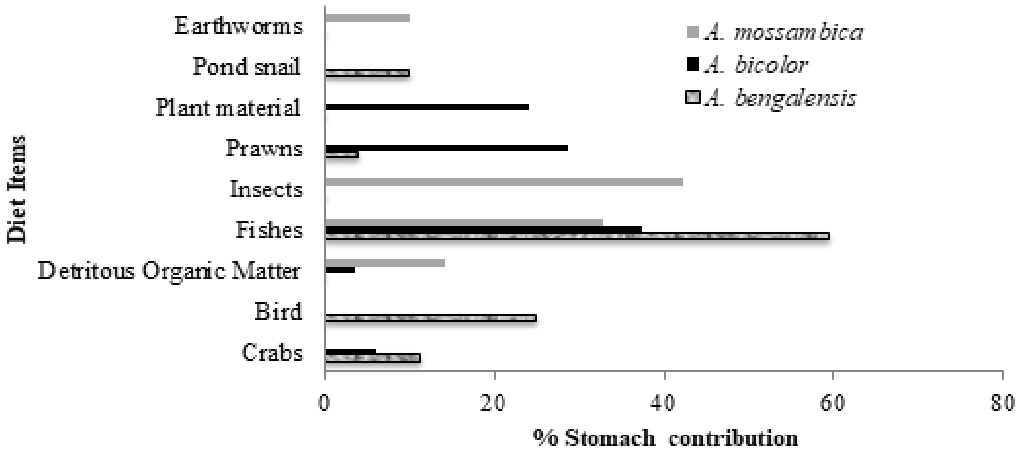


Figure 13: Stomach composition of anguillids in Rivers Ramisi and AGS

4.3.3 Diet Preference in the Stomachs of the Rivers AGS and Ramisi Eels

Nine diet items were successfully identified. Fishes were the most frequently occurring diet (0.9) in the eels' stomachs, followed by prawns (0.7) and plant material (0.5), while earthworms (0.03) and pond snails (0.02) had the least frequency, respectively (Figure 14).

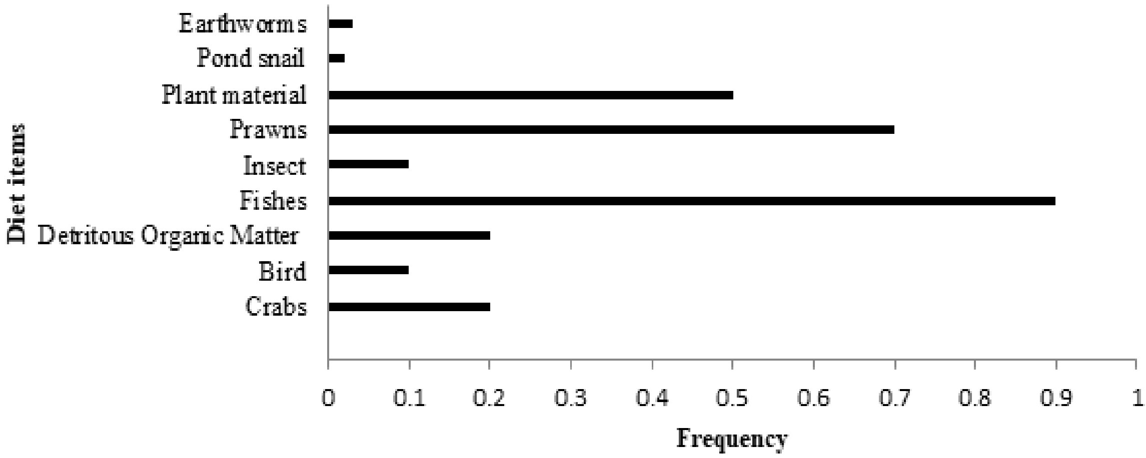


Figure 14: Frequency of Diet Items of the Eels.

4.3.4 Growth variables of the four anguillid eel species of the Rivers AGS and Ramisi

The weight of the eels ranged between 107.34g±51.21 (*A. mossambica*) and 406.70g±66.11 (*A. marmorata*) and was dependent on the species ($F = 5.13$, $df = 303$; $p<0.05$), while length did not differ significantly ($F = 2.55$, $df = 303$; $p = 0.06$) among the species (Table 5). The body depth (BD) was comparable ($F = 1.587$, $df = 303$; $p=0.19$), ranging from 2.0cm ±0.0 for (*A. bengalensis*) to 2.74cm±0.42 (*A. marmorata*). Generally, all the eel species demonstrated poor growth conditions ($Kn<1$) (Table 5), which statistically had no significant difference (Kruskal-Wallis test; $H=8$, $p>0.05$) among the eel species.

Table 5: Summary of Mean Total Lengths, Body depth, Weights, and Relative Condition factor (Kn) of the anguillid eels.

Species	n	BD (cm± SE)	L (cm±)	W (g)	Kn ± SE
A. bengalensis	229	2.00±0.10	37.20±1.10	157.82±16.92	0.86±0.42
A. bicolor	35	2.17±0.28	40.02±2.85	202.33±43.28	0.15±1.08
A. marmorata	15	2.74±0.42	47.05±4.38	406.70±66.11	0.18±1.65
A. mossambica	25	2.45 ±0.33	32.62±3.34	107.34±51.21	0.15±1.28

Note: *n* = Sample size; L= mean length, SE = Standard error; W =weight; Kn =condition factor

4.3.5 Length- weight relationships of the anguillid species

The results showed a positive allometric growth (b>3) for both *A. mossambica* (b=3.55) and *A. bicolor* (b=3.23) (Figure 15a and 15b), which corresponded to coefficient of determination; R^2=0.93 and R^2=0.81, respectively while *A. bengalensis* (b = 2.7) and *A. marmorata* (b = 2.8) exhibited negative allometric growth (Figure 15c and 15d), respectively. R^2 = Coefficient of determination (* b < 3 = negative allometric growth, b > 3, positive allometric growth).

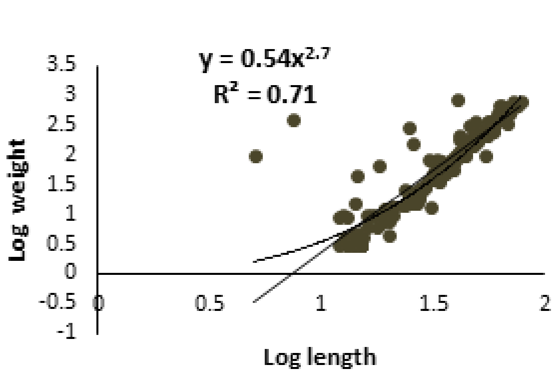


Figure 15a: Length-weight Relationship of *A. bengalensis* of the Rivers AGS and Ramisi

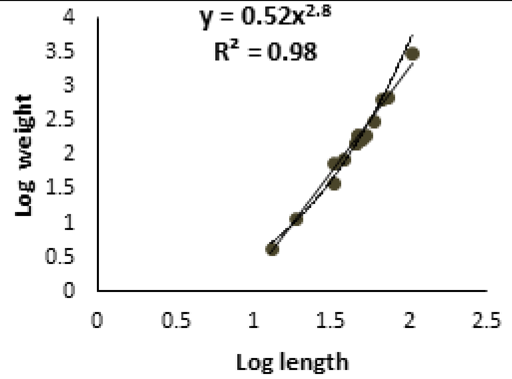


Figure 15b: Length-weight Relationship of *A. marmorata* of Rivers the AGS and Ramisi

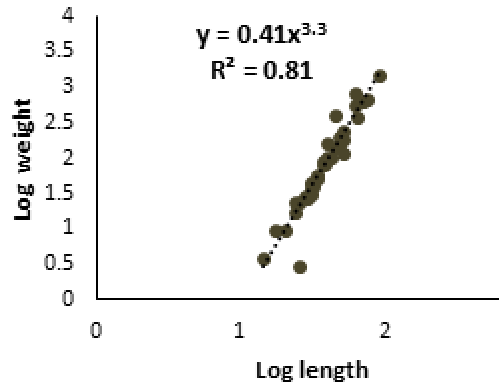


Figure 15c: Length-weight Relationship of *A. bicolor* of the Rivers AGS and Ramisi

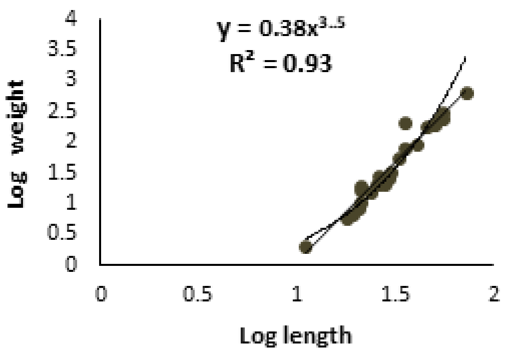


Figure 15d: Length-weight Relationship of *A. mossambica* of the Rivers AGS and Ramisi

4.3.6 Mean lengths, weights, and condition factors of anguillid life stages

Among all the life stages and species, Elvers recorded the lowest mean length, ranging from (16.85±2.62) for *A. bengalensis* to (22.83±8.55) for *A. bicolor*. Except for *A. marmorata,* silver eels were the longest among the life stages, with *A. bengalensis* leading at 54.57±4.94 cm, and Elvers of *A. marmorata* were the shortest at 12±14.81 (Table 6).

The highest condition factor (Kn) was recorded on the silver stage of *A. bicolor* (0.10±0.06) followed by the yellow stage of *A. bengalensis* (0.09±0.01), and the lowest in Elvers of *A. bicolor* (0.04±0.09). However, the condition factors were not statistically different at $p \leq 0.05$ among the life stages.

Table 6: Summary of Lengths, Weights, and Relative Condition Factors (Kn) of Anguillid Eels' Life Stages of the Rivers AGS and Ramisi

Species	**Stage**	**n**	**L (Length) (cm) ±SE**	**W(weight) (g) ±SE**	**Kn ± SE**
A. bengalensis	Elver	42	16.85±2.62	7.74±43.60	0.05±0.03
	Silver	9	54.57±4.94	339.56±82.23	0.07±0.05
	Yellow	225	39.83±1.08	174.66±17.10	0.09±0.01
A. bicolor	Elver	3	22.83±8.55	14.6±142.43	0.04±0.09
	Silver	6	51.75±6.05	424.17±100.71	0.10±0.06
	Yellow	37	39.49±2.85	181.39±47.48	0.06±0.03
A. marmorata	Elver	1	12±14.81	3.2±246.69	0.07±0.16
	Silver	1	36.7±14.81	85±246.69	0.06±0.16
	Yellow	33	50.55±4.11	462.48±68.42	0.07±0.04
A. mossambica	Elver	7	20.21±5.60	13.48±93.24	0.05±0.06
	Silver	1	44.5±14.81	180±246.69	0.07±0.16
	Yellow	27	37.02±3.59	141.718±59.83	0.06±0.04

Note: *n* = Sample size; L= mean length, SE = Standard error; Kn = relative condition factor.

4.3.7 Length/Weight relationships of the different life stages of the Anguillids species between Rivers

Elvers (b=4.80) and yellow eels (b=3.18) of *A. mossambica* (Figure18a and 18b, respectively), and the Elvers of *A. bengalensis* (b=4.01) (Figure 16a), showed a positive allometric growth while all the other growth stages of the different species showed negative allometric growth (b<3) (Figure 16b and c, Figure17a, b and c; Figure 18 a, b and c; Figure 19c). The length-weight relationship equations are demonstrated in the curves of Figures 16 to 19 (R^2 = Coefficient of determination (* b < 3 = negative allometric growth, b > 3, positive allometric growth).

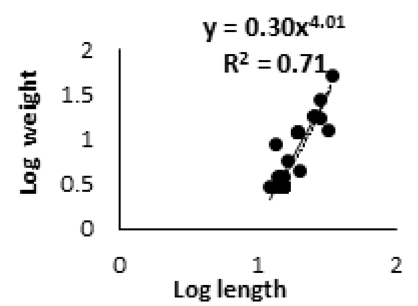


Figure 16a: L/W relationship of elvers of *A. bengalensis*

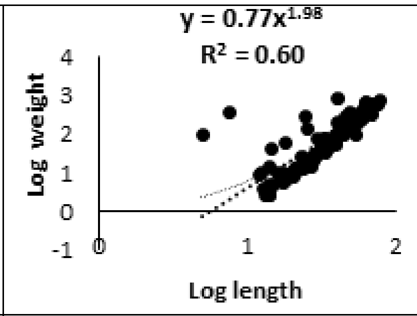


Figure 16b: L/W relationship of yellow eels of *A. bengalensis*

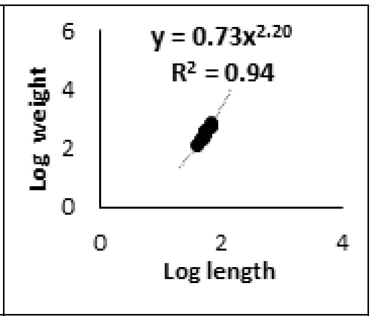


Figure 16c: L/W relationship of silver eels of *A. bengalensis*

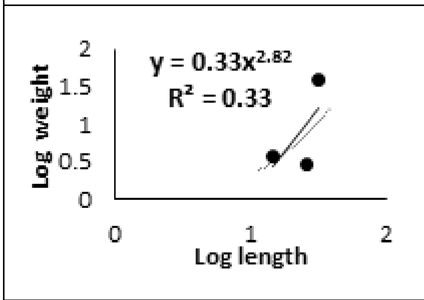


Figure 17a: L/W relationship of elvers of *A. bicolor*

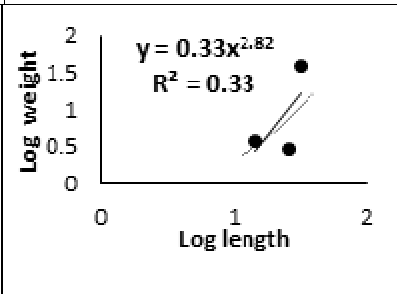


Figure 17b: L/W relationship of yellow eels of *A. bicolor*

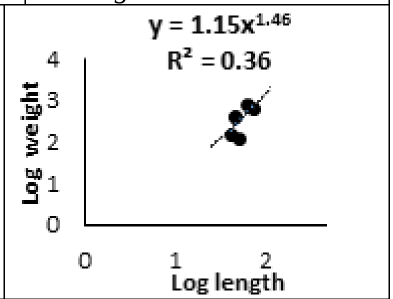


Figure 17c: L/W relationship of silver eels of *A. bicolor*

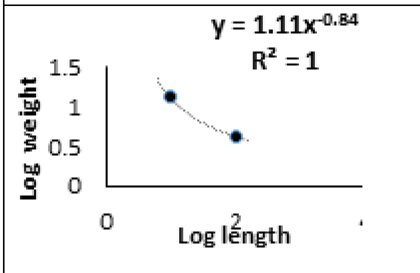


Figure 18a: L/W relationship of elvers of *A. marmorata*

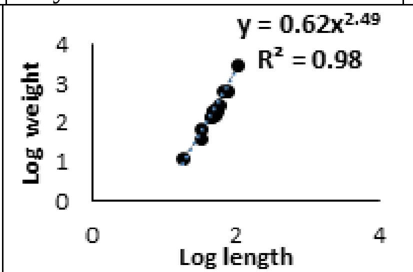


Figure 18b: L/W relationship of yellow eels of *A. marmorata*

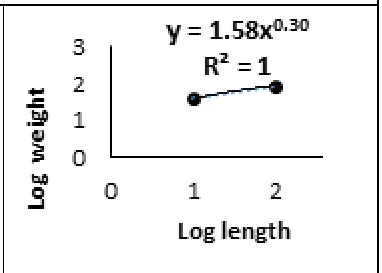


Figure 18c: L/W relationship of silver eels of *A. marmorata*

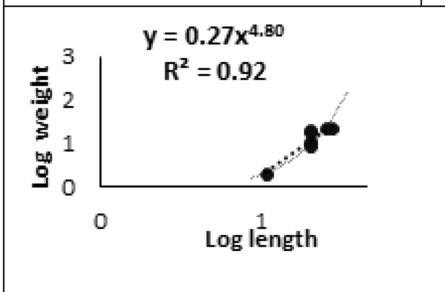


Figure 19a: L/W relationship of elvers of *A. mossambica*

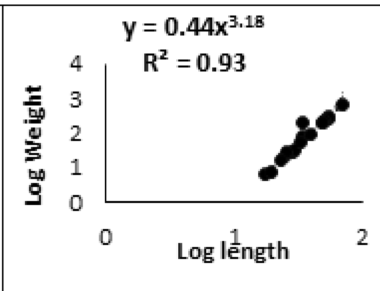


Figure 19b: L/W relationship of yellow eels of *A. mossambica*

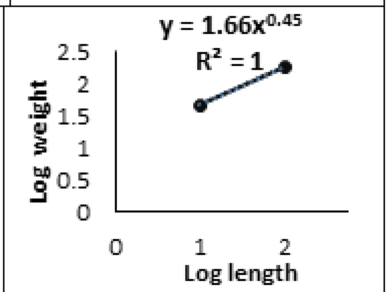


Figure 19c: L/W relationship of silver eels of *A. mossambica*

4.3.8 The Habitat preferences of the anguillid eels of the River Ramisi and the River AGS

Biotopes preference at the River Ramisi

The PCA results indicated that the most important contributors to variations in the occurrence of the four eel species across sampled habitat types and reaches along Ramisi River were the first two axes (PCA 1 and PCA 2), explaining 54% and 33%, respectively (Figure 20). Conversely, the third and fourth axes had a relatively smaller influence, contributing only 12% to the occurrences. *Anguilla bengalensis* and *Anguilla marmorata* were notably prevalent in pools and areas with vegetation cover, in the upper river reach. Conversely, these species exhibited a negative association with boulders and coarse substrates. On the other hand, *Anguilla mossambica* and *Anguilla bicolor* showed an affinity for fine substrate and displayed a negative association with riffles, boulders, and vegetation cover, in the lower reach of the River Ramisi.

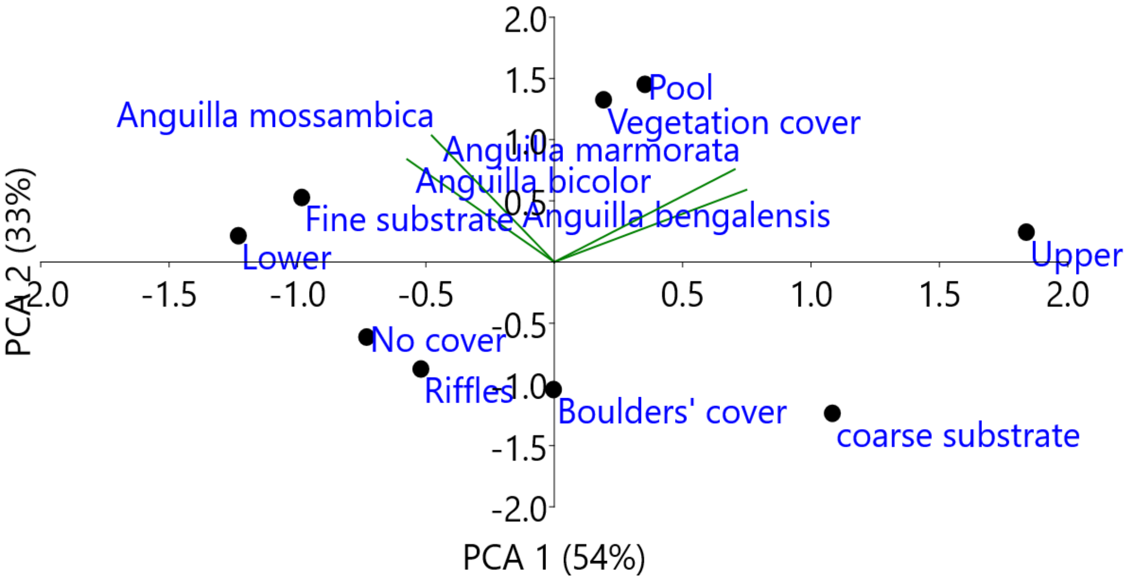


Figure 20: Principal Component Analysis (PCA) for Eel Species Assemblage among Habitat Types and Reaches at Ramisi River

The Biotopes preference at the River AGS

The PCA results revealed that the first two axes, PCA 1 and PCA 2, explained 66% and 28% of the variations in the distribution of the four eel species across different habitat types and reaches at AGS River sampling locations (Figure 21). PCA 1 indicated distinct habitat preferences among the eel species at the sampled locations. *A. bengalensis* and *A. marmorata* predominantly favoured pools and showed a negative association with vegetation cover and coarse substrate, in the upper river reach. Conversely, *A. bicolor* and *A. mossambica* exhibited negative correlations with fine substrate, boulders, and riffles, without cover, in the lower reach of the river.

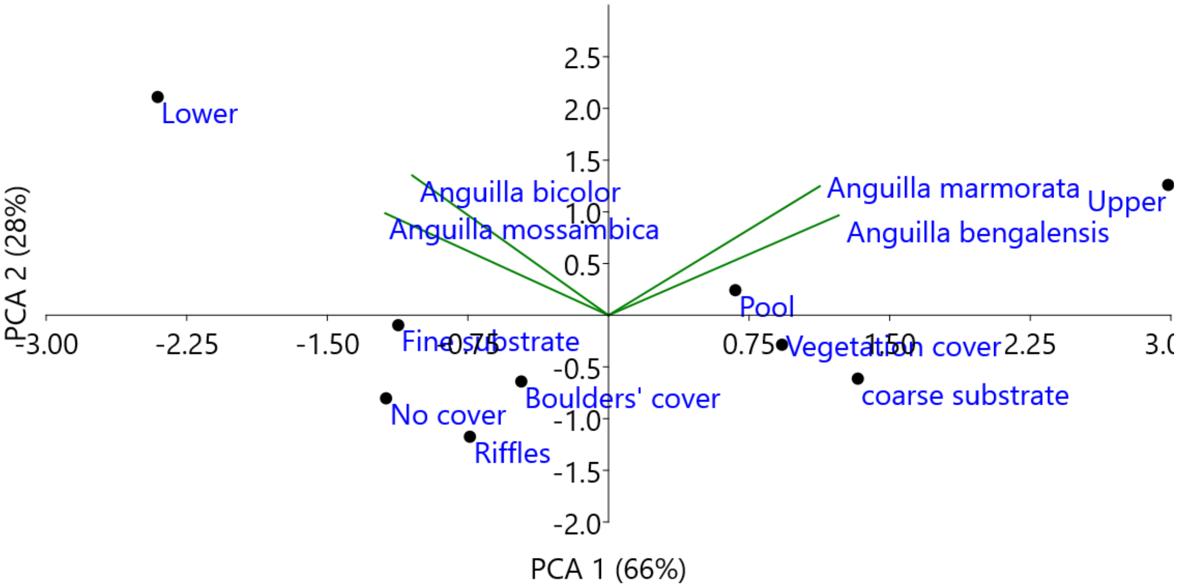


Figure 21: Principal Component Analysis (PCA) for Eel Species Assemblage among Habitat Types and Reaches at River AGS.

4.3.7 Water quality and hydrological parameters of the upper and lower reaches of Rivers AGS and Ramisi

Statistical analysis of water and hydrological parameters by post-hoc test (p≤0.05) to compare group means between river reaches (upper and lower), showed statistical differences in some of the parameters (Table 7). Total Dissolved Solids (TDS) and conductivity were statistically different between all river reaches. Salinity differed statistically between upper reaches of River AGS (0.2±0.28) and Upper River Ramisi (2.9±0.26), and between the corresponding lower reaches of the River AGS (0.6±0.26) and the lower Ramisi (3.6±0.34). Discharge was significantly different between all the river reaches. Depth was statistically different between the upper AGS and the upper Ramisi. Dissolved oxygen was significantly different between lower AGS and lower Ramisi. (Table 7). The mean temperature at the focal rivers was 27.8+1.1^{o}C but varied significantly between river reaches (F=, P<0.05). Mean river water temp was higher at the Ramisi (29.1+1.1) than AGS (26.4+1). Upper Ramisi had the highest temperature (31.4±1.02), which differed significantly from the upper AGS and the lower Ramisi but was comparable with the lower AGS.

Table 7: **Tukey's Range Analysis Results for Water Variables Across River Reaches in the AGS and Ramisi Rivers**

Sampling site/Reach	Temp. (°C)	TDS (mg/L)	Conductivity (S/m)	Salinity (ppt)	Depth (m)	Velocity (m/s)	Discharge (m^3/s)	Dissolved oxygen (mg/L)	pH
Upper AGS	24.3±1.05c	220.6±181.7a	400.0±394.21a	0.2±0.28b	1.9±0.12a	0.3±0.07a	46.3±12.5a	7.5±0.24a	7.6±0.08a
Upper Ramisi	31.4±1.02a	3488±598.91b	5878.6±406.8b	2.9±0.26a	0.2±0.10b	0.2±0.06a	0.2±10.04b	7.1±0.22a	7.4±0.07a
Lower AGS	28.6±1.01ab	720.0±593.4c	926.0±322.34c	0.6±0.26b	0.58±0.1b	0.3±0.06a	24.1±11.5c	7.1±0.27a	7.0±0.08b
Lower Ramisi	26.7±1.35bc	5836±803.56d	6624.4±545.8d	3.6±0.34a	0.3±0.10b	0.3±0.06a	0.9±10.04b	4.4±0.28b	6.8±0.10b

Variables with different letters (a, b, c, d) in the same columns indicate significant differences ($p \leq 0.05$)

4.3.8 Influence of water quality and hydrological features on habitat preference of Anguillids of River AGS and River Ramisi

Canonical correspondence analysis (CCA) revealed that the first two axes (CCA 1 and CCA 2) accounted for 69.6% and 23% of the variation in the occurrence of the four eel species, respectively (Figure 22). The third axis (CCA 3) only contributed to 8% of the occurrences. *A. mossambica* at was positively influenced by TDS (5836±803.56), conductivity (6624.4±545.8), and salinity (3.6±0.34) while *A. marmorata* was largely influenced by discharge (46.3±12.5 (m^3/s), dissolved oxygen (7.5 mg/L ±0.24), pH (7.6±0.08) and temperature (24.3 ^{o} C ±1.05). *A. bengalensis* at the upstream location of River Ramisi was negatively influenced by TDS and salinity, while *A. bengalensis* occurring at the upstream location of River AGS was negatively influenced by pH, dissolved oxygen, and discharge.

(Figure 4.23). *A. bicolor*, which was negatively influenced by discharge (0.9±10.04 m^3/s) and TDS (5836±803.56 mg/L). However, the results demonstrated relative influence but no statistical difference in the influence of environmental conditions on habitat preference by the eel species in CCA 1 (p=0.28, Eigen value=0.32) and CCA 2 (p=0.82, Eigen value=0.10).

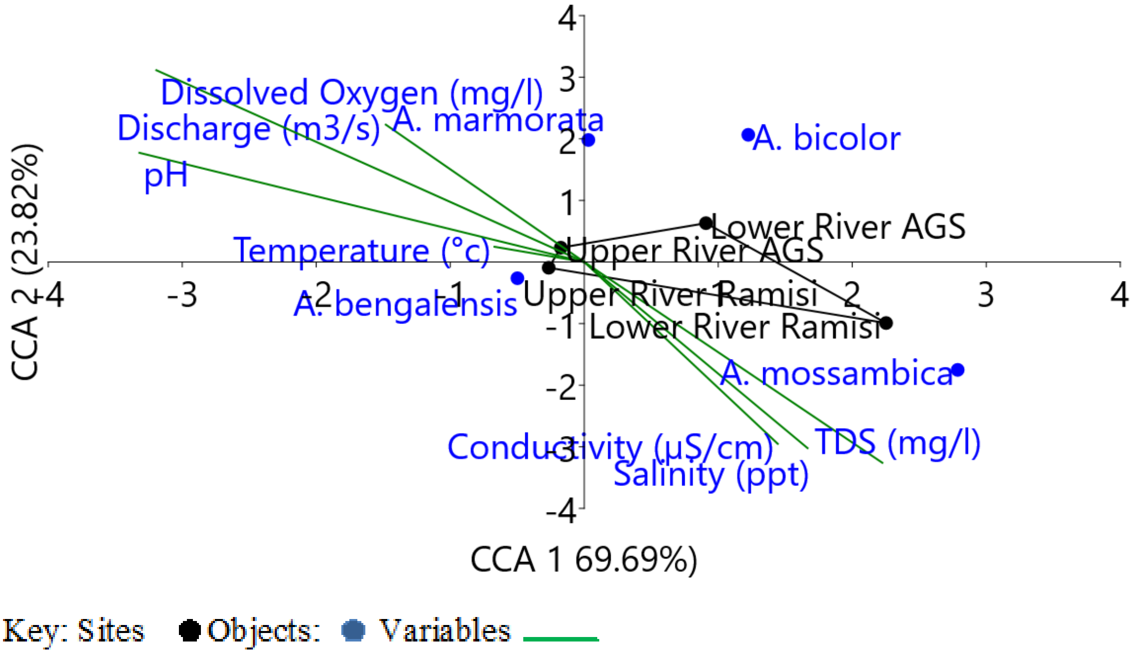


Figure 22: Canonical Correspondence Analysis (CCA) on the Influence of Environmental Variables on Eels' Habitat Preference, among Sampling Locations

4.4 Discussion

The anguillids in this study fed on invertebrates (shrimps, crabs, insects, worms) and vertebrates (fishes and birds) diet items. Among these, fish emerged as the predominant dietary component, particularly notable in *A. bengalensis* and *A. bicolor*; however, Insects constituted the highest stomach content of *Anguilla mossambica*. In similar studies fish also dominated the diet component of *Anguilla anguilla* in similar studies by Radke and Eckmann (1996) and Dörner and Benndorf (2003). Another study by Jellyman (1989) in Lake Pounui, New Zealand, confirmed the consumption of similar organisms as in this study, among them 'Aves' (birds) by *Anguilla australis* and *Anguilla dieffenbachii*. Jellyman (1989) deduced that the eels consumed foods of terrestrial origin, such as birds during the flood periods, compared to non-flood periods. Although yellow and silver eels have been established as carnivorous, high diet of plant material was retrieved from the stomachs of *Anguilla bicolor*. The high diet component of plant material in the stomachs of *A. bicolor* could be a result of the eel feeding on large numbers of Prawns. Prawns are known to feed on a variety of plant material (Lima *et al*., 2014), which could accumulate in the stomachs of *A. bicolor*. Similarly, in the studies by Jellyman (1989), vegetable matter and substrate (mud and sand) were present in the stomachs of *Anguilla australis* and *Anguilla dieffenbachii,* which was attributed to being ingested accidentally with prey species. The presence of pond snail, earthworms, crabs, and detritus organic matter as diet items of anguillid eels was also reported in studies carried out on *Anguilla anguilla* by Denoncourt and Stauffer (1993).

It was confirmed that anguillids of River AGS and River Ramisi were third trophic level fish, ranging between 3.1 and 3.7. The trophic level range in the current study falls within the range (between 3.5 and 4.1) obtained from the studies of Froese *et al*. (2012), on four temperate eels, namely *Anguilla rostrata, A. anguilla, A. japonica,* and *A. australis*. The trophic levels of *Anguilla bengalensis* (3.7) in the current study correlate with those of *Anguilla rostrata* (3.7) (Froese *et al*., 2012). The presence of anguillid eels as secondary consumers in Rivers AGS and Ramisi is critical for ecological balancing through population control of primary consumers. This trophic position is ecologically important in offering energy transfer to higher trophic level creatures for ecosystem health and sustainability. Moreover, the trophic position makes them important in the assessment of pollution in the ecosystem, through studying the bioaccumulation of pollutant elements in eel organs.

The diet breadth for anguillids of River AGS and River Ramisi, a range of 0.27 – 0.38, was an indication that the eels were utilizing approximately 27 to 38 % of the available resources in their environment. This value represents the proportion of different types of diet resources included in the anguillids' diet relative to the total number of potential resources available. This low diet breadth is linked to diet specialization on food items identified in eels' stomachs in this study. Due to the narrow diet breadth (<0.5) revealed in the study the eels have been considered 'specialists' that favour specific diet resources over others, in the ecosystem. Owing to their higher proficiency to acquire preferred resources, the eel species have restricted food preferences, thus, narrow dietary niches that overlap with other specialists having more generalized feeding patterns). Jellyman (1989) in a study on *Anguilla anguilla* concluded that individual eels were normally very selective in their diet and this was also found to be the case in the current study.

The species under study generally overlapped in diet consumption, with the highest overlap occurring between *Anguilla bicolor* and *Anguilla bengalensis*. These results agree with those of Arai and Abdul (2017), on research carried out to assess the niche breadth and overlap between *Anguilla bicolor* and *Anguilla bengalensis*, which found out that, both species coexisted in most of habitats of Penang Island (Malaysia), and shared the same niche breadth, used the same demersal habitats and foraged for the same prey.

The anguillid species of Rivers AGS and Ramisi had poor growth conditions. In the process of growth, *A. bengalensis* and *A. marmorata* became lighter with increasing length (Negative allometry) while *A. bicolor* and *A. mossambica* became heavier with increasing length (Positive allometry). The poor growth condition (Kn) demonstrated by *A. marmorata, A. bicolor,* and *A. mossambica* in the current study was comparable to that reported on *Anguilla anguilla* by Milošević *et al*. (2022). The *b* values were within the same range as in the current study and a study by Sidqi *et al*. (2018) on *A. bicolor* in the Banda Aceh waters of Indonesia. Froese (2006), through a meta-analysis of data involving the length-weight relationship of 1773 species, indicated that 90% of the intercept values ranged between 0.001 and 0.05. In our study, the intercepts disagreed with these values and ranged between 0.38 and 0.54. Regarding the *b* values, our results ranged between 2.7 and 3.5, whereas Froese (2006) reported that *b* values for teleost fish should fall within the expected range of 2.5 and 3.5. The lower limit in this study slightly, fell slightly short of agreement with the work done by Froese (2006) by a value of 0.2. The growth analysis of the different life stages of the anguillid species revealed generally poor growth conditions (Kn<1) in all life stages of the different species under study. The elver life stage of *A. bengalensis* and *A. mossambica*, and the yellow eel stage of *A. mossambica,* showed positive allometry. This meant that the elvers of *A. bengalensis* and *A. mossambica*, and the yellow eels of *A. mossambica,* became heavier-bodied as they grew longer. Muscle development in the elvers and fat accumulation in the yellow eels were the probable reasons for the positive allometry.

The current study reports that the marbled eels (*A. bengalensis and A. marmorata*) were mainly found in the upper River AGS and River Ramisi, whereas plain-coloured *A. bicolor* was absent in the upper AGS. This scenario in the current study agrees with the findings of Hsu *et al*. (2020) which revealed that marbled eels were more abundant at the upper River reaches than the plain coloured eels which occur in mid to lower reaches of Rivers. The scenario to inhabit either the upper or lower river reaches by the eels was attributed to the need to camouflage from predators and in search for food. The lower reaches are suitable habitats for the plain coloured eels since siltation from upstream and sand deposition from ocean tides provide plain suitable camouflage habitats, whereas the marbled eels prefer habitats with fallen leaves and vegetation found mostly at the upstream of Rivers (Hsu *et al*., 2020). Although *A.*

bicolor (plain coloured) was observed in the upper reaches of the River Ramisi, contradicting the findings of Hsu *et al*. (2020), this observation aligns with the research by Arai *et al*. (2020). Arai (2020) noted that *A. bicolor* was present in both upstream and downstream areas of rivers but was more plentiful in the midstream and downstream sections, but less frequently found in upstream regions. Moreover, as was the case in this study, Arai *et al*. (2020) revealed that *A. bengalensis* was found in the upper reaches of Rivers and rarely in mid and downstream. Nguyen (2018) and Hsu (2019) reported that *Anguilla marmorata* inhabited freshwater areas more than brackish water areas, which was found to be the case in this study. However, the current study was contrary to that by Tsukamoto *et al*. (1998), who found that *Anguilla mossambica* was distributed at the lower reaches of Rivers and the estuaries, whereas this study found *Anguilla mossambica* inhabited the upper and lower reaches of Rivers AGS and Ramisi.

Although the physical-chemical parameters showed significant differences among some variables, they did not influence the habitat preference of the eels. The different biotopes did not have significant differences, giving an insinuation, as reported by Jellyman (2022), that anguillid eels have a special set of behavioural and morphological attributes that have enabled them to become a globally widespread and successful genus inhabiting diverse niches. These attributes include physiological and behavioural adaptability to varying aquatic environments and plasticity to extreme environmental situations (Arai, 2020; Jellyman, 2022). Therefore, the non-significant difference in the influence of habitat preference by, biotopes, physicochemical and hydrological parameters in the current study would be attributed to the plasticity of eels to extreme environmental conditions as reported by Arai (2020) and Jellyman (2022).

4.5 Conclusion

In conclusion, this study highlights significant ecological insights into the dietary habits, niche dynamics, and habitat preferences of anguillid eels in Rivers AGS and Ramisi. The findings indicate that fish constitute a major portion of the diet across species, confirming a strong reliance on piscivory. However, variations in niche breadth and trophic levels among species suggest differing ecological roles and dietary strategies, contributing to ecosystem stability. The high niche overlap observed between *Anguilla bengalensis* and *Anguilla bicolor* suggests potential competition for resources, emphasizing the importance of understanding interspecific interactions in riverine ecosystems.

Contrary to the null hypothesis that there is no difference in the ecological niches of Anguillid eels between River AGS and River Ramisi, the study reveals habitat- and species-specific variations. Differences in relative condition factors, length-weight relationships, and habitat preferences indicate species-specific responses to local environmental conditions, such as substrate type, water quality, and resource availability. The observed river-specific biotope selection further suggests that eels exhibit nuanced adaptations to habitat heterogeneity, reinforcing the need for location-specific conservation and management strategies.

Additionally, positive associations between water quality parameters and species presence highlight the critical role of environmental conditions in shaping eel distribution and abundance. These findings underscore the importance of preserving habitat diversity and maintaining favourable ecological conditions to support sustainable eel populations. Future research should focus on long-term monitoring of niche dynamics and environmental influences to enhance conservation strategies and ensure the resilience of Anguillid eel populations in Kenya's river systems.

CHAPTER FIVE

ECOLOGICAL RISK ASSESSMENT OF ANTHROPOGENIC STRESSORS ON CATADROMOUS ANGUILLID EELS IN RIVER AGS AND RIVER RAMISI

Abstract

This study assessed the ecological risks to anguillid eels in the east-flowing rivers of Kenya, specifically focusing on River Ramisi and River AGS. Using the Relative Risk Assessment (RRA) model, the research evaluates the impact of various environmental stressors on different life stages of the eels' larval recruitment, yellow eel maintenance, and silver eel escapement across natural, present, and future scenarios. The findings reveal that while River Ramisi, with an overall habitat integrity score of 92.3%, remains largely natural with minimal anthropogenic impacts, River AGS, with a score of 56.6%, exhibits significant degradation (ANOVA; $p = 0.04$, $F = 6.49$). In the natural scenario, both rivers displayed low-risk scores across all life stages, but present and future scenarios show a marked increase in risk, particularly in River AGS. The yellow eel maintenance and silver eel escapement phases are the most vulnerable to habitat degradation, water quality issues, and migration barriers, with the risk escalating in future scenarios due to the anticipated intensification of anthropogenic activities. The study underscores the critical need for targeted conservation strategies to mitigate risks and preserve eel populations, particularly in the more disturbed River AGS. The findings also contribute to the understanding of how environmental changes influence the life stages of eels, providing a foundation for future research and management interventions.

5.1 Introduction

5.1.1 Vulnerability of Anguillid eels to human impacts and the role of Relative Risk Assessment (RRA)

Anguillid eels face significant vulnerability due to human impacts across their life cycle, which can disrupt their complex migratory patterns and survival (Wu & Zhao, 2024). Human activities, such as habitat destruction, pollution, overfishing, and the construction of barriers like dams, have profound effects on eel populations. These impacts can alter key environmental parameters, such as water quality, temperature, and flow regimes, which are critical for the eels' various life stages (Tamario *et al*., 2019). The application of Relative Risk Assessment (RRA) is instrumental in understanding and mitigating these effects. RRA helps identify and quantify the risks associated with human-induced stressors and assesses the eels' capacity to withstand and recover from these disturbances (Landis &Thomas, 2009). By evaluating the potential impacts on each life stage (glass eel, yellow eel, and silver eel) RRA provides valuable insights into the specific vulnerabilities and critical areas for conservation efforts. This approach enables the development of targeted management strategies aimed at preserving eel populations and ensuring their long-term sustainability in the face of ongoing human pressures (Dudgeon *et al*., 2019; Froehlicher *et al*., 2023).

5.1.2 Importance of studying anguillid eels

The study of anguillid eels is of critical importance due to their ecological, economic, and cultural significance (Drouineau, 2018). Eels serve as an important source of food and livelihood for many communities around the world, particularly in regions where eel fishing has been a traditional practice for centuries (Shanmughan *et al*., 2022). The decline in eel populations due to environmental stressors not only threatens the biodiversity of aquatic ecosystems but also jeopardizes the livelihoods of those who depend on eel fisheries. Additionally, eels play a vital role in aquatic food webs, acting as both predators and prey, and their decline can have cascading effects on other species and the overall health of aquatic ecosystems (Righton *et al*., 2021). Understanding the factors that contribute to eel population declines through RRA is essential for developing effective conservation strategies that protect both the species and the human communities that rely on them.

5.1.3 Eels' life stages and their vulnerability to environmental stressors

Anguillid eels exhibit complex life cycles that involve migration between freshwater and marine environments (Arai, 2014). Each life stage, glass eel, yellow eel, and silver eel face unique environmental challenges that can be assessed using RRA. Glass Eels represent the early juvenile stage of anguillid eels and are particularly susceptible to environmental stressors during their migration from the ocean to freshwater habitats (Jacoby *et al*., 2015). This stage is marked by a transparent, larval form that is highly vulnerable to changes in water quality and habitat availability. The journey from the Sea to estuaries and freshwater systems exposes glass eels to varying levels of pollutants, changes in salinity, and habitat degradation, all of which can significantly impact their survival rates (Bolliet *et al*., 2017; Martínez-Gómez *et al*., 2023). Additionally, their delicate nature makes them prone to disturbances from human activities, such as fishing and dam construction, which can further exacerbate their vulnerability during this critical life stage (Denis *et al*., 2024; Maes *et al*., 2006).

Yellow Eels are in the growth stage, where eels have migrated into freshwater or brackish environments and have developed their characteristic yellow-brown coloration (Arai, 2014). This stage can last several years and is crucial for their growth and development. Yellow eels face different environmental stressors compared to their earlier stage; they are affected by factors such as water temperature, oxygen levels, and habitat structure (Wright *et al*., 2022). Variations in these parameters can influence their feeding efficiency, growth rates, and overall health. For instance, higher water temperatures and reduced oxygen levels can lead to decreased food availability and lower growth rates, impacting their overall fitness and preparedness for the next life stage (Cairns *et al*., 2021; Ovidio *et al*., 2013). Habitat modifications and pollution can also alter their living conditions, further stressing these eels (Lambert & D'Adamo, 2011; Sarre *et al*., 2015).

Silver Eels are in the final life stage before migration back to the ocean to spawn. At this stage, eels undergo significant physiological changes, including the development of a silvered body and increased fat reserves, to prepare for the long migration and spawning in the Sea (Arai & Chino, 2018). Silver eels encounter numerous obstacles, such as barriers to migration, including dams and weirs, and changes in ocean currents driven by climate change. These factors can hinder their ability to reach spawning grounds and successfully reproduce (Durif *et al*., 2022; Miller, 2023). Additionally, alterations in oceanic conditions, such as shifts in temperature and currents, can affect their migration routes and timing, potentially leading to decreased reproductive success and population declines (Bevacqua *et al*., 2014; Haro *et al*., 2014).

5.2 Habitat modifiers and Anguillid eels' ecology

Habitat quality and complexity are essential drivers of freshwater biodiversity, particularly for species with complex life cycles such as Anguillid eels. These catadromous fishes rely on a mosaic of interconnected habitats throughout their continental phase, with specific environmental conditions influencing their growth, survival, and recruitment (Tesch, 2003). Various habitat modifiers, natural and anthropogenic, directly or indirectly alter physical structure, hydrology, water quality, and biological productivity within riverine systems, thereby shaping the ecological niches eels occupy. Habitat modifiers can act as either enhancers or stressors. Structural features such as submerged woody debris, undercut banks, vegetated margins, and variable substrates promote habitat heterogeneity and provide critical functions, including refuge, foraging grounds, and thermal regulation (Sullivan *et al*., 2006). These attributes are particularly important for the yellow eel stage, during which individuals exhibit strong site fidelity and depend on cover to avoid predation and conserve energy.

Conversely, anthropogenic habitat alterations, including channelization, pollution, sedimentation, and vegetation removal, reduce complexity and disrupt ecosystem function. Simplified or degraded habitats limit eel access to essential microhabitats and can alter behavioural patterns, condition factors, and population dynamics (Baras *et al*., 1998). Furthermore, habitat continuity is crucial for eels, as their life history requires migration between marine and freshwater environments. Modifiers that create physical barriers or fragmented habitats compromise longitudinal connectivity, leading to recruitment failure or increased mortality during upstream and downstream movement (Béguer-Pon *et al*., 2015).

Habitat modifiers also influence ecological processes such as organic matter retention, prey availability, and oxygen dynamics. For example, areas with intact habitat features often support higher benthic invertebrate diversity and productivity, thereby sustaining the diverse diets of eels across developmental stages (Sugeha *et al*., 2001). In degraded systems, the decline in prey diversity, coupled with elevated pollutants and reduced oxygen levels, affects eel physiology and feeding behaviour, potentially reducing growth rates and reproductive fitness.

Water temperature and flow regimes, often shaped by surrounding land use and climate modifiers, also affect habitat suitability for eels. Thermal extremes and unnatural flow fluctuations may push individuals beyond their physiological limits, particularly for glass eels and elvers during initial freshwater entry (Acou *et al*., 2008). Additionally, changes in sediment dynamics—whether through erosion or fine sediment deposition, can alter substrate composition, affecting eel burrowing and reducing interstitial spaces critical for shelter and ambush hunting.

Overall, habitat modifiers play a foundational role in defining the ecological conditions under which Anguillid eels thrive or decline. The cumulative and often synergistic impact of these modifiers determines the resilience of eel populations, especially in systems experiencing increasing anthropogenic pressure. Sustainable management and restoration of riverine habitats thus require a comprehensive understanding of how both in-stream and adjacent landscape elements influence eel ecology across spatial and temporal scales.

5.2.1 Impact of water abstraction and quality on anguillid eel populations

Water abstraction, the process of extracting water from natural sources for agricultural, industrial, and domestic purposes, has been identified as a significant anthropogenic stressor affecting freshwater ecosystems (Grill *et al*., 2019). This activity alters hydrological regimes, disrupts habitat integrity, and influences the survival and recruitment of aquatic species, including catadromous anguillid eels. In rivers such as the Athi-Galana-Sabaki and Ramisi, extensive water abstraction for irrigation, municipal supply, and hydropower generation poses a severe ecological threat to eel populations by modifying essential habitat conditions (McClain *et al*., 2020).

Water abstraction leads to significant hydrological alterations, such as reduced river discharge, increased water temperature, and altered sediment transport (Poff & Zimmerman, 2017). Anguillid eels, which depend on stable flow regimes for migration and feeding, are particularly vulnerable to these changes. Reduced water levels can limit access to critical habitats, disrupt connectivity between freshwater and marine environments, and increase predation risks due to habitat fragmentation (Dekker, 2019). Additionally, lower water volumes lead to decreased dissolved oxygen levels, which can cause physiological stress and reduced growth rates in eels (Bardonnet & Acolas, 2021).

In Kenyan rivers such as the AGS and Ramisi, seasonal variations exacerbate the effects of water abstraction. During dry seasons, excessive water withdrawals intensify low-flow conditions, leading to habitat desiccation and increased competition among aquatic organisms (Masese *et al*., 2018). These conditions negatively impact eel recruitment and growth, as juveniles require stable freshwater environments before migrating to marine breeding grounds (Schmidt *et al*., 2020).

Catadromous eels, such as *Anguilla bengalensis*, *Anguilla mossambica*, and *Anguilla bicolor*, rely on unimpeded river flow for their upstream and downstream migrations (Jacoby *et al*., 2015). Water abstraction-related reductions in river discharge create barriers to migration by lowering water levels at critical points such as waterfalls, rapids, and estuarine zones. In extreme cases, over-abstraction may result in completely dry riverbeds, cutting off migration routes entirely (Limburg & Waldman, 2018). Moreover, abstraction infrastructure such as pumps, diversion channels, and reservoirs can physically harm eels, leading to injury or mortality. Studies have shown that juvenile eels attempting to navigate through water intake structures face significant mortality risks due to impingement and entrainment (Friedland *et al*., 2021). In the AGS and Ramisi Rivers, unregulated abstraction points may contribute to similar risks, exacerbating population declines and threatening the long-term viability of eel stocks.

Water abstraction affects not only quantity but also quality. Reduced flow volumes concentrate pollutants, increase water temperatures, and promote algal blooms, all of which negatively impact eel health (Grill *et al*., 2019). In Kenya, agricultural and industrial activities along river catchments introduce pesticides, heavy metals, and organic pollutants, further exacerbating water quality issues (Masese *et al*., 2018). Eels are particularly susceptible to bioaccumulation of contaminants due to their long lifespan and position in the food chain (Limburg & Waldman, 2018). Declining water quality can impair physiological functions such as respiration, osmoregulation, and immune response in eels. Studies have demonstrated that exposure to pollutants in low-flow environments increases stress hormone levels and reduces reproductive success in catadromous eels (Friedland *et al*., 2021). Given the importance of the River AGS and the River Ramisi for eel populations in Kenya, ensuring adequate water quality is crucial for maintaining healthy stocks.

The impact of water abstraction extends beyond direct effects on eel populations to their prey base and overall ecosystem structure. Reduced water levels alter sediment transport and nutrient cycling, leading to declines in macroinvertebrate populations, which constitute a major food source for eels (Benejam *et al*., 2016). Reduced water availability can alter habitat structure, leading to changes in species composition and a decline in the abundance of key prey such as insects, worms, fish, and crustaceans (Masese *et al*., 2021). Eels are opportunistic feeders whose diet is influenced by habitat conditions. In the AGS and Ramisi rivers, habitat degradation due to water abstraction may force eels to shift their dietary preferences, potentially leading to nutritional stress and reduced fitness (Bardonnet & Acolas, 2021). Such disruptions to food web interactions lead to cascading effects on the overall riverine ecosystem, affecting biodiversity and ecosystem stability (Schmidt *et al*., 2020).

Ecological risks associated with water abstraction require a multi-faceted approach, incorporating sustainable water management, habitat conservation, and regulatory enforcement. Environmental flow assessments can help determine minimum flow requirements necessary to sustain eel populations while balancing human water needs (Poff & Zimmerman, 2017). Implementing water allocation policies that prioritize ecological integrity can mitigate the adverse effects of abstraction on riverine habitats (Jacoby *et al*., 2015). Additionally, integrating fish-friendly infrastructure, such as eel ladders and bypass channels, can enhance migration success and reduce mortality at water abstraction sites (Dekker, 2019). Community-based conservation initiatives involving local stakeholders in water governance can also contribute to sustainable water management in the AGS and Ramisi rivers (Masese *et al*., 2021).

5.2.2 Impact of water quality on anguillid eel populations

Water quality is a critical determinant of aquatic ecosystem health and directly influences the survival, growth, and recruitment of catadromous Anguillid eels in freshwater systems (Matthews *et al*., 2019). The ecological risk assessment of anthropogenic stressors, such as pollution, sedimentation, and habitat degradation, necessitates an understanding of key water quality parameters, including temperature, dissolved oxygen (DO), pH, turbidity, and contaminants (Friedland *et al*., 2020). In River Athi-Galana-Sabaki (AGS) and River Ramisi, increasing human activities have altered water quality, posing significant risks to Anguillid eel populations and their habitats (Koei, 2013).

Water temperature plays a crucial role in regulating the physiological processes of Anguillid eels, including metabolism, growth, and migration (Tesch, 2003). Ideal temperature ranges for Anguillid eels vary by species, but fluctuations due to deforestation, climate change, and industrial effluents can disrupt their life cycles (Morrison & Secor, 2016). Additionally, dissolved oxygen (DO) is essential for eel survival, as low oxygen concentrations can lead to stress, reduced feeding efficiency, and increased mortality (Arai, 2016). Anthropogenic stressors, such as organic pollution from agricultural runoff and sewage discharge, contribute to hypoxic conditions, reducing habitat suitability for eels in AGS and Ramisi rivers (Jacoby *et al*., 2015).

The pH of freshwater environments is a critical factor influencing the physiological and biochemical processes of aquatic organisms. Anguillid eels thrive in a pH range of 6.5 to 8.0, but acidification or alkalization due to industrial discharge and acid rain can negatively impact their survival and growth (Daverat *et al*., 2012). Changes in pH can also affect the bioavailability of toxic substances such as heavy metals, which accumulate in eel tissues, causing sub lethal effects that impair reproductive success (Corsi *et al*., 2018). In AGS and Ramisi, the encroachment of human settlements and increased industrial activities threaten the natural pH balance, leading to ecological consequences for eel populations.

Turbidity and sedimentation affect the ecological integrity of freshwater habitats, influencing the feeding behaviour and recruitment success of Anguillid eels (Pratt & Threader, 2011). Increased sediment loads from deforestation, agriculture, and urbanization in AGS and Ramisi rivers degrade eel habitats by smothering benthic organisms, reducing prey availability, and obstructing migration routes (Gollock *et al*., 2016). High turbidity levels also reduce light penetration, disrupting primary production and altering the food web dynamics essential for sustaining eel populations (Maes *et al*., 2007).

Industrial and agricultural pollutants significantly impact the health and survival of Anguillid eels in freshwater ecosystems. Heavy metals such as mercury (Hg), lead (Pb), and cadmium (Cd) accumulate in eel tissues, leading to bioaccumulation and biomagnification, which impair neurological and reproductive functions (Pierron *et al*., 2013). Additionally, persistent organic pollutants (POPs), including pesticides, polychlorinated biphenyls (PCBs), and pharmaceuticals, pose long-term ecological risks by altering endocrine function and reducing recruitment success (MA & Belpaire, 2010).

In AGS and Ramisi rivers, increased agricultural runoff introduces nitrogen and phosphorus into the water, resulting in eutrophication, algal blooms, and hypoxic conditions that further threaten eel habitats (Njiru *et al*., 2021). Pesticide residues from sugarcane and horticultural farms contribute to endocrine disruption in eels, affecting their migration and reproductive cycles (Schmidt *et al*., 2015). The cumulative effects of these contaminants underscore the need for stringent water quality monitoring and mitigation strategies to safeguard Anguillid eel populations.

Reduced water quality can lead to physiological stress, impaired immune function, and increased susceptibility to diseases, thereby affecting eel population dynamics (Silberschneider *et al*., 2001). Furthermore, the disruption of olfactory cues due to chemical pollutants hinders the ability of eels to locate suitable habitats and spawning grounds (Cresci *et al*., 2019). Research has shown that Anguillid eels in polluted rivers exhibit slower growth rates and lower body condition factors compared to those in pristine environments (Belpaire *et al*., 2016). This decline in health and fitness directly affects the sustainability of eel populations, given their complex life cycle that involves long-distance migration to the ocean for spawning (Righton *et al*., 2013). In the context of AGS and Ramisi rivers, effective water quality management is essential to mitigate the negative impacts of anthropogenic stressors on eel populations. To address the ecological risks associated with declining water quality, integrated water resource management (IWRM) approaches should be implemented to regulate pollution sources and improve aquatic habitat conditions (European Eel Regulation, 2007). Riparian buffer zones, afforestation programs, and wastewater treatment facilities can help reduce sedimentation, nutrient loading, and chemical contamination in AGS and Ramisi rivers (Dekker, 2003). Additionally, community-based conservation efforts involving local stakeholders can enhance sustainable fishing practices and promote awareness of eel conservation needs (Harrison *et al*., 2014).

Furthermore, regular monitoring of water quality parameters and pollutant levels is crucial for assessing the health of freshwater ecosystems and implementing timely mitigation measures (Jacoby & Gollock, 2014). The integration of eco-toxicological assessments with population studies can provide a comprehensive understanding of the impact of water quality degradation on Anguillid eels and inform adaptive management strategies (Van den Thillart *et al*., 2009).

5.2.3 Modification of the River bed and channel

Riverbed modification is a significant anthropogenic stressor that alters the physical and ecological characteristics of freshwater ecosystems, impacting biodiversity, hydrology, and species habitat availability. In the context of catadromous anguillid eels in the Athi-Galana-Sabaki (AGS) and Ramisi Rivers, bed modification due to human activities such as sand mining, dam construction, and channelization poses severe threats to their recruitment, migration, and overall survival (Harrison *et al*., 2021). These alterations disrupt the natural sediment transport processes, leading to habitat degradation that directly affects eel populations and their ecological functions.

Riverbed modification significantly alters habitat structure, influencing the physical conditions necessary for anguillid eel survival (Pinder *et al*., 2022). Bed degradation through sand and gravel extraction reduces substrate complexity, which is essential for the shelter and feeding grounds of juvenile and adult eels (Breckling *et al*., 2020). The removal of sediments not only decreases habitat heterogeneity but also exposes fine sediment layers to increased erosion and siltation, which can smother benthic organisms, reducing prey availability for eels (Pinder *et al*., 2022). Moreover, bed modification alters channel morphology, leading to increased water velocity and decreased flow regime, making it difficult for eels to navigate and find suitable habitats (Moyle & Marchetti, 2019).

Anguillid eels are highly migratory species requiring unimpeded passage between freshwater and marine environments. Bed modification disrupts migration corridors by altering flow dynamics and sediment deposition patterns, creating barriers that hinder eel movement (Crook *et al*., 2021). Channelization and dredging lead to reduced water depth and flow alterations that can disorient migrating eels, increasing their vulnerability to predation and anthropogenic mortality (Miller *et al*., 2020). Additionally, modified riverbeds often contribute to the loss of critical recruitment habitats, especially in areas where sediment removal results in reduced organic matter accumulation, an essential factor for juvenile eel settlement (Crook *et al*., 2021).

Bed modification significantly influences the trophic interactions within river ecosystems by disrupting the benthic food web. Many benthic invertebrates, which form a crucial part of the diet of anguillid eels, depend on stable sediment structures (Clarke *et al*., 2020). The destruction of riverbeds through activities such as mining and dam-induced sediment trapping reduces the availability of these prey species, subsequently affecting eel foraging success and growth rates (Doughty *et al*., 2021). Reduced prey abundance due to bed modification has been linked to decreased condition factors in eels, which can influence their overall fitness and survival rates (Briand *et al*., 2022). Another consequence of bed modification is the alteration of sediment composition, leading to increased pollution and contaminant accumulation. Disturbed riverbeds release previously sequestered pollutants, including heavy metals and organic contaminants, which can have deleterious effects on eel physiology (Tesch, 2019). Elevated levels of pollutants such as mercury and polychlorinated biphenyls (PCBs) have been found in eels from modified riverbeds, with evidence suggesting bioaccumulation and biomagnification within the food web (Laffaille *et al*., 2003). Exposure to these pollutants not only affects eel health but also poses risks to human populations reliant on eels for subsistence and economic activities (Acou *et al*., 2020). The effects of bed modification are exacerbated by climate change, which alters hydrological regimes and sediment transport dynamics. Increased rainfall variability and extreme weather events accelerate erosion in modified riverbeds, leading to more pronounced habitat degradation (Laffaille *et al*., 2003). In addition, climate-induced changes in temperature and flow regimes can compound the negative effects of sediment removal, further reducing the resilience of eel populations (Poff *et al*., 2021). These combined stressors necessitate integrated management approaches that consider both anthropogenic and climate-related impacts on river ecosystems.

Addressing the ecological risks associated with bed modification requires sustainable river management strategies aimed at reducing anthropogenic pressures on riverbeds. Restoration efforts such as sediment replenishment, re-naturalization of riverbanks, and flow regulation can help mitigate habitat loss and improve recruitment conditions for eels (Kemp *et al*., 2022). Additionally, establishing protected zones where activities such as sand mining are restricted can aid in preserving critical habitats necessary for eel populations (Geist & Hawkins, 2020). Policies and regulations should also promote sustainable practices that balance economic activities with ecological conservation, ensuring the long-term sustainability of riverine ecosystems.

Anthropogenic alterations to river channels, including channelization, dam construction, and bank stabilization, have profound ecological consequences on aquatic biodiversity, particularly on catadromous species such as anguillid eels (Kemp *et al*., 2015). Channel modification alters the physical and biological integrity of river systems, affecting habitat availability, water flow regimes, sediment transport, and connectivity essential for eel recruitment, growth, and migration (Kemp *et al*., 2015). Channelization, involving the straightening and deepening of river channels for flood control, irrigation, and navigation, alters the natural hydrological and geomorphological processes of river systems. These modifications typically lead to increased water velocity, reduced habitat complexity, and loss of critical habitats such as riparian vegetation and floodplains (Jellyman & Bowen, 2019). Anguillid eels rely on diverse microhabitats, including undercut banks, woody debris, and submerged vegetation, which are often lost due to channelization (Bardonnet & Riera, 2021).

The loss of habitat complexity directly affects juvenile eels, which depend on slow-flowing, structurally rich environments for refuge and foraging (Arai *et al*., 2016). In the AGS and Ramisi Rivers, channelization may exacerbate habitat fragmentation, limiting the recruitment of glass eels and elvers into suitable upstream habitats, ultimately affecting population dynamics. Studies have shown that increased water velocity from channelization can hinder upstream migration of juvenile eels, particularly in species such as *Anguilla bengalensis* and *Anguilla mossambica* that require diverse habitat structures for growth and development (Silva *et al*., 2020). Concrete barriers and riprap used for bank stabilization prevent natural erosion but significantly impact the ecological functionality of river systems (Masese *et al*., 2021). These structures reduce the natural exchange between rivers and floodplains, thereby limiting nutrient cycling and habitat availability for benthic organisms, which form part of the diet of eels (Drouineau *et al*., 2018). Moreover, hardened banks reduce the availability of refuge for eels, exposing them to increased predation risk (Braga *et al*., 2012).

In the Kenyan context, riverbank stabilization projects aimed at preventing erosion have inadvertently contributed to habitat degradation for freshwater fish (Masese *et al*., 2021). The installation of concrete embankments along sections of the AGS and Ramisi Rivers may hinder the natural recruitment and dispersal of eels, restricting their movement between freshwater and estuarine environments. Additionally, bank stabilization efforts often lead to substrate homogenization, reducing the diversity of benthic invertebrates essential for eel foraging (Piper *et al*., 2021). Dams and weirs are among the most significant anthropogenic stressors affecting the migratory patterns of catadromous Anguillid eels. These structures disrupt natural flow regimes, alter sediment transport, and act as physical barriers to migration (Foulds & Lucas, 2013). In particular, upstream migration of juvenile eels and downstream migration of silver eels to spawning grounds are impeded by dams, leading to population declines (Acolas *et al*., 2020).

In the AGS and Ramisi Rivers, existing and proposed dam projects for hydropower and irrigation threaten eel populations by reducing water flow and connectivity between critical habitats. Studies have indicated that eels face increased mortality rates when attempting to pass through dam turbines, leading to significant losses of reproductive individuals (Calles *et al*., 2013). Furthermore, the alteration of flow regimes by dams can affect downstream estuarine conditions, influencing the recruitment success of glass eels into freshwater systems (Trancart *et al*., 2021). Mitigation measures such as fish ladders and eel passes have been proposed to facilitate eel migration past barriers. However, their effectiveness is often limited, as many eel species exhibit poor climbing abilities, and these structures do not fully mitigate the broader ecological impacts of altered flow regimes (Clarkin *et al*., 2019). In Kenya, there is limited research on the effectiveness of existing mitigation measures, highlighting the need for further studies on eel passage solutions specific to the region's river systems.

Channel modification affects sediment transport, which is crucial for maintaining suitable benthic habitats for eels. The reduction of sediment load due to dam construction and bank stabilization can lead to the degradation of downstream habitats, impacting prey availability and shelter for juvenile eels (Schneider *et al*., 2018). Conversely, excessive sedimentation from poorly managed land-use activities, such as agriculture and deforestation, can smother eel habitats and reduce water quality by increasing turbidity and reducing oxygen levels (Walker & Jones, 2018). In the AGS and Ramisi Rivers, sedimentation linked to riparian deforestation and construction activities threatens the integrity of eel habitats. Fine sediments can clog interstitial spaces in riverbeds, reducing habitat suitability for eels and their prey (Daverat *et al*., 2022). Additionally, altered water quality parameters, such as increased nutrient loads from agricultural runoff, can lead to algal blooms, further degrading eel habitats and affecting food availability.

Given the ecological importance of anguillid eels in riverine ecosystems and their role in sustaining local fisheries, addressing the impacts of channel modification is crucial for their conservation in Kenya. Effective management strategies should prioritize habitat restoration, maintaining connectivity between freshwater and marine environments, and implementing eel-friendly infrastructure modifications (Jacoby *et al*., 2015). Conservation measures such as reforestation of riparian zones, the creation of artificial "refugia", and improved fish passage solutions could mitigate some of the adverse effects of channel modification on eel populations.

5.2.4 Effect of flow modification on anguillid eels' ecology

Flow modification is a critical anthropogenic stressor affecting freshwater ecosystems, particularly for catadromous species such as anguillid eels (Jansen *et al*., 2007). These species rely on natural flow regimes for migration, feeding, and reproduction, making them highly vulnerable to alterations in river discharge, water velocity, and seasonal flow patterns. Flow modifications arise from human activities such as dam construction, water abstraction, channelization, and climate change-induced hydrological shifts, all of which impact the ecological integrity of riverine habitats (Poff *et al*., 2017). Catadromous Anguillid eels, including *Anguilla bengalensis*, *A. bicolor*, and *A. mossambica*, depend on unimpeded freshwater systems for their upstream and downstream migrations (Arai, 2016). Barriers to movement, such as dams and weirs, disrupt these life cycle processes, reducing recruitment success and population viability (Verhelst *et al*., 2018). In the context of River AGS and River Ramisi, hydrological modifications, including the construction of irrigation dams and water abstraction for agriculture, have altered flow regimes, potentially impeding the movement of juvenile eels (elvers) from marine to freshwater environments (Mbaka & Mwaniki, 2015).

Low flow conditions exacerbate recruitment challenges by reducing water depth and connectivity among essential habitats, leading to higher mortality rates and lower population densities (Silva *et al*., 2018). Conversely, sudden increases in flow due to flood events from artificial reservoir releases may disorient migrating eels, increasing their vulnerability to predation and energy depletion (Bardonnet & Baglinière, 2018). This highlights the need for eco-hydrological management approaches to balance human water needs with the biological requirements of eels. Flow modification influences sediment transport, nutrient cycling, and habitat structure, thereby affecting the quality of eel habitats and the availability of prey species (Jacoby *et al*., 2015). Reduced flow rates contribute to sediment deposition, leading to habitat homogenization and the loss of structurally complex microhabitats where eels forage and seek refuge (Baumgartner *et al*., 2020). For example, excessive sedimentation in low-flow conditions can smother benthic invertebrates and aquatic vegetation, key dietary components for eels (Jacoby *et al*., 2015). Additionally, changes in flow regimes can alter the composition of fish and invertebrate communities, indirectly affecting eel feeding ecology. In highly modified river sections, reductions in benthic macroinvertebrate diversity and abundance have been observed, which may lead to increased competition for food resources and shifts in eel diet composition (Aedo *et al*., 2020). Hydrological alterations affect eel growth rates, condition factors, and overall physiological health (Arai *et al*., 2014). Eels in flow-modified rivers often exhibit lower condition factors due to reduced food availability and increased energetic costs associated with navigating altered hydrodynamic conditions (Trancart *et al*., 2020).

Seasonal flow reductions can lead to hypoxic conditions, particularly in stagnant water zones, which negatively affect eel metabolism and survival. Studies have demonstrated that hypoxia tolerance in anguillid eels varies among species, with prolonged exposure to low oxygen levels leading to increased stress responses and potential mortality (Cullen & McCarthy, 2017). These physiological constraints further underscore the necessity of maintaining natural flow regimes to sustain eel populations in rivers experiencing hydrological modifications.

To mitigate the adverse effects of flow modification on anguillid eels, various conservation and management strategies have been proposed. The implementation of fish-friendly infrastructure, such as eel ladders and bypass channels, has shown promise in facilitating upstream migration despite flow barriers (Briand *et al*., 2018). Additionally, environmental flow assessments have been employed to establish minimum flow requirements that support eel habitats while accommodating human water demands (Arthington *et al*., 2018). In the Kenyan context, the integration of ecosystem-based water management approaches, including the restoration of riparian buffer zones and the regulation of water abstraction, is essential to sustaining healthy eel populations. Community engagement in sustainable water resource management can further enhance conservation efforts, ensuring that local livelihoods dependent on eels are safeguarded while minimizing ecological risks (Ogutu-Ohwayo *et al*., 2016).

5.2.5 Impact of inundation on habitat integrity for anguillid eels

Inundation, defined as the temporary or seasonal flooding of terrestrial or semi-aquatic environments, plays a critical role in shaping the structure and function of riverine habitats (Junk *et al*., 1989; Tockner & Stanford, 2002). Anguillid eels are facultative catadromous fishes that utilize both freshwater and estuarine systems. Flooding regimes influence key aspects of habitat availability, connectivity, and ecological quality. The impacts of inundation on eel habitats can be both beneficial and detrimental, depending on the magnitude, timing, duration, and frequency of flooding events. Moderate, seasonal inundation can enhance habitat integrity by increasing habitat complexity and access to productive floodplain zones (Van Appledorn *et al*., 2024). Floodplains and inundated riparian zones provide critical nursery and foraging habitats, particularly for juvenile eels (elvers and yellow eels). These areas are often rich in macroinvertebrates, crustaceans, and small fish, which constitute important dietary items for anguillid species (Sugeha *et al*., 2001). In this context, inundation supports biological productivity and energy flow, facilitating the growth and survival of eels during their freshwater residency phase. In addition, inundation plays a key role in restoring longitudinal and lateral connectivity within river networks (Ward *et al*., 2002). During high-flow or flood events, eels can access upstream habitats or side streams that may be otherwise inaccessible during low-flow periods. This connectivity is essential for the dispersal and migration of juvenile stages, and for accessing diverse microhabitats necessary for different life-history strategies. Elvers benefit from increased access to upstream rearing habitats, while adult yellow eels may exploit newly inundated zones for feeding (Tsukamoto & Arai, 2001).

However, when inundation patterns are altered due to anthropogenic activities, such as dam operations, deforestation, and climate change-induced shifts in rainfall, negative consequences for habitat integrity may arise (Tsukamoto & Arai, 2001). Artificial or prolonged inundation may lead to habitat degradation through increased turbidity, sedimentation, and organic loading, which can decrease dissolved oxygen levels and affect benthic prey communities (Bunn & Arthington, 2002; Kingsford, 2000). These changes can reduce habitat suitability for eels, particularly in their critical growth stages. Furthermore, abrupt changes in water levels following inundation events may result in the stranding of eels in ephemeral pools, increasing mortality risk. Habitat fragmentation caused by prolonged flooding or water infrastructure may also interfere with the upstream migration of glass eels and elvers, disrupting natural recruitment processes (Ibbotson *et al*., 2002). In highly modified systems, altered flood regimes can also disrupt the environmental cues which eels rely on for movement and reproduction. The timing of downstream migrations, often triggered by hydrological and lunar cycles, may be misaligned, which can impact successful spawning migrations (Behrmann-Godel & Eckmann, 2003; Tsukamoto, 1992). Thus, the ecological integrity of inundated habitats must be evaluated in the context of natural versus anthropogenic drivers.

5.2.6 Impact of exotic flora and fauna on anguillid eels

The introduction of exotic species, both macrophytes and fauna, into freshwater ecosystems has emerged as a significant driver of ecological change, with wide-ranging impacts on native biodiversity and ecosystem functioning (Strayer, 2010; Thomaz *et al*., 2015). For Anguillid eels, which are already threatened by habitat fragmentation, overfishing, and pollution, the presence of invasive species compounds the pressures on their habitats, feeding ecology, and population dynamics (Jacoby *et al*., 2015). Exotic aquatic plants such as *Eichhornia crassipes* (water hyacinth), *Salvinia molesta*, and *Hydrilla verticillata* have proliferated in many tropical and subtropical freshwater systems, including rivers and lakes inhabited by Anguillid eels. These macrophytes can significantly alter the physical and chemical characteristics of aquatic habitats (Thomaz & Cunha, 2010). Dense mats of floating vegetation reduce light penetration, leading to oxygen depletion in the water column, particularly during night-time respiration and decay phases (Villamagna & Murphy, 2010). Such hypoxic conditions can negatively affect the metabolic performance and survival of eels, particularly juveniles. Moreover, invasive macrophytes alter habitat structure by modifying water flow, reducing habitat heterogeneity, and outcompeting native aquatic vegetation that may support diverse macroinvertebrate communities (Thomaz & Cunha, 2010). This shift in habitat complexity and food web composition can influence the availability and diversity of prey items for eels, especially during their yellow eel growth stage when benthic foraging is predominant (Katselis *et al*., 2003). In systems where submerged macrophytes dominate, the reduction in open-water zones can also interfere with eel movements and hunting efficiency.

Invasive fish species and crustaceans also pose significant threats to Anguillid eels. Predatory species such as *Micropterus salmoides* (largemouth bass), *Clarias gariepinus* (African sharptooth catfish, when introduced outside its native range), and *Oreochromis niloticus* (Nile tilapia) have been observed to compete with eels for food resources or prey on early life stages such as elvers (Gkenas *et al*., 2012). Invasive fish can disrupt native food webs, potentially reducing the abundance of key prey items such as prawns, fish fry, and aquatic insects, which constitute significant portions of eel diets (Sugeha *et al*., 2001). In addition, exotic crustaceans such as *Procambarus clarkii* (red swamp crayfish) can alter sediment dynamics through bioturbation and can predate on or outcompete native invertebrates, further modifying benthic food resources (Twardochleb *et al*., 2013). These changes in prey dynamics can lead to altered foraging behaviour, reduced growth rates, and even displacement of eels from preferred habitats. Beyond trophic interactions, some invasive species may introduce novel pathogens or parasites, as was the case with *Anguillicola crassus*, a parasitic nematode originally from East Asia, associated with reduced swimming performance and impaired migration in European eels (*Anguilla anguilla*) (Kirk, 2003). While this parasite has not been extensively reported in African Anguillid populations, its potential spread remains a concern in interconnected or poorly monitored ecosystems.

The combined effects of exotic macrophytes and fauna may lead to habitat simplification, prey base alteration, and increased competition or predation risk, all of which can impact the recruitment, growth, and survival of anguillid eels. In river systems such as the Athi-Galana-Sabaki and Ramisi, which are increasingly exposed to biological invasions due to catchment degradation and aquaculture introductions, these effects warrant close monitoring. Altered ecological conditions may shift eel distributions, limit the availability of nursery habitats, and compromise population resilience under changing environmental conditions. The interaction of exotic species with other stressors, such as water abstraction, pollution, and habitat fragmentation, can exacerbate their impact on eel populations. Therefore, effective management of anguillid habitats must integrate invasive species control with broader ecosystem restoration strategies to maintain the ecological integrity and productivity of eel-supporting systems.

5.2.7 Impact of indigenous vegetation removal and bank erosion on anguillid eels

The removal of indigenous riparian vegetation and the subsequent increase in bank erosion are critical factors influencing riverine ecosystem health. These disturbances significantly affect the structural integrity, hydrology, and ecological functioning of freshwater habitats (Gregory *et al*., 1991; Naiman & Décamps, 1997). For anguillid eels, which depend on structurally diverse and stable habitats during various life stages, the degradation of riparian zones can result in the loss of essential microhabitats, disruption of food webs, and increased vulnerability to both natural and anthropogenic stressors. Riparian vegetation provides multiple ecological services crucial for maintaining eel habitats. Overhanging vegetation moderates stream temperature by offering shade, regulates sediment input, and supplies all ochthonous organic matter, which fuels aquatic food webs (Sweeney & Newbold, 2014). Leaf litter and woody debris contribute to the formation of coarse benthic substrates and instream cover—

features essential for foraging, shelter, and predator avoidance, particularly during the yellow eel stage (Tesch, 2003).

The removal of native vegetation, often driven by agricultural expansion, urban development, or logging, leads to reduced bank stability and increased exposure of the river channel to solar radiation and runoff. Elevated water temperatures, sedimentation, and nutrient loading can negatively affect eel physiology, reduce dissolved oxygen levels, and degrade benthic habitats where eels typically forage (Broadmeadow & Nisbet, 2004). Such conditions may deter glass eels and elvers from colonizing affected reaches, ultimately impacting local recruitment and population structure.

Bank erosion, often a direct consequence of vegetation loss, alters channel morphology, increases turbidity, and reduces habitat complexity (Rosgen, 1996). Erosion-driven sedimentation can smother benthic substrates and microhabitats used by eels for ambush hunting or concealment. Fine sediment deposition also clogs interstitial spaces in gravel beds and root mats that are vital for invertebrate production, thereby reducing prey availability for eels (Bilby & Bisson, 1992). Moreover, high sediment loads can interfere with eel gill function and hinder migration, particularly for juvenile stages that rely on visual and tactile cues for upstream movement. In lowland tropical rivers such as the Athi-Galana-Sabaki and Ramisi, where seasonal flows are common, bank erosion may intensify during the rainy season, causing widespread habitat instability and potential stranding of eels in disconnected or deoxygenated pools. The combined effect of vegetation loss and erosion contributes to the fragmentation of suitable eel habitats, especially in rivers where migration corridors are already constrained by other anthropogenic factors such as dams and water abstraction. Disrupted flow regimes and reduced habitat quality may affect eel behaviour, condition, and growth rates, as well as limit the resilience of populations to environmental change (Schmidt *et al.*, 2009).

5.2.8 Impact of solid waste disposal on anguillid eels

Solid waste pollution in aquatic environments poses a growing threat to freshwater ecosystems globally, with critical implications for aquatic biodiversity, including catadromous species such as anguillid eels (Ogello *et al.*, 2024). Improper disposal of solid waste, ranging from plastics and metals to organic debris and urban refuse, degrades water quality, alters habitat structure, and introduces toxic compounds into aquatic food webs (Windsor *et al.*, 2019). Plastics, bottles, polythene bags, construction debris, and other non-biodegradable materials can accumulate along riverbanks and within channels, leading to blockage of critical microhabitats used by eels for foraging and shelter (Moore, 2008). This accumulation reduces habitat heterogeneity by covering benthic substrates, smothering vegetation, and obstructing eel movement, especially during their upstream migration as glass eels and elvers. In urbanized or peri-urban river systems, solid waste disposal often occurs near riparian zones, contributing to localized anoxia due to the decomposition of organic waste and the retention of debris during low-flow periods (Tesch, 2003). Such conditions are particularly detrimental to yellow eels, which rely on structurally complex habitats with adequate dissolved oxygen levels and interstitial spaces for concealment (Kaifu *et al*., 2013). Solid waste also serves as a vector for chemical contamination in freshwater systems. As plastics degrade, they release microplastics and associated toxicants such as phthalates, bisphenol A, and heavy metals into the water column and sediments (Rochman *et al*., 2013). These compounds can bio-accumulate in aquatic organisms, with potential physiological effects on eels, including endocrine disruption, reduced reproductive capacity, and immunosuppression (Browne *et al*., 2011).

Moreover, eels are known to bio-accumulate persistent organic pollutants (POPs) and heavy metals due to their long lifespan, benthic behaviour, and lipid-rich tissues (Belpaire & Goemans, 2007). In habitats contaminated with urban and industrial waste, the ingestion or dermal exposure to such pollutants can impair eel health, leading to reduced growth rates, lesions, or even mortality, particularly in juvenile life stages. Chronic exposure may also affect the condition factor and hepatosomatic index, important indicators of eel health and metabolic function (Ferrante *et al*., 2010). Solid waste impacts aquatic food webs by disrupting the base of the trophic structure. Organic debris and leachate from decomposing waste materials can promote algal blooms and bacterial proliferation, leading to hypoxic conditions that reduce macroinvertebrate diversity and abundance (Allan & Castillo, 2007). Since eels are opportunistic feeders that depend on aquatic invertebrates, fish, and crustaceans, a reduction in prey base due to water quality decline and sediment contamination can impair feeding efficiency and growth (Sugeha *et al*., 2001).

5.2.9 Introduction to Relative Risk Assessment (RRA)

Relative Risk Assessment (RRA) is a widely recognized tool to evaluate the risks posed by multiple stressors to species and ecosystems. The Relative Risk Assessment (RRA) model is particularly valuable in environmental management and conservation due to its ability to integrate multiple sources of risk across diverse spatial and temporal scales. Unlike other assessment models that may focus on singular factors or static conditions, the RRA model offers a more comprehensive approach by considering the cumulative and synergistic effects of various stressors, such as pollution, habitat loss, and climate change, on ecosystems and species (Landis & Wiegers, 1997). This holistic perspective is crucial for understanding the complex interactions between different environmental variables and their combined impact on vulnerable species like anguillid eels. Additionally, the RRA model allows for the prioritization of management actions by quantifying the relative risks posed by different stressors, enabling more effective allocation of resources and targeted interventions (Landis, 2003). By incorporating uncertainty and variability in ecological responses, the RRA model also provides a more realistic and adaptable framework for risk management, making it superior to more traditional, deterministic models that may overlook the nuances of ecosystem dynamics (Hope, 2006). This adaptability is essential in addressing the multifaceted and evolving nature of environmental threats, particularly in the context of global environmental change.

The RRA model is structured around nodes that represent different environmental factors or variables influencing the overall risk to a particular species or ecosystem (Landis, 2003). These nodes are assigned values based on the extent of their impact, and their relative importance is weighted to provide a comprehensive risk score. The model is particularly valuable in ecological risk assessment, as it integrates qualitative and quantitative data from various sources, including expert judgment, field observations, and community input, to assess risks holistically (Landis & Wiegers, 2007). This comprehensive approach is crucial for managing complex ecosystems, where multiple interacting stressors must be considered to understand and mitigate risks effectively (Hope, 2006).

RRA has been extensively applied in aquatic ecosystems to assess the impact of multiple stressors on species' health and ecosystem integrity. Aquatic ecosystems are highly susceptible to cumulative impacts from human activities, such as habitat modification, pollution, and the introduction of invasive species (Dudgeon, 2019; Fahrig, 2003). These activities often interact synergistically, leading to more severe ecological consequences than the sum of individual stressors alone (Brown *et al.*, 2013; Crain *et al.*, 2008). The RRA framework is particularly suited to addressing these complexities by allowing for the integration of diverse stressors and their interactions, providing a more accurate picture of the potential risks to aquatic species. By using RRA, conservationists and policymakers can prioritize management actions that target the most significant threats to aquatic biodiversity.

Given the complex life cycle of Anguillid eels and the diverse stressors they face, RRA provides a valuable framework for identifying key risk factors and informing conservation strategies (Hope, 2006). By assessing the relative impact of different stressors on each life stage, RRA can help prioritize management actions that address the most critical threats. For example, reducing barriers to migration and improving water quality in freshwater habitats could be targeted to enhance the survival of yellow and silver eels (Edeline *et al*., 2006; McIntyre *et al*., 2016). Additionally, addressing global issues like climate change through policy interventions can mitigate broader risks to eel populations across their migratory range (Hartmann *et al*., 2013). The application of RRA to the study of anguillid eels provides a robust method for evaluating the risks posed by multiple environmental stressors across different life stages. By integrating qualitative and quantitative data, RRA enables a comprehensive assessment of the factors contributing to eel population declines and supports the development of targeted conservation strategies. As human activities continue to impact aquatic ecosystems, the use of RRA will be increasingly important in ensuring the survival and sustainability of vulnerable species like anguillid eels. Through this model, conservationists can develop more effective strategies to mitigate risks and preserve the ecological and economic value of eel populations for future generations.

This study aimed to evaluate the effects of multiple stressors on all life stages of anguillid eels in two east-flowing rivers in Kenya, which experience varying levels of anthropogenic stress (River AGS and River Ramisi). While numerous studies have addressed freshwater ecosystem degradation and its impacts on other organisms, no research has yet focused on the ecological risks faced by Kenya's anguillid eels, despite their listing on the IUCN Red List of endangered species. This study is, therefore, a pioneering effort in the region. Understanding the ecological risks faced by eels is crucial for conservation efforts aimed at preserving the species' population viability.

5.3 Materials and Methods

The study area is as described in Chapter Three (3.2.1)

5.3.1 Habitat integrity of the study sites

The general habitat integrity of the study Rivers was evaluated using the method by Kleynhans (1996) as follows. In-stream and riparian habitat modifiers were assessed and scores were assigned. The final habitat integrity score (HIS) was calculated using equation (11)

$$HIS = 100 - \left[\frac{\sum IS \times C_i}{25}\right] \qquad \ldots 11$$

Where HIS is Habitat Integrity Score; IS is each score; Ci is each weight

5.3.2 Ecological risk assessment

Relative Risk Model (RRM) derived from the Regional-scale Ecological Risk Assessment framework established by Wiegers *et al*. (1998) and operationalized for application in Africa by O’Brien and Wepener (2012). The model was used to rank ecological risks impacting on life stages of Anguillids of River AGS and River Ramisi. The RRM application involves the use of both Bayesian Networks (BNs) and Monte Carlo randomization procedures that adapt to current best international practice (Van *et al*., 2016). The term "relative" means the assessment is a ranking exercise that applies relative probability (O'Brien *et al*., 2021; Vezi *et al*., 2019). The basis of the study focused on describing the cause-and-effect relationships of multiple stressor sources, stressors, and ecosystem/habitat relationships and how these interactions affected stakeholders' concerns (end points). In the current study, the "causes" are habitat degradation, pollution, climate change, alien species, and overharvesting of eels, all of which produce multiple stressors (e.g., fragmented habitat, chemical compounds, increased ocean temperatures, competition for food, and pressure on eel populations, among others). The stressors affect ("effects") spawning, larvae recruitment (leptocephali), yellow eel maintenance, and silver eel escapement. The risk of ecological impacts for each eel life stage was calculated and ranked by quantitatively determining the interactions of the stressors and habitats. A ranking of ecological risks presented the results in terms of Zero, low, moderate, and high classifications. The presentation of the methods consisted of the five parts for ecological risk assessment: problem formulation, exposure assessment, effects assessment, risk characterization, and uncertainty analysis as described below;

Problem Formulation: This initial step involved defining the scope and objectives of the Ecological Risk Assessment by identifying the stressors of concern, the ecological receptors (eel life stages) that may be affected, and the pathways through which stressors may interact with receptors. Problem formulation also regards the potential for exposure and the ecological importance of the effects.

Exposure Assessment: Exposure assessment considered quantifying the magnitude, and spatial extent of stressor exposure to the eels' life stages.

Effects Assessment: Effects assessment aimed to characterize the potential negative impacts of the stressor exposure on the eels' life stages. This involved integrating ecological indicators to evaluate the relationship between stressor exposure and potential ecological responses. Effects assessment considered various endpoints, such as changes in population of spawners, recruiting larvae numbers, successful yellow eels' maintenance and silver eels' escapement.

Risk Characterization: Risk characterization was used to integrate the information gathered in the problem formulation, exposure assessment, and effects assessment stages to approximate the overall ecological risk posed by the stressors. This involved combining data on stressor exposure and effects to quantify the potential and scale of adverse ecological effects. Risk characterization included probability distributions, to express the likelihood and severity of adverse effects.

Uncertainty Analysis: Uncertainty analysis involved scenario analysis and probabilistic modelling to quantify and spread the range and sources of uncertainty in risk estimates. In the current study where there was uncertainty with the different input nodes from specific sites, the risk rank was assigned a zero or low rank in order not to overcompensate for the uncertainty.

The critical life stages of the eels were denoted as; recruitment of larvae in the ocean (LARVAE _ RECRUIT), Elver and yellow eel maintenance in river (ELV_YEL_MAINT), adult silver eel escapement from river into the ocean (SILVER_ESCAP) and oceanic spawning of adult eels (SPAWNNING).

Ranking of risks

A ranking scheme incorporated in the software, for each node helped to calculate the relative risk to each specific endpoint. End points (ranked as either zero, low, moderate, or high) described the wellbeing condition or severity of impact for the respective nodes. The causal-effect nodes that helped in understanding the complex relationships and dynamics within the ecosystem aided in assessing the potential risks posed by stressors such as pollutants, invasive species, habitat destruction, or climate change as presented in Appendix 7. Table 8 shows the ranking values assigned to four categories of risks as derived from relative risk model.

Table 8: Risk Ranks and Node Descriptions

Rank	Description of node (Appendix7)
Zero	The node is in pristine condition, and no impact or risk (less than 25%) is present compared to pre-anthropogenic establishment or reference conditions.
Low	The node is in a largely natural condition, and low impact or risk (25 – 50%) is present compared to pre-anthropogenic establishment or reference conditions. The ecosystem is functioning sustainably
Moderate	The node is in a moderately modified condition, and moderate impact or risk (50 – 75%) is present compared to pre-anthropogenic establishment or reference conditions. Ecosystem functioning is on the threshold for possible failure
High	The node is in a significantly altered condition, and high impact or risk (75 – 100%) is present compared to pre-anthropogenic establishment or reference conditions. Ecosystem functioning is at the failure threshold. The ecosystem can sustain functioning (Wade *et al*., 2020)

5.3.2 Scenario modelling and stressor impact analyses

Qualitative data were collected through expert judgments, community inputs, and field observations. Each node received a score based on its impact on the overall risk. For instance, a node representing "habitat degradation" might be assigned a high score in areas with significant development and pollution. The importance of each node was then weighted, reflecting its relative significance in contributing to the overall risk. For example, if "barriers to migration" were deemed more critical than "predation," they were assigned a higher weight in the model. These nodes are selected based on their relevance to the specific life stage of the species being studied and the environmental context of the area (Table 9). To predict future risks, hypothetical scenarios were developed based on assumptions about changes in environmental conditions, human activities, and other influencing factors over time. This process involved considering secondary data from government policies, strategic plans, and input from experts and the community to identify potential future risks. For instance, a scenario might assume increased habitat degradation due to urbanization, climate change, or intensified agricultural practices, leading to adjustments in the node weights. If new risks, such as the introduction of invasive species or new migration barriers, were anticipated, additional nodes were introduced, or existing ones were modified accordingly.

Under the adjusted node values and weights, the model was run to generate new risk scores, providing a quantitative assessment of overall risk under the future scenario. These scores were then compared to the present-day risk assessment to evaluate the extent of potential changes. The analysis of future risk scores helped identify the factors that contribute most to the predicted risks and the life stages of the species that are most vulnerable. A hypothetical future risk assessment scenario, informed by conservation and management strategies, was conducted by adjusting node weights that had a significant impact on eel life stages. To model this, the score for a node expected to have increasing impacts was adjusted accordingly. The results were presented graphically to guide conservationists and managers in developing effective intervention strategies.

Table 9: Risks Impacting Different Eels' Life Stages

Life stage	Risk assessment nodes
Larvae/Glass eels recruitment	*Presence of barriers (Dams, weirs, sluices, tidal barriers) Climate change (Temperature fluctuations, sea-level rise, altered ocean currents) Habitat degradation (Coastal development, estuarine degradation) Water quality (Pollution (chemical runoff, heavy metals, pesticides), salinity changes, oxygen levels Fishing pressure (Overfishing, illegal harvesting, bycatch) Predation (Increased predator populations, introduced species) Competition (Invasive species, competition with other juvenile fish) Changes in Oceanic Conditions (Altered ocean currents, changes in plankton availability, oceanic pollution)
Elvers- Yellow eels maintenance	*Habitat Degradation (Loss of aquatic vegetation, alteration of river channels, sedimentation, damming, and other physical alterations to their habitat). Water Quality Degradation (Pollution levels (e.g., chemical, agricultural runoff), pH changes, dissolved oxygen levels, temperature fluctuations).

	Food Availability (Changes in prey species populations, competition with other species, effects of habitat degradation on the food web). Predation Risk (Presence of invasive species, changes in predator populations, altered predator-prey dynamics due to environmental changes). Fishing Pressure (Overfishing, bycatch, and illegal harvesting). Hydrological Changes (Altered flow regimes due to damming, water withdrawals for irrigation, and urbanization). Climate Change (Rising temperatures, altered precipitation patterns, changes in seasonal cycles) Contaminant Accumulation (Bioaccumulation of heavy metals, persistent organic pollutants (POPs), and other contaminants that can affect eel health and reproduction). Competition with Other Species (Interactions with other fish species that may compete for resources such as food or habitat).
Silver eels escapement	*****Barrier Effects (Physical barriers like dams and weirs can prevent or delay the migration of silver eels from freshwater to the ocean, reducing their chances of successful spawning. Habitat Degradation (Degraded habitats, especially in downstream areas, can affect the health and condition of silver eels, making them less fit for migration. Poor water quality, lack of cover, and altered flow regimes can also hinder their ability to escape. Water Quality (e.g., Pollution, Oxygen Levels) (Poor water quality, including pollution and low oxygen levels, can stress or kill silver eels during their migration. Contaminants may also accumulate in their bodies, affecting reproductive success). Fishing Pressure (Silver eels are often targeted by fisheries due to their size and maturity. High fishing pressure can significantly reduce the number of eels reaching the sea). Predation. (Predation pressure, particularly in degraded habitats or at bottlenecks like dams, can increase mortality rates during migration). Climate Change (Changes in temperature and hydrological regimes due to climate change can alter migration timing and success, with

	warmer waters potentially stressing eels or leading to mismatches in migration cues). Flow Regime Changes (Alterations in natural river flow, such as reduced flow due to water extraction or modified flow regimes from dam operations, can delay migration or reduce the number of eels able to escape. Turbines and Water Intakes (Hydroelectric turbines and water intakes can physically harm or kill migrating eels, reducing the number of silver eels that successfully reach the ocean).
Spawning	*Oceanic Conditions (The conditions in the ocean where eels spawn, such as temperature, salinity, and currents, are crucial). Unfavourable changes in these conditions can impact the success of spawning and the survival of larvae. Availability of Spawning Habitat (The specific areas where eels spawn is limited and specific. Any changes or loss of these habitats due to factors like climate change or oceanographic shifts can severely impact spawning success). Predation; (Both adult eels and their eggs or larvae are subject to predation. Increased predation rates can significantly reduce the number of eels that successfully spawn or the number of viable offspring.) Pollution (Oceanic pollution, including plastic debris, chemical contaminants, and oil spills, can affect the spawning environment or directly harm adult eels and their progeny). Climate Change (Changes in ocean temperature and currents due to climate change can disrupt spawning cues, alter spawning grounds, and affect the survival of larvae). Overfishing of Adult Eels (High levels of fishing pressure on adult eels, particularly before they reach the spawning grounds, can reduce the number of individuals that spawn, impacting population recruitment).

	Genetic Diversity (Reduced genetic diversity within eel populations can affect their adaptability to changing conditions in the spawning grounds, potentially leading to lower survival rates of offspring).

*Risk assessment node with the highest impact on the respective eels' life stage.

Table 10: The Different Scenarios and Node Descriptions Used in the Study

Scenario name	Description	Node variables
Natural (NAT)	This scenario represents the natural pre-anthropogenic state of the site. Assumed that no or little modification has occurred to the environment.	All input nodes were set to have a zero rank (0–25%) range.
Present (PRS)	This scenario represents the present state of node variables and the impacts or risks that are currently present at each site.	All input nodes were adjusted to fit the current impact rank.
Future 1 (FUT 1)	This scenario represents the future state over the next 20 years if no interventions take place.	All input nodes were adjusted in relation to the current scenario and with consideration of escalating stressor drivers.
Future 2 (FUT 2)	This scenario represents the future state over the next 30 years if no interventions take place.	All input nodes were adjusted in relation to the FUT 1 scenario and with consideration of escalating stressor drivers.

Note. The scenario timelines assume linear progression of anthropogenic stressors if no management interventions are implemented.

5.3.3 Natural, present and future risk assessment impact variables

Three scenarios were assigned impact variables according to stressor impacts with natural (NAT)- pristine scenario having the lowest of or between 0 and 25% risk scores (Table 10). Natural risk scenarios were considered as potential hazards that are inherent in the environment without any human influence. Present (PRS) risks were considered as threats and stressors influenced by humans and impacting ecosystems at the present time. Future (FUT) risks involved projecting potential threats and challenges that ecosystems may face in the future based on current trends and anticipated changes.

5.3.4 Data analysis

The overall habitat integrity data for the study sites was analysed using the methods developed by Kleynhans (1996). To assess the difference in habitat degradation between River Ramisi and River AGS, an ANOVA test was conducted with a significance level of $p \leq 0.05$. Tukey's range analysis was employed to analyse water parameters. Risk assessment data analysis was conducted using the modelling program Netica, developed by Norsys Software (http://www.norsys.com/). By using Netica, the data was structured in such a way as to visualize and quantify the dependencies between different factors influencing the system under study. This approach facilitated the identification of key variables and the assessment of their relative impacts on the outcomes of interest. Netica played a crucial role in the data analysis by enabling a distinct understanding of the interrelationships within the data, thereby supporting the information on effective strategies and recommendations.

5.4 Results

5.4.1 The habitat integrity of Rivers AGS and Ramisi ecosystems

The overall habitat integrity score for River Ramisi was 92.3%, placing it in class 'B' according to Kleynhans' (1996) classification. This indicates that the ecosystem is largely natural with minimal modifications; while some changes in habitat and biota may have occurred, the ecosystem functions remain essentially unchanged. Both the instream and riparian environments were largely intact. In contrast, River AGS had a habitat integrity score of 56.6%, placing it in class 'D' according to the same classification system. There was a significant difference in habitat integrity between River AGS and River Ramisi, with River AGS showing significantly greater anthropogenic impact (ANOVA $p = 0.04$, $F = 6.49$). The most significant habitat modifier impacts in River AGS were bank erosion, water abstraction, vegetation removal, water quality degradation, and barriers (Table 11). Some examples of these anthropogenic activities are shown in Figures 23a to k

Table 11: Instream and Riparian Habitat Modifier Scores in Rivers AGS and Ramisi

Instream Habitat Modifier	River AGS Score	River Ramisi Score	Riparian Modifier	River AGS Score	River Ramisi Score
Water abstraction	12.0	0.0	Bank erosion	12.5	6.0
Water quality	10.5	4.0	Indigenous vegetation removal	11.5	1.5
Bed modification	6.0	3.0	Water abstraction	12.0	0.0
Channel modification	11.0	2.5	Water quality	10.0	4.0
Flow modification	11.0	2.5	Exotic vegetation encroachment	7.5	1.0
Inundation	4.0	2.0	Channel modification	10.5	0.0
Exotic macrophytes	5.5	0.0	Flow modification	10.0	2.0
Exotic fauna	5.0	3.0	Inundation	4.0	0.0
Solid waste disposal	3.5	0.0	—	—	—

Note. Dashes (—) indicate that no corresponding riparian modifier was assessed for that instream factor

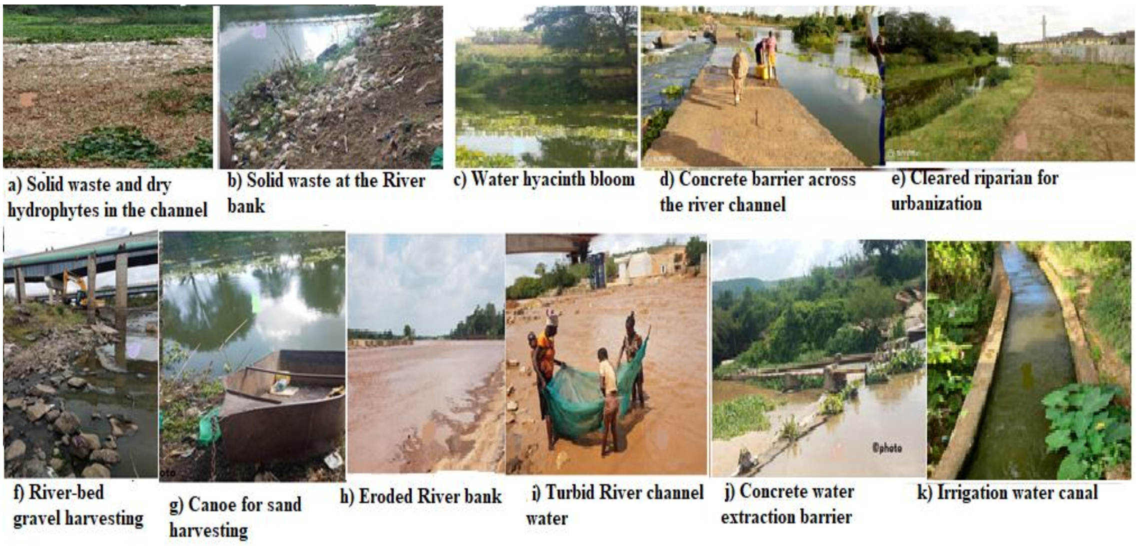


Figure 23: Overview of Various Anthropogenic Activities Impacting the River AGS Ecosystem

5.4.2 Analysis of risks across the eels' life stages in the natural and present scenarios in the River Ramisi

The risk scores for the life cycle ecology in natural scenario (NAT) in the Ramisi River ranged from 22.41 ± 18.06 to 24.29 ± 15.99, with the larval recruitment phase having the lowest risk, while the Silver eel escapement phase had the highest, respectively (Figure 24). The error bars indicate standard deviation. In the Present Scenario (PRS), the risk scores ranged from 24.49 ± 16.1 to 39.73 ± 25.20, with the larval recruitment phase having the lowest risk while the yellow eel maintenance phase had the highest risk scores, respectively (Figure 24).

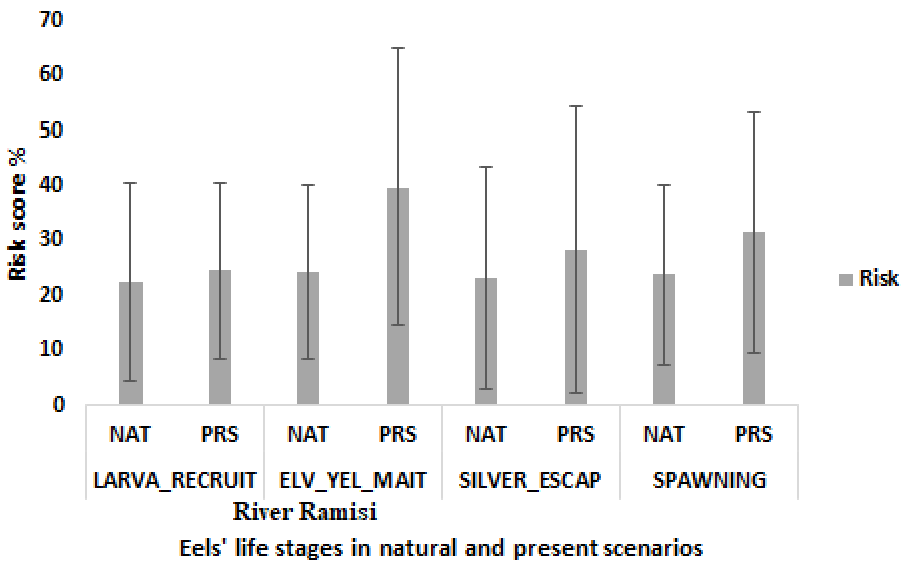


Figure 24: Percentage Risk of the Eels' Life Stages in the Natural and Present Conditions in the River Ramisi

5.4.3 Risk distribution in the natural and present scenarios among the eel life stages in the River Ramisi

Generally, the hypothetical natural (NAT) and present (PRS) scenarios for all life stages in River Ramisi were dominated by a zero-risk percentage distribution (Figure 25)

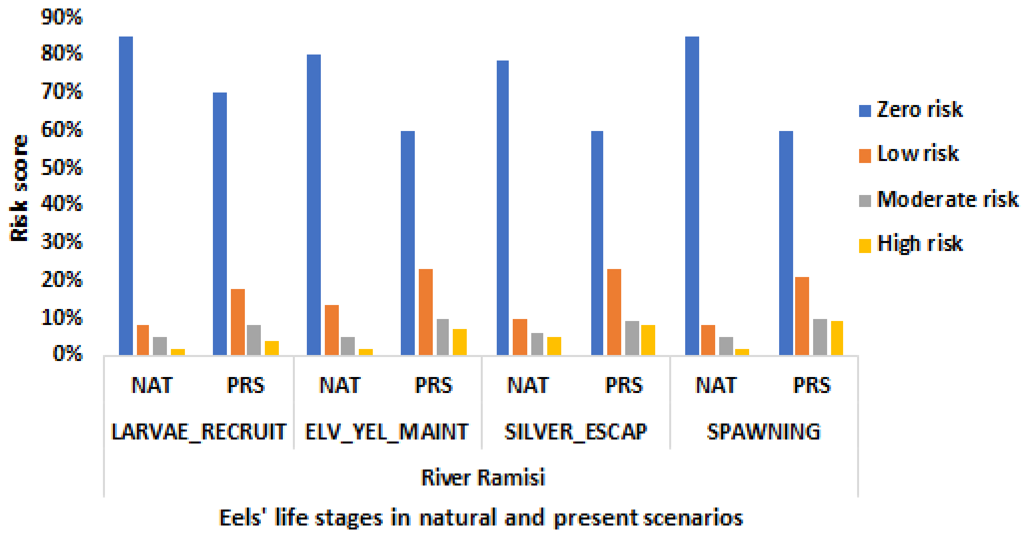


Figure 25: Relative Risk Percentage Distribution for the Hypothetical Natural (NAT) and Present (PRS) Scenarios for the Different Life Stages at the River Ramisi Ecosystem

5.4.4 Analysis of risks across eel life stages in natural and present scenarios in the River AGS

In the Natural (NAT) scenario in the life cycle ecology in the AGS River, the risk scores ranged from 23.58 ± 14.65 to 24.40 ± 21.22, with the lowest risk in the Larvae Recruitment phase and yellow eel maintenance phase, respectively (Figure 26). In the present scenario (PRS) the risk scores ranged from 38.56 ± 27.12 to 58.37 ± 24.82, with the lowest risk for the Larvae Recruitment phase and the highest in the yellow eel maintenance phase (Figure 26).

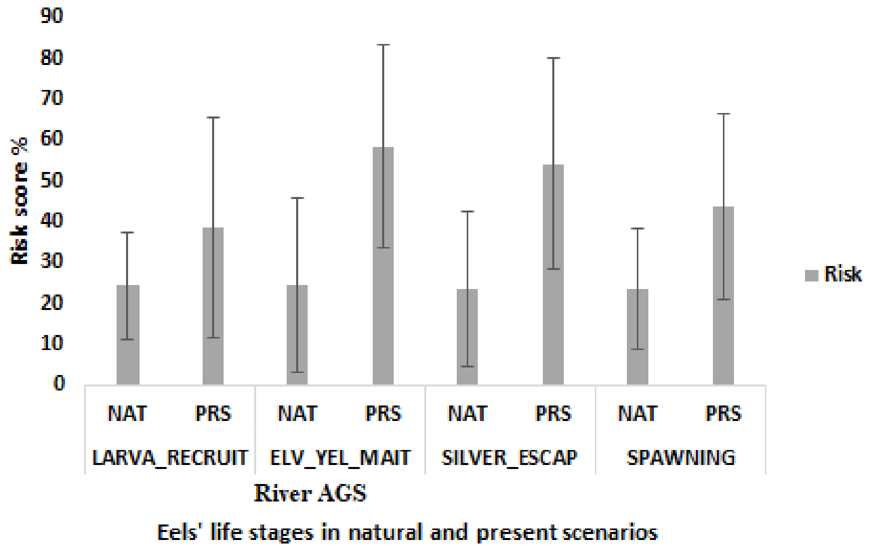


Figure 26: Percentage Risk of Eels' Life Stages in Natural and Present Conditions in the River AGS

5.4.5 Risk distribution in the natural and present scenarios among the eel life stages in River AGS

All life stages in the hypothetical natural scenario (NAT) were dominated by a zero-risk percentage distribution. All life stages in the ecological present scenario (PRS) were dominated by a zero-risk percentage distribution (Figure 27) except in yellow eel maintenance and silver eel escapement, which were dominated by low risk

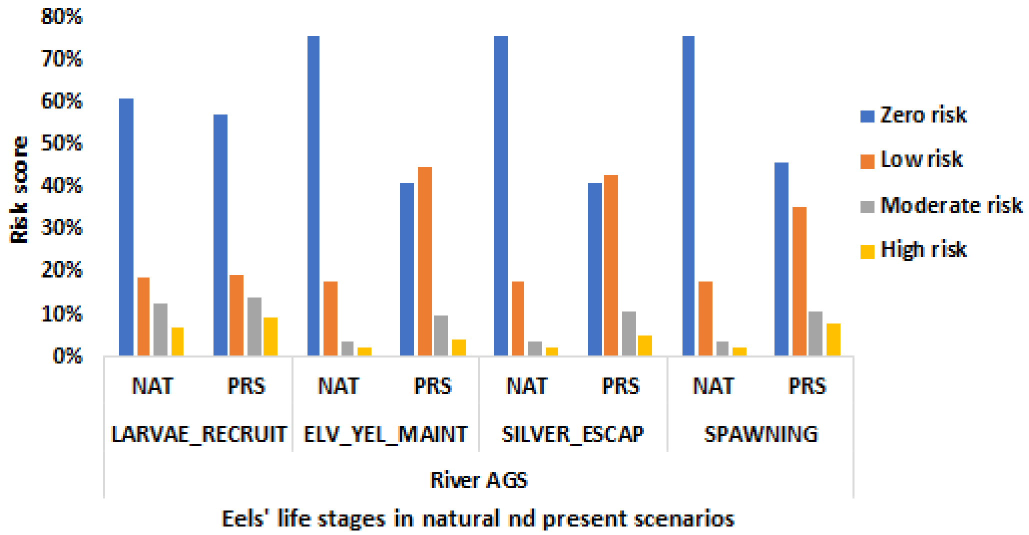


Figure 27: Relative Risk Percentage Distribution for the Hypothetical Natural (NAT) and Present (PRS) Scenarios for the Eel Life Stages at the River AGS Ecosystem

5.4.6 Analysis of risks across eels' life stages in natural, present and future scenarios in River Ramisi

In Future 1 scenario (FUT1), the risk scores ranged from 32.92 ± 27.21 to 46.65 ± 26.96, with the lowest risk for the Larvae Recruitment phase and highest was in yellow eel maintenance phase (Figure 28). In Future 2 scenario (FUT2), the risk scores ranged from 45.299 ± 29.72 to 49.18 ± 22.7, with the lowest risk in Spawning phase and the highest in yellow eel maintenance phase. (Figure 28)

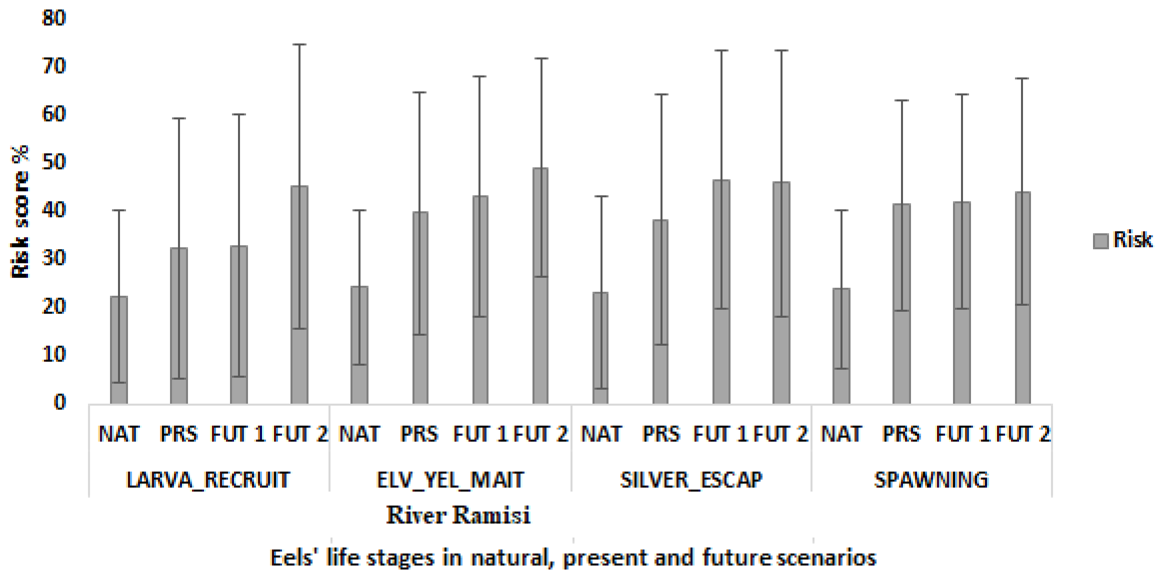


Figure 28: Percentage Risk of Eel Life Stages in Natural, Present and Future Scenarios in the River Ramisi

5.4.7 Risk distribution in the natural, present, and future scenarios among the eels' life stages in River Ramisi

The hypothetical scenario (FUT1) for all life stages in River Ramisi was dominated by a zero-risk percentage distribution except for silver eel escapement, which was dominated by low-risk (Figure 29), The hypothetical Future 2 scenario (FUT2) was dominated by a low-risk distribution across all life stages (Figure 29).

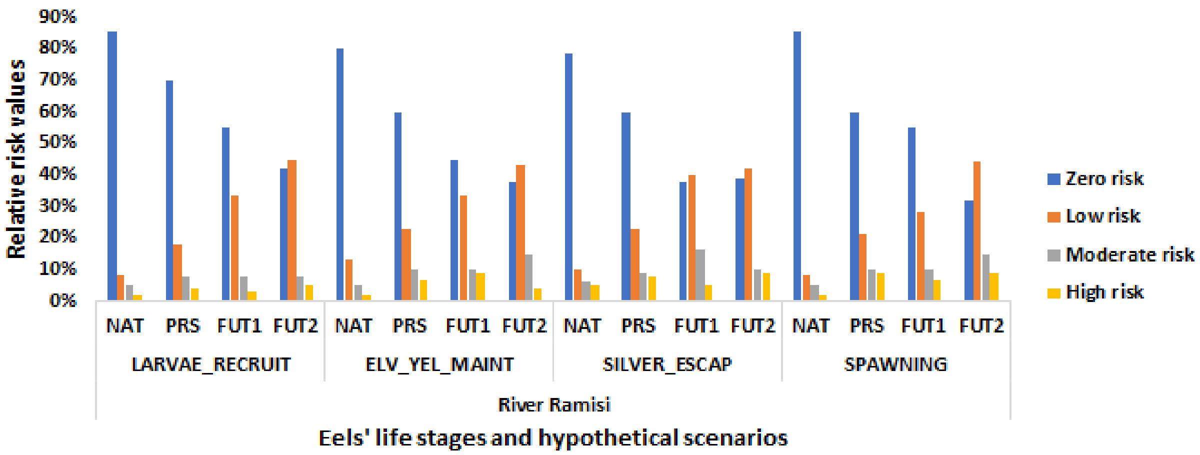


Figure **29:** Relative Risk Percentage Distribution for the Hypothetical Natural (NAT), Present (PRS), and Future Scenarios for the Eel Life Stages at the River Ramisi

5.4.8 Analysis of risks across eel life stages in the natural, present, and future scenarios in the River AGS

Future 1 scenario (FUT1) risk scores in River AGS ranged from 42.99 ± 27.25 to 63.94 ± 25.68, with the lowest risk in the Larvae Recruitment phase and yellow eel maintenance phase having the highest risk (Figure 30). Future 2 (FUT2) scenario risk scores ranged from 44.10 ± 24.14 to 71.7 ± 21.9, with the lowest risk in the Spawning phase and the highest score in the silver el escapement phase, respectively (Figure 30).

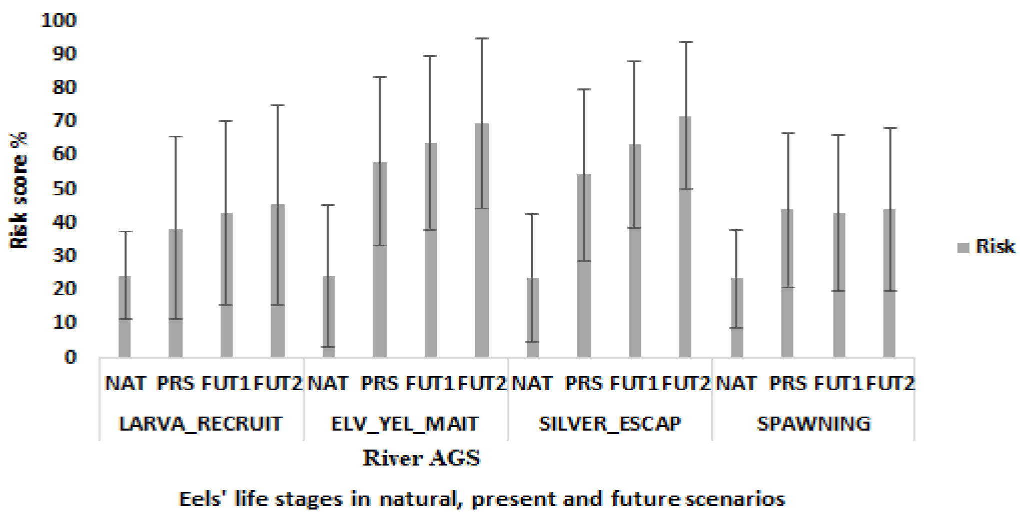


Figure 30: Percentage Risk of Eel Life Stages in Natural, Present, and Future Conditions in River AGS

5.4.9 Risk distribution in the natural, present, and future scenarios among the eels' life stages in River AGS

The hypothetical future scenario (FUT1) (Figure 31) had a low-risk percentage distribution for all life stages except in spawning, which was dominated by a zero-risk distribution. The hypothetical FUT2 scenario (Figure 31) had a low-risk percentage distribution for larval recruitment and spawning phase, whereas yellow eel maintenance and silver eel escapement portrayed a moderate-risk distribution

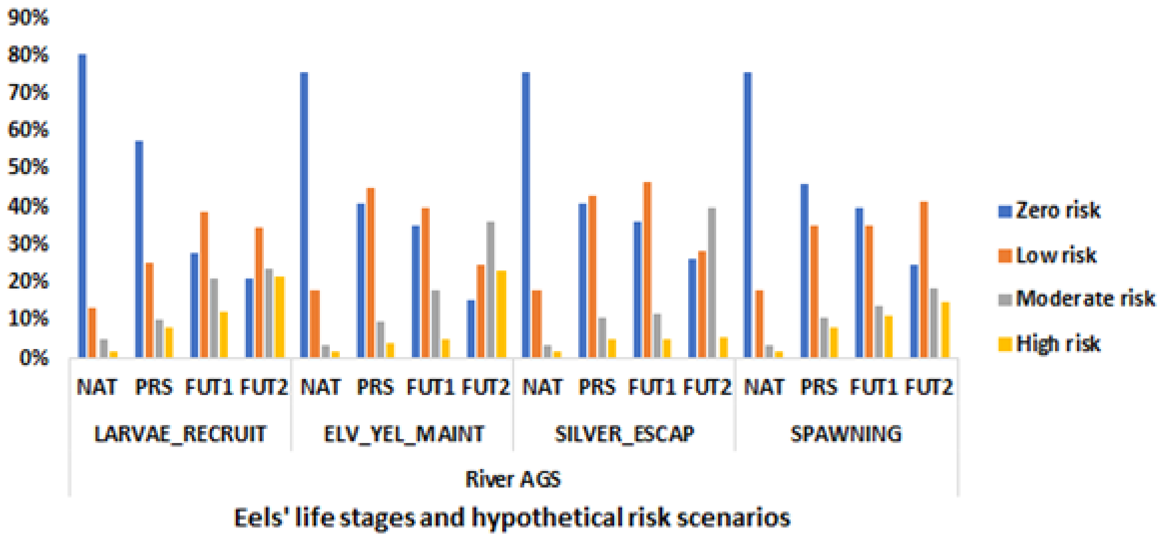


Figure 31: Relative Risk Percentage Distribution for the Hypothetical Natural (NAT), Present (PRS), and Future Scenarios for the Eel Life Stages at the River AGS Ecosystem

5.5 Discussion

This is the first Relative Risk Assessment (RRA) study on Anguillid eels, therefore, literature on similar studies elsewhere is lacking. The findings highlight the varying degrees of risk across different eel life stages in River AGS and River Ramisi, providing critical insights into anguillid eels' ecological challenges in Kenya. The risk scores across natural, present, and future scenarios indicate the impact of environmental conditions on the well-being of eels at different life stages, with significant implications for conservation and management strategies.

In the natural or pristine scenario, both River AGS and River Ramisi show relatively low-risk scores across all life stages, indicating that under undisturbed conditions, the eel populations would face minimal ecological threats. This suggests that historically, these rivers would have provided suitable habitats for the various life stages of eels, supporting healthy population dynamics. The slightly higher risk score for the silver eel escapement phase in River Ramisi, and the yellow eel maintenance phase in River AGS, could be indicative of natural environmental challenges, such as predation or natural barriers, that these life stages may have encountered.

In the present scenario, risk scores increase, particularly in the yellow eel maintenance phase for both rivers, reflecting the impact of ongoing anthropogenic activities such as habitat degradation, water pollution, and barriers to migration. In River AGS, the escalation in risk scores highlights the significant environmental stressors currently affecting eel populations, with the yellow eel maintenance phase being particularly vulnerable. This stage is critical for the growth and maturation of eels, and an increased risk could lead to a decline in population resilience, eel maturity, and recruitment.

For River Ramisi, the increase in risk scores in the present scenario, though less severe than in River AGS, indicates that even relatively undisturbed environments are not immune to human impacts. The highest risk during the yellow eel maintenance phase suggests that factors such as habitat fragmentation and water quality deterioration are becoming more pronounced, threatening the sustainability of eel populations.

The future scenarios paint a more concerning picture, especially for River AGS. In Future 1 (FUT1), the risk scores for the yellow eel maintenance and silver eel escapement phase reach moderate levels, and in Future 2 (FUT2), the risk scores for the phase escalate. These scenarios assume worsening environmental conditions due to increased urbanization, climate change, and intensified agricultural practices. The elevated risk in these future scenarios indicates that, without significant intervention, the survival of eels, particularly in River AGS, could be severely compromised. The moderate risk for yellow eel maintenance and silver eel escapement in the FUT2 scenario suggests that these life stages will be under severe pressure, potentially leading to a significant decline in eel populations if mitigation measures are not implemented.

In contrast, ecological risks in the River Ramisi's future scenarios show a less drastic increase in risk, with FUT2 indicating a shift from zero to low-risk distribution across all life stages. However, the trend suggests that even this relatively undisturbed river could face mounting challenges in the future, necessitating proactive conservation efforts to maintain its ecological integrity.

The findings underline the urgent need for targeted conservation strategies in Kenya to address the varying risks faced by eel populations. The results suggest that without intervention, the future of anguillid eels, particularly in River AGS, is at significant risk. Strategies should focus on mitigating habitat degradation, improving water quality, and removing barriers to migration. Additionally, conservation efforts in River Ramisi should aim to preserve its current state while addressing emerging threats to prevent the escalation of risks in the future. By prioritizing these actions, Kenya can safeguard its eel populations, ensuring their continued ecological and economic importance.

5.6 Conclusion

In conclusion, this study underscores the necessity of conducting risk assessments to evaluate the conservation status of Anguillid eels in the Athi-Galana-Sabaki (AGS) and Ramisi rivers, providing crucial insights for conservation planning. Despite the hypothesis that there is no difference in ecological risks against Anguillid eels in these rivers, the findings indicate distinct risk levels influenced by anthropogenic impacts and habitat conditions. River AGS presents higher risk levels due to significant habitat degradation, water pollution, and barriers to migration, while River Ramisi, though experiencing lower current risks, is projected to face increasing threats in future scenarios.

The study highlights that different life stages of Anguillid eels are subjected to varying levels of risk, with the yellow eel maintenance and silver eel escapement phases being particularly vulnerable. This suggests that conservation strategies must be tailored to address these critical life stages. Additionally, scenario analyses reveal that if current trends persist, risks to eel populations, especially in River AGS, will escalate, potentially leading to severe population declines. The increasing risk scores for River Ramisi in future projections emphasize the importance of proactive conservation even in relatively undisturbed environments.

These findings reinforce the need for targeted conservation interventions, including habitat restoration, water quality improvement, and ensuring unobstructed migration routes. The use of Relative Risk Assessment (RRA) as a tool for ecological risk evaluation has proven effective in this study, demonstrating its value in conservation planning and policy-making. Future research should focus on long-term monitoring and adaptive management strategies to mitigate risks and ensure the sustainability of anguillid eel populations in Kenya and similar riverine ecosystems worldwide.

CHAPTER SIX

GENERAL DISCUSSION, CONCLUSIONS, AND RECOMMENDATIONS

6.1 Discussion

The study analysed the population structure of anguillids in the Athi-Galana-Sabaki and Ramisi rivers. This study involved examining factors such as population density, size distribution, age structure, and species diversity. Through survey methods using sampling fyke nets, the research gave insights into the demographic characteristics of anguillid populations in both river systems.

The study explored the ecological niche of Anguillids in the Athi-Galana-Sabaki and Ramisi rivers, encompassing aspects such as habitat preferences, trophic interactions, and ecological roles. Through habitat surveys, diet analysis, and ecological modelling, factors shaping the distribution and behaviour of Anguillids within their respective river ecosystems were elucidated.

The study aimed to determine the conservation status of Anguillids in the Athi-Galana-Sabaki and Ramisi rivers and develop appropriate conservation interventions based on the findings. This involved assessing threats, evaluating population trends, and identifying priority areas for conservation action.

The integrated analysis of population structure, ecological niche, and conservation status provides a holistic understanding of anguillid populations in the Athi-Galana-Sabaki and Ramisi rivers. This knowledge serves as a foundation for evidence-based conservation management aimed at preserving anguillid biodiversity and ecosystem integrity.

By implementing targeted conservation interventions informed by scientific research, stakeholders can mitigate threats, enhance habitat quality, and promote the sustainable management of anguillid populations. Collaboration between scientists, policymakers, local communities, and conservation organizations is crucial for achieving long-term conservation goals and ensuring the persistence of anguillids in these river systems.

6.2 Conclusions

The findings from this study collectively highlight the critical importance of understanding the population structure, ecological niches, and ecological risks facing anguillid eels in Kenya's river systems. The population assessment revealed significant differences between anguillid eel populations in the River Athi-Galana-Sabaki (AGS) and the River

Ramisi, refuting the null hypothesis that no difference exists in their population structures. Variations in eel length, biomass distribution, and habitat modifiers underscore the influence of environmental factors and anthropogenic activities on eel populations, emphasizing the need for river-specific conservation strategies.

Additionally, the ecological niche assessment demonstrated distinct differences in dietary habits, trophic interactions, and habitat preferences among anguillid species in AGS and Ramisi rivers. The dominance of fish in eel diets, coupled with variations in niche breadth and habitat selection, challenges the assumption of uniform ecological niches between the two rivers. The study underscores the importance of localized management approaches that account for species-specific and river-specific ecological dynamics.

Furthermore, the evaluation of ecological risks highlights varying degrees of anthropogenic pressures on eel populations, disproving the null hypothesis of uniform risk exposure. Differences in habitat integrity, barriers to movement, and resource availability indicate that conservation efforts must be tailored to mitigate specific threats in each river system.

Overall, these findings underscore the urgent need for targeted conservation interventions to safeguard anguillid eel populations and their habitats. By informing adaptive management strategies, this study contributes to the sustainable conservation of eel biodiversity and freshwater ecosystems, ensuring the ecological balance and livelihood support functions of these critical aquatic habitats.

6.3 Recommendations

6.3.1. Habitat restoration and protection

Immediate restoration of degraded catchments, especially in the Athi-Galana-Sabaki (AGS) River, must be prioritized. riparian zones and wetlands, such as those encroached upon by urban developments like Jam City Estate, should be rehabilitated and legally protected. Strict enforcement of land use regulations must ensure that designated forestry and wetland areas are not repurposed for urbanization. Additionally, pollution control measures should be strengthened by enforcing environmental impact assessments, conducting regular industry audits, and implementing stringent waste management policies to improve water quality.

6.3.2. Fish passage improvement

All physical barriers obstructing eel migration must be addressed through the installation of effective fish passage systems. Relevant agencies must collaborate to construct and enhance fish ladders, weirs, and bypass channels to facilitate free movement of eels, particularly for silver eel escapement and recruitment phases. Any future river infrastructure projects must integrate fish-friendly designs to minimize disruption to eel migration routes.

6.3.3. Regulatory and policy interventions

Policymakers must enact and enforce strict regulations to curb habitat destruction, pollution, and unregulated water abstraction in eel habitats. Although eels are not currently overexploited in Kenya, proactive regulations must be implemented to prevent future threats. These should include the establishment of fishing quotas, seasonal protection measures, and habitat conservation mandates to secure eel populations for posterity.

6.3.4. Community engagement and awareness

Local communities must be actively involved in eel conservation initiatives. Government agencies and conservation organizations should develop targeted educational programs to raise awareness among riverine communities, fishermen, and policymakers about the ecological importance of eels. Sustainable fishing practices should be promoted, alongside alternative livelihoods such as aquaculture and ecotourism, to reduce dependency on wild eel populations. Engaging indigenous communities and integrating their traditional knowledge into conservation planning will enhance the effectiveness of these efforts.

6.3.5. Research, monitoring, and data-driven management

Long-term monitoring programs must be established to assess eel population trends, habitat conditions, and the effectiveness of conservation interventions. Government agencies and academic institutions should invest in research to further understand eel population dynamics, ecological niches, and the impacts of environmental disturbances. Specific focus should be placed on the factors influencing recruitment success, growth variations, and species interactions in different habitats. Regular risk assessments should be conducted to anticipate emerging threats and adapt conservation strategies accordingly.

6.3.6. Regional and international collaboration

Given the catadromous nature of Anguillid eels, Kenya must actively engage in regional and international conservation efforts. Cross-border collaboration with neighbouring countries and international organizations is essential to address transboundary issues such as habitat degradation, illegal fishing, and trade regulations. Participation in global eel conservation initiatives and information-sharing on best practices will strengthen conservation efforts.

6.3.7. Climate adaptation and habitat connectivity

Climate change adaptation strategies must be developed and integrated into conservation plans. Efforts should focus on restoring habitat connectivity to allow eels to respond to changing environmental conditions. This includes maintaining adequate water flow in rivers, protecting breeding and feeding grounds, and implementing adaptive management strategies to mitigate climate-induced habitat shifts.

6.3.8. Areas for further research

Despite the progress made in understanding Anguillid eel population structures, ecological niches, and associated risks, key knowledge gaps remain. Future research should investigate:

i. The mechanisms driving population fluctuations and declines.
ii. The effectiveness of conservation measures, particularly habitat restoration and fish passage systems.
iii. The impacts of climate change on eel migration and habitat suitability.
iv. Innovative and sustainable approaches to eel management and restoration.

REFERENCES

Aarestrup, K., Økland, F., Hansen, M. M., Righton, D., Gargan, P., Castonguay, M., & McKinley, R. S. (2009). Oceanic spawning migration of the European eel (*Anguilla anguilla*). *Science, 325*(5948), 1660. https://doi.org/10.1126/science.1178120

Adam, G., Feunteun, E., Prouzet, P., & Rigaud, C. (2008). The European eel: Indicators of abundance and colonization. *Cybium, 32*(2), 63–66.

Abdoli, L., Kamrani, E., Abdoli, A., & Kiabi, B. (2009). Length-weight relationships for three species of mudskippers (Gobiidae: Oxudercinae) in the coastal areas of the Persian Gulf, Iran. *Journal of Applied Ichthyology, 25*(2), 236-237. https://doi.org/10.1111/j.1439-0426.2009.01287.x

Acolas, M. L., Le Pichon, C., Rochard, E., & Lambert, P. (2020). Effects of river fragmentation on European eel populations. *Freshwater Biology, 65*(4), 724–739. https://doi.org/10.1111/fwb.13461

Acou, A., Laffaille, P., Legault, A., & Feunteun, E. (2008). Migration pattern of silver eels in an obstructed river system. *Ecology of Freshwater Fish, 17*(3), 432–442. https://doi.org/10.1111/j.1600-0633.2008.00288.x

Acou, A., Laffaille, P., Legault, A., & Feunteun, E. (2020). Influence of anthropogenic stressors on Anguillid eel populations: A global perspective. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *30*(4), 567–583. https://doi.org/10.1002/aqc.3274

Adams, S. M., Kimmel, B. L., & Ploskey, G. R. (1983). Sources of organic matter for reservoir fish production: A trophic-dynamics analysis. *Canadian Journal of Fisheries and Aquatic Sciences*, *40*(9), 1480-1495. https://doi.org/10.1139/f83-181

Aedo, J. E., Vargas, J. A., & Soto, D. (2020). Impacts of altered flow regimes on freshwater biodiversity. *Ecological Indicators, 117,* 106674. https://doi.org/10.1016/j.ecolind.2020.106674

Alberdi, A., Razgour, O., Aizpurua, O., Novella-Fernandez, R., Aihartza, J., Budinski, I., & Gilbert, M. T. P. (2020). DNA metabarcoding and spatial modelling link diet diversification with distribution homogeneity in European bats. *Nature Communications, 11*(1), 1154. https://doi.org/10.1038/s41467-020-14931-2

Albertoni, E. F., Palma-Silva, C., & Esteves, F. D. A. (2003). Natural diet of three species of shrimp in a tropical coastal lagoon. *Brazilian Archives of Biology and Technology, 46*, 395-403. https://doi.org/10.1590/S1516-89132003000300002

Alexandre, C. M., Almeida, P. R., Neves, T., Mateus, C. S., Costa, J. L., & Quintella, B. R. (2016). Effects of flow regulation on the movement patterns and habitat use of a potamodromous cyprinid species (*Luciobarbus bocagei*) in a Mediterranean-type river. *Ecology of Freshwater Fish*, *25*(3), 405–417. https://doi.org/10.1111/eff.12223

Allan, J. D., & Castillo, M. M. (2007). *Stream ecology: Structure and function of running waters* (2nd ed.). Springer. https://doi.org/10.1007/978-1-4020-5583-6

Als, T. D., Hansen, M. M., Maes, G. E., Castonguay, M., Riemann, L., Aarestrup, K., & Bernatchez, L. (2011). All roads lead to home: Panmixia of European eel in the Sargasso Sea. *Molecular Ecology*, *20*(7), 1333–1346. https://doi.org/10.1111/j.1365-294X.2011.05011.x

Antunes, C. (2008). Report on the eel stock and fishery in Portugal 2008. In *Report of the 2008 session of the Joint EIFAC/ICES Working Group on Eels* (pp. 3–9).

Aoyama, J., Shinoda, A., Yoshinaga, T., & Tsukamoto, K. (2012). Late arrival of *Anguilla japonica* glass eels at the Sagami River estuary in two recent consecutive year classes: Ecology and socio-economic impacts. *Fisheries Science, 78*, 1195-1204. https://doi.org/10.1007/s12562-012-0514-3

Aoyama, J., Wouthuyzen, S., Miller, M. J., Sugeha, H. Y., Kuroki, M., Watanabe, S., & Tsukamoto, K. (2018). Reproductive ecology and biodiversity of freshwater eels around Sulawesi Island, Indonesia. *Zoological Studies, 57*, Article 30. https://doi.org/10.6620/ZS.2018.57-30

Arai, T. (2014). Do we protect freshwater eels or do we drive them to extinction? *SpringerPlus, 3*, 1-10. https://doi.org/10.1186/2193-1801-3-124

Arai, T. (2014). Evidence of local short- distance spawning migration of tropical freshwater eels, and implications for the evolution of freshwater eel migration. *Ecology and Evolution, 4*(19), 3812-3819. https://doi.org/10.1002/ece3.1205

Arai, T. (2016). *Biology and ecology of anguillid eels* (Vol. 340). CRC Press.

Arai, T. (2020). Ecology and evolution of migration in the freshwater eels of the genus *Anguilla* Schrank, 1798. *Heliyon, 6*(10), e05160. https://doi.org/10.1016/j.heliyon.2020.e05160

Arai, T., & Abdul Kadir, S. R. (2017). Diversity, distribution and different habitat use among the tropical freshwater eels of the genus *Anguilla*. *Scientific Reports, 7*(1), 7593. https://doi.org/10.1038/s41598-017-08142-1

Arai, T., & Chino, N. (2018). Opportunistic migration and habitat use of the giant mottled eel *Anguilla marmorata* (Teleostei: Elopomorpha). *Scientific Reports, 8*, 5666. https://doi.org/10.1038/s41598-018-24029-5

Arai, T., Chai, I. J., Iizuka, Y., & Chang, C. W. (2020). Habitat segregation and migration in tropical anguillid eels, *Anguilla bengalensis bengalensis* and *A. bicolor bicolor*. *Scientific Reports, 10*(1), 16890. https://doi.org/10.1038/s41598-020-73979-1

Arribas, C., Fernández-Delgado, C., Oliva-Paterna, F. J., & Drake, P. (2012). Oceanic and local environmental conditions as forcing mechanisms of the glass eel recruitment to the southernmost European estuary. *Estuarine, Coastal and Shelf Science, 107*, 46–57. https://doi.org/10.1016/j.ecss.2012.04.021

Arthington, A. H., Kennen, J. G., Stein, E. D., & Webb, J. A. (2018). Environmental flows: Ecological importance and management challenges. *Aquatic Sciences, 80*(4), 401–419. https://doi.org/10.1007/s00027-018-0570-z

Arunachalam, M. (2000). Assemblage structure of stream fishes in the Western Ghats (India). *Hydrobiologia, 430*, 1–31. https://doi.org/10.1023/A:1004080829387

Aschonitis, V., Castaldelli, G., Lanzoni, M., Rossi, R., Kennedy, C., & Fano, E. A. (2017). Long-term records (1781–2013) of European eel (*Anguilla anguilla* L.) production in the Comacchio Lagoon (Italy): Evaluation of local and global factors as causes of the population collapse. *Aquatic Conservation: Marine and Freshwater Ecosystems, 27*(2), 502–520. https://doi.org/10.1002/aqc.2700

Aubone, N., Palma, E. D., & Piola, A. R. (2021). The surface salinity maximum of the South Atlantic. *Progress in Oceanography, 191*, 102499. https://doi.org/10.1016/j.pocean.2020.102499

Bănăduc, D., Simić, V., Cianfaglione, K., Barinova, S., Afanasyev, S., Öktener, A., ... & Curtean-Bănăduc, A. (2022). Freshwater as a sustainable resource and generator of secondary resources in the 21st century: Stressors, threats, risks, management and protection strategies, and conservation approaches. *International Journal of Environmental Research and Public Health, 19*(24), Article 16570. https://doi.org/10.3390/ijerph192416570

Baras, E., Jeandrain, D., & Philippart, J. C. (1998). Spatial distribution of the European eel (*Anguilla anguilla*) in a small tributary of the Meuse River, Belgium. *Environmental Biology of Fishes, 51*(1), 73–82. https://doi.org/10.1023/A:1007485201626

Bardonnet, A., & Acolas, M. L. (2021). Ecology of anguillid eels: Recent advances and perspectives. *Journal of Fish Biology, 99*(3), 579–599. https://doi.org/10.1111/jfb.14737

Bardonnet, A., & Baglinière, J. L. (2018). Freshwater eels: Migration and habitat selection. *Journal of Fish Biology, 93*(2), 219–236. https://doi.org/10.1111/jfb.13768

Bardonnet, A., & Riera, P. (2021). Habitat use and ecological requirements of Anguillid eels. *Aquatic Conservation: Marine and Freshwater Ecosystems, 31*(5), 985–1001. https://doi.org/10.1002/aqc.3551

Bark, A., Williams, B., & Knights, B. (2007). Current status and temporal trends in stocks of European eel in England and Wales. *ICES Journal of Marine Science, 64*(7), 1368–1378. https://doi.org/10.1093/icesjms/fsm083

Basset, A., Barbone, E., Elliott, M., Li, B. L., Jorgensen, S. E., Lucena-Moya, P., & Mouillot, D. (2013). A unifying approach to understanding transitional waters: Fundamental properties emerging from ecotone ecosystems. *Estuarine, Coastal and Shelf Science, 132*, 5–16. https://doi.org/10.1016/j.ecss.2013.03.020

Baumgartner, L. J., Boys, C. A., Stuart, I. G., & Zampatti, B. P. (2020). Hydrological connectivity and fish migration in modified rivers. *River Research and Applications, 36*(3), 312–324. https://doi.org/10.1002/rra.3582

Becerra-Jurado, G., Cruikshanks, R., O'Leary, C., Kelly, F., Poole, R., & Gargan, P. (2014). Distribution, prevalence and intensity of *Anguillicola crassus* (Nematoda) infection in *Anguilla anguilla* in the Republic of Ireland. *Journal of Fish Biology, 84*(4), 1046–1062. https://doi.org/10.1111/jfb.12356

Béguer-Pon, M., Castonguay, M., Shan, S., Benchetrit, J., & Dodson, J. J. (2015). Direct observations of American eels migrating across the continental shelf to the Sargasso Sea. *Nature Communications, 6,* 8705. https://doi.org/10.1038/ncomms9705

Béguer-Pon, M., Dodson, J. J., Castonguay, M., Jellyman, D., Aarestrup, K., & Tsukamoto, K. (2017). Tracking anguillid eels: Five decades of telemetry-based research. *Marine and Freshwater Research, 69*(2), 199–219. https://doi.org/10.1071/MF16318

Behrmann-Godel, J., & Eckmann, R. (2003). Brown trout spawning migration in fragmented central European headwaters: Effect of isolation by artificial obstacles and the moon phase. *Transactions of the American Fisheries Society, 132*(4), 759–768. https://doi.org/10.1577/T02-122:contentReference[oaicite:1] {index=1}

Bell-Cross, G., & Minshull, J. L. (1988). *The fishes of Zimbabwe*. National Museums and Monuments of Zimbabwe. Retrieved from https://bit.ly/32UihoS

Belpaire, C., & Goemans, G. (2007). Eels: Contaminant cocktails pinpointing environmental contamination. *ICES Journal of Marine Science, 64*(7), 1423–1436. https://doi.org/10.1093/icesjms/fsm099

Belpaire, C., Geeraerts, C., Evans, D., Ciccotti, E., & Polet, H. (2016). Contaminants in eels and their impact on recruitment. *Aquatic Toxicology, 176*, 17–26. https://doi.org/10.1016/j.aquatox.2015.12.003

Belpaire, C., Pujolar, J. M., Geeraerts, C., & Maes, G. E. (2016). Contaminants in eels and their role in the collapse of the eel stocks. In T. Arai (Ed.), *Biology and ecology of anguillid eels* (pp. 225–250). CRC Press.

Benejam, L., Saura-Mas, S., Bardina, M., Solà, C., Munné, A., & García-Berthou, E. (2016). Ecological impacts of water abstraction and hydrological alteration in Mediterranean streams. *Science of the Total Environment, 572*, 1057–1066. https://doi.org/10.1016/j.scitotenv.2016.07.197

Benson, T., De Bie, J., Gaskell, J., Vezza, P., Kerr, J. R., Lumbroso, D., Owen, M. R., & Kemp, P. S. (2021). Agent-based modelling of juvenile eel migration via selective tidal stream transport. *Ecological Modelling, 443,* 109448. https://doi.org/10.1016/j.ecolmodel.2021.109448

Bernotas, P., Öğlü, B., & Nõges, P. (2020). How do environmental factors affect the yield of European eel (*Anguilla anguilla*) in a restocked population? *Fisheries Research, 230,* 105649. https://doi.org/10.1016/j.fishres.2020.105649

Bevacqua, D., Roussel, J. M., & Rivot, E. (2014). Environmental factors affecting eel migration in European rivers. *Journal of Fish Biology, 84*(6), 1904-1923. https://doi.org/10.1111/jfb.12444

Bice, C. M., Huisman, J., Kimball, M. E., Mallen-Cooper, M., Zampatti, B. P., & Gillanders, B. M. (2023). Tidal barriers and fish – Impacts and remediation in the face of increasing demand for freshwater and climate change. *Estuarine, Coastal and Shelf Science, 289,* 108376. https://doi.org/10.1016/j.ecss.2023.108376

Bilby, R. E., & Bisson, P. A. (1992). Allochthonous versus autochthonous organic matter contributions to the trophic support of fish populations in forested streams. *Canadian Journal of Fisheries and Aquatic Sciences, 49*(3), 540–551. https://doi.org/10.1139/f92-064

Bolliet, V., Claveau, J., Jarry, M., Gonzalez, P., Baudrimont, M., & Monperrus, M. (2017). Migratory behaviour, metabolism, oxidative stress, and mercury concentrations in marine and estuarine European glass eels (*Anguilla anguilla*). *Physiology & Behavior, 169*, 33–40. https://doi.org/10.1016/j.physbeh.2016.11.008

Bonhommeau, S., Chassot, E., Planque, B., Rivot, E., Knap, A. H., & Le Pape, O. (2008). Impact of climate on eel populations of the Northern Hemisphere. *Marine Ecology Progress Series, 373*, 71–80. https://doi.org/10.3354/meps07696

Bonhommeau, S., Le Pape, O., Gascuel, D., Blanke, B., Tréguier, A. M., Grima, N., & Rivot, E. (2009). Estimates of the mortality and the duration of the trans-Atlantic migration of European eel *Anguilla anguilla* leptocephali using a particle tracking model. *Journal of Fish Biology, 74*(9), 1891–1914. https://doi.org/10.1111/j.1095-8649.2009.02298.x

Boston, C. M., Randall, R. G., Hoyle, J. A., Mossman, J. L., & Bowlby, J. N. (2016). The fish community of Hamilton Harbour, Lake Ontario: Status, stressors, and remediation over 25 years. *Aquatic Ecosystem Health & Management, 19*(2), 206–218. https://doi.org/10.1080/14634988.2016.1183203

Braga, R. R., Bornatowski, H., & Vitule, J. R. S. (2012). Feeding ecology of fishes: An overview of worldwide publications. *Reviews in Fish Biology and Fisheries, 22*(4), 915–929. https://doi.org/10.1007/s11160-012-9262-8

Breckling, B., Drechsler, M., & Ristau, K. (2020). Ecological consequences of riverbed degradation on freshwater biodiversity. *Freshwater Biology, 65*(7), 1132–1148. https://doi.org/10.1111/fwb.13493

Briand, C., Bonhommeau, S., Beaulaton, L., & Castelnaud, G. (2009). An appraisal of historical glass eel fisheries and markets: Landings, trade routes and future prospect for management. In *The Institute of Fisheries Management Annual Conference* (Vol. 21).

Briand, C., Fatin, D., & Fontenelle, G. (2022). Trophic alterations in Anguillid eels due to habitat modification. *Hydrobiologia, 849*(2), 431–449. https://doi.org/10.1007/s10750-021-04681-z

Briand, C., Fatin, D., Fontenelle, G., & Lambert, P. (2018). Fish passage solutions for eels: Advances and challenges. *Fisheries Management and Ecology, 25*(2), 129–140. https://doi.org/10.1111/fme.12275

Briand, C., Sauvaget, B., Girard, P., Fatin, D., & Beaulaton, L. (2012). Push net fishing seems to be responsible for injuries and post fishing mortality in glass eel in the Vilaine estuary (France) in 2007. *Knowledge and Management of Aquatic Ecosystems, 404*, 02. https://doi.org/10.1051/kmae/2012002

Broadmeadow, S. B., & Nisbet, T. R. (2004). The effects of riparian forest management on the freshwater environment: A literature review of best management practices. *Hydrology and Earth System Sciences, 8*(3), 286–305. https://doi.org/10.5194/hess-8-286-2004

Brown, C. J., Saunders, M. I., Possingham, H. P., & Richardson, A. J. (2013). Managing for interactions between local and global stressors of ecosystems. *PLOS ONE, 8*(6), e65424. https://doi.org/10.1371/journal.pone.0065424

Browne, M. A., Galloway, T. S., & Thompson, R. C. (2011). Accumulation of microplastic on shorelines worldwide: Sources and sinks. *Environmental Science & Technology, 45*(21), 9175–9179. https://doi.org/10.1021/es201811s

Bru, N., Prouzet, P., & Lejeune, M. (2009). Daily and seasonal estimates of the recruitment and biomass of glass eel runs (*Anguilla anguilla*) and exploitation rates in the Adour open estuary (Southwestern France). *Aquatic Living Resources, 22*(4), 509–523. https://doi.org/10.1051/alr/2009044

Bunn, S. E., & Arthington, A. H. (2002). Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management, 30*(4), 492–507. https://doi.org/10.1007/s00267-002-2737-0

Cadiz, R., & Trafalgar, R. F. (2020). Optimum salinity for growth of tropical eel *Anguilla marmorata* Quoy & Gaimard, 1824 in nursery culture. *Asian Fisheries Science, 33*, 315-320.

Cairns, D. K., Avery, T. S., Benchetrit, J., Bornarel, V., Casselman, J. M., Castonguay, M., Crow, S. K., Dorow, M., Drouineau, H., Frankowski, J., Galbraith, H. S., Haro, A., Hoyle, S. D., Knickle, D. C., Koops, M. A., Poirier, L. A., Rudd, M. B., Thorson, J. T., Williams, E. K., Young, J., & Zhu, X. (2021). *Methods and data sources to support American eel population analysis*. Canadian Science Advisory Secretariat (CSAS).

Cairns, D. K., Benchetrit, J., Bernatchez, L., Bornarel, V., Casselman, J. M., Castonguay, M., & Zhu, X. (2022). Thirteen novel ideas and underutilized resources to support progress towards a range-wide American eel stock assessment. *Fisheries Management and Ecology, 29*(5), 516–541. https://doi.org/10.1111/fme.12572

Cairns, D. K., Chaput, G., Poirier, L. A., Avery, T. S., Castonguay, M., Mathers, A., & Clarke, K. D. (2014). *Recovery potential assessment for the American eel (Anguilla rostrata) for eastern Canada: Life history, distribution, reported landings, status indicators, and demographic parameters* (Research Document 2013/134). Canadian Science Advisory Secretariat. https://waves-vagues.dfo-mpo.gc.ca/Library/358302.pdf

Calles, O., Karlsson, S., Hebrand, M., & Gustavsson, S. (2013). Mortality of silver eels at hydropower stations. *Ecology of Freshwater Fish, 22*(1), 106–116. https://doi.org/10.1111/eff.12043

Cárcamo, C., Estrada, J. M., Canales-Cerro, C., Hernandez, M. F., Herrera, R., Molina-Burgos, B. E., & Klarian, S. A. (2019). Evaluating the relationship between the growth of fish and energy component of their prey. *Cogent Environmental Science, 5*(1), 1609225. https://doi.org/10.1080/23311843.2019.1609225

Chaput, G. J., Pratt, T. C., Cairns, D. K., Clarke, K. D., Bradford, R. G., Mathers, A., & Verreault, G. (2014). *Recovery potential assessment for the American eel (Anguilla rostrata) for eastern Canada: Description and quantification of threats* (Research Document 2013/134). Canadian Science Advisory Secretariat. https://publications.gc.ca/collections/collection_2014/mpo-dfo/Fs70-4-2013-134-eng.pdf

Chen, J. Z., Huang, S. L., & Han, Y. S. (2014). Impact of long-term habitat loss on the Japanese eel *Anguilla japonica*. *Estuarine, Coastal and Shelf Science, 151*, 361–369. https://doi.org/10.1016/j.ecss.2014.04.001

Chen, Q., Li, Q., Lin, Y., Zhang, J., Xia, J., Ni, J., & Tang, L. (2023). River damming impacts on fish habitat and associated conservation measures. *Reviews of Geophysics*, *61*(4). https://doi.org/10.1029/2023RG000819

Cheung, W. W., Lam, V. W., Sarmiento, J. L., Kearney, K., Watson, R. E. G., Zeller, D., & Pauly, D. (2010). Large- scale redistribution of maximum fisheries catch potential in the global ocean under climate change. *Global Change Biology, 16*(1), 24–35. https://doi.org/10.1111/j.1365-2486.2009.01995.x

Chevillot, X., Drouineau, H., Lambert, P., Carassou, L., Sautour, B., & Lobry, J. (2017). Toward a phenological mismatch in estuarine pelagic food web? *PLOS ONE*, *12*(3), e0173752. https://doi.org/10.1371/journal.pone.0173752

Chilton, K. D., & Spotila, J. A. (2022). Uncovering the controls on fluvial bedrock erodibility and knickpoint expression: A high- resolution comparison of bedrock properties between knickpoints and non- knickpoint reaches. *Journal of Geophysical Research: Earth Surface, 127*(3), e2021JF006511. https://doi.org/10.1029/2021JF006511

Clarke, K. R., Warwick, R. M., & Somerfield, P. J. (2020). The effects of sediment alteration on riverine biodiversity. *Journal of Applied Ecology, 57*(5), 1032–1041. https://doi.org/10.1111/1365-2664.13594

Clarkin, C., O’Hanley, J. R., Wright, J., & Kerr, J. T. (2019). Evaluating fish passage solutions for Anguillid eels. *Journal of Applied Ecology, 56*(6), 1342–1353. https://doi.org/10.1111/1365-2664.13364

Coad, B. W. (2016). Review of the freshwater eels of Iran (Family Anguillidae). *International Journal of Aquatic Biology, 4*(2), 102-107. https://doi.org/10.22034/ijab.2016.453368

Collier, K. J., Probert, P. K., & Jeffries, M. (2016). Conservation of aquatic invertebrates: Concerns, challenges and conundrums. *Aquatic Conservation: Marine and Freshwater Ecosystems, 26*(5), 817-837. https://doi.org/10.1002/aqc.2705

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). (2007). Inclusion of *Anguilla anguilla* (L.) in Appendix II in accordance with Article II s2(a). http://www.cites.org/eng/cop/14/prop/E14-P18.pdf

Cooke, S. J., Bergman, J. N., Twardek, W. M., Piczak, M. L., Casselberry, G. A., Lutek, K., & Lennox, R. J. (2022). The movement ecology of fishes. *Journal of Fish Biology, 101*(4), 756–779. https://doi.org/10.1111/jfb.15153

Correia, M. J., Costa, J. L., Antunes, C., De Leo, G., & Domingos, I. (2018). The decline in recruitment of the European eel: New insights from a 40-year-long time series in the Minho estuary (Portugal). *ICES Journal of Marine Science, 75*(6), 1975–1983. https://doi.org/10.1093/icesjms/fsy086

Corsi, I., Mariottini, M., Caruso, T., & Bonacci, S. (2018). Eco toxicological effects of heavy metals on Anguillid eels. *Environmental Pollution, 239*, 317–329. https://doi.org/10.1016/j.envpol.2018.04.034

Côté, C. L., Gagnaire, P. A., Bourret, V., Verreault, G., Castonguay, M., & Bernatchez, L. (2014). Population genetics of the American eel (*Anguilla rostrata*): FST = 0 and North Atlantic Oscillation effects on demographic fluctuations of a panmictic species. *Molecular Ecology, 22*(7), 1763–1776. https://doi.org/10.1111/mec.12241

Côté, I. M., Green, S. J., & Hixon, M. A. (2013). Predatory fish invaders: Insights from Indo-Pacific lionfish in the western Atlantic and Caribbean. *Biological Conservation, 164*, 50–61. https://doi.org/10.1016/j.biocon.2013.04.014

Couillard, C. M., Verreault, G., Dumont, P., Stanley, D., & Threader, R. W. (2014). Assessment of fat reserves adequacy in the first migrant silver American eels of a large-scale stocking experiment. *North American Journal of Fisheries Management, 34*(4), 802–813. https://doi.org/10.1080/02755947.2014.919878

Crain, C. M., Kroeker, K., & Halpern, B. S. (2008). Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters, 11*(12), 1304-1315. https://doi.org/10.1111/j.1461-0248.2008.01253.x

Cresci, A. (2020). A comprehensive hypothesis on the migration of European glass eels (*Anguilla anguilla*). *Biological Reviews*, *95*(5), 1273–1286. https://doi.org/10.1111/brv.12589

Crivelli, A. J., Auphan, N., Chauvelon, P., Sandoz, A., Menella, J. Y., & Poizat, G. (2008). Glass eel recruitment, *Anguilla anguilla* (L.), in a Mediterranean lagoon assessed by a glass eel trap: Factors explaining the catches. In P. Fontaine, F. Lambert, A. C. Belaud, & G. R. J. L. Dufour (Eds.), *Fish and diadromy in Europe: Ecology, management, conservation* (pp. 79–86). Springer. https://doi.org/10.1007/978-1-4020-8437-0_9

Crook, D. A., Macdonald, J. I., & Douglas, J. W. (2021). Migration disruption in freshwater eels due to riverbed modification. *Environmental Management*, *67*(2), 278–290. https://doi.org/10.1007/s00267-020-01363-4

Crook, D. A., Macdonald, J. I., Morrongiello, J. R., Belcher, C. A., Lovett, D., Walker, A., & Nicol, S. J. (2014). Environmental cues and extended estuarine residence in seaward migrating eels (*Anguilla australis*). *Freshwater Biology*, *59*(8), 1710–1720. https://doi.org/10.1111/fwb.12376

Cullen, J. J., & McCarthy, I. D. (2017). Physiological responses of eels to hypoxia. *Comparative Biochemistry and Physiology Part A*, *205*, 13–22. https://doi.org/10.1016/j.cbpa.2016.12.008

Daverat, F., Beaulaton, L., Danancher, D., Courret, D., & Lambert, P. (2022). Sediment dynamics and habitat degradation in riverine eel populations. *River Research and Applications*, *38*(2), 201–214. https://doi.org/10.1002/rra.3886

Davey, A., & Jellyman, D. (2005). Sex determination in freshwater eels and management options for manipulation of sex. *Reviews in Fish Biology and Fisheries*, *15*(1–2), 37–52. https://doi.org/10.1007/s11160-005-7431-x

Davidson, N. C. (2014). How much wetland has the world lost? Long-term and recent trends in global wetland area. *Marine and Freshwater Research*, *65*(10), 934–941. https://doi.org/10.1071/MF14173

Davies, P., Britton, J. R., Nunn, A. D., Dodd, J. R., Bainger, C., Velterop, R., & Bolland, J. D. (2021). Cumulative impacts of habitat fragmentation and the environmental factors affecting upstream migration in the threatened sea lamprey, *Petromyzon marinus*. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *31*(9), 2560–2574. https://doi.org/10.1002/aqc.3625

Dekker, W. (2019). The history of European eel management. *ICES Journal of Marine Science*, *76*(4), 902–914. https://doi.org/10.1093/icesjms/fsz059

Dekker, W., Casselman, J. M., Cairns, D. K., Tsukamoto, K., Jellyman, D., & Lickers, H. (2003). Quebec Declaration of Concern: Worldwide decline of eel resources necessitates immediate action. *Fisheries, 28*(12), 28–30. https://doi.org/10.1577/1548-8446(2003)28[28: QDOCWD]2.0.CO;2

Denis, J., Lepage, M., Gruselle, M.-C., & Amara, R. (2024). The influence of natural and anthropogenic environmental pressures on European eel abundances in French estuaries. *Fishes, 9*(2), Article 44. https://doi.org/10.3390/fishes9020044

Denis, J., Mahé, K., & Amara, R. (2022). Abundance and growth of the European eels (*Anguilla anguilla* Linnaeus, 1758) in small estuarine habitats from the eastern English Channel. *Fishes, 7*(5), Article 213. https://doi.org/10.3390/fishes7050213

Denis, J., Rabhi, K., Le Loc'h, F., Ben Rais Lasram, F., Boutin, K., Kazour, M., & Amara, R. (2022). Role of estuarine habitats for the feeding ecology of the European eel (*Anguilla anguilla* L.). *PLOS ONE, 17*(7), e0270348. https://doi.org/10.1371/journal.pone.0270348

Denoncourt, C. E., & Stauffer Jr, J. R. (1993). Feeding selectivity of the American eel *Anguilla rostrata* (LeSueur) in the upper Delaware River. *American Midland Naturalist*, 301-308. https://doi.org/10.2307/2426127

Dinh, Q. M., Qin, J. G., Dittmann, S., & Tran, D. D. (2016). Morphometric variation of *Parapocryptes serperaster* (Gobiidae) in dry and wet seasons in the Mekong Delta, Vietnam. *Ichthyological Research, 63*, 267-274. https://doi.org/10.1007/s10228-016-0532-3

Doole, G. J. (2005). Optimal management of the New Zealand longfin eel (*Anguilla dieffenbachii*). *Australian Journal of Agricultural and Resource Economics, 49*(4), 395–411. https://doi.org/10.1111/j.1467-8489.2005.00247.x

Dörner, H., & Benndorf, J. (2003). Piscivory by large eels on young-of-the-year fishes: Its potential as a biomanipulation tool. *Journal of Fish Biology, 62*(2), 491-494. https://doi.org/10.1046/j.1095-8649.2003.00052.x

Dörner, H., & Berg, S. (2016). Feeding ecology. In T. Arai (Ed.), *Biology and ecology of anguillid eels* (pp. 171–191). CRC Press. https://doi.org/10.1201/b19925-9

Dörner, H., & Berg, S. (2016). Feeding habits of freshwater fish: Insights into trophic ecology and predator-prey interactions. *Aquatic Ecology, 50*(4), 651–662. https://doi.org/10.1007/s10452-016-9587-4

Doughty, C. R., Mackey, B., & Bryan, B. A. (2021). The impact of anthropogenic sediment modification on freshwater fish populations. *Global Change Biology*, *27*(4), 659–671. https://doi.org/10.1111/gcb.15457

Drouineau, H., Bau, F., Alric, A., Deligne, N., Gomes, P., & Sagnes, P. (2017). Silver eel downstream migration in fragmented rivers: Use of a Bayesian model to track movements triggering and duration. *Aquatic Living Resources*, *30*, Article 5. https://doi.org/10.1051/alr/2017004

Drouineau, H., Durif, C., Castonguay, M., Mateo, M., Rochard, E., Verreault, G., & Lambert, P. (2018). Freshwater eels: A symbol of the effects of global change. *Fish and Fisheries, 19*(5), 903–930. https://doi.org/10.1111/faf.12301

Drouineau, H., Lambert, P., Briand, C., & Fatin, D. (2018). The impact of river infrastructure on eel migration. *Fisheries Management and Ecology, 25*(3), 183–194. https://doi.org/10.1111/fme.12283

Drouineau, H., Rigaud, C., Laharanne, A., Fabre, R., Alric, A., & Baran, P. (2015). Assessing the efficiency of an elver ladder using a multi-state mark-recapture model. *River Research and Applications, 31*(3), 291–300. https://doi.org/10.1002/rra.2750

Dudgeon, D. (2019). Multiple threats imperil freshwater biodiversity in the Anthropocene. *Current Biology, 29*(19), R960–R967. https://doi.org/10.1016/j.cub.2019.08.002

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Lévêque, C., & Sullivan, C. A. (2006). Freshwater biodiversity: Importance, threats, status and conservation challenges. *Biological Reviews, 81*(2), 163–182. https://doi.org/10.1017/S1464793105006950

Durif, C. M., Stockhausen, H. H., Skiftesvik, A. B., Cresci, A., Nyqvist, D., & Browman, H. I. (2022). A unifying hypothesis for the spawning migrations of temperate anguillid eels. *Fish and Fisheries, 23*(2), 358–375. https://doi.org/10.1111/faf.12623

Durif, C., Guibert, A., & Elie, P. (2009). Morphological discrimination of the silvering stages of the European eel. In *American Fisheries Society Symposium* (Vol. 58, pp. 103-111).

Edeline, E., Dufour, S., & Elie, P. (2009). Proximate and ultimate control of eel continental dispersal. In G. van den Thillart, S. Dufour, & J. C. Rankin (Eds.), *Spawning migration of the European eel: Reproduction index, a useful tool for conservation management* (pp. 433–461). Springer. https://doi.org/10.1007/978-1-4020-9095-0_18

Edeline, E., Lambert, P., Rigaud, C., & Elie, P. (2006). Effects of body condition and water temperature on *Anguilla anguilla* glass eel migratory behaviour. *Journal of Experimental*

Marine Biology and Ecology, 331(2), 217-225. https://doi.org/10.1016/j.jembe.2005.11.002

Ege, V. (1939). A revision of the genus *Anguilla* Shaw: A systematic, phylogenetic, and geographical study. *Dana Reports,* 16.

Elliott, M., & Hemingway, K. (2002). *Fishes in estuaries*. Blackwell Science.

Elliott, M., & Whitfield, A. K. (2011). Challenging paradigms in estuarine ecology and management. *Estuarine, Coastal and Shelf Science, 94*(4), 306–314. https://doi.org/10.1016/j.ecss.2011.06.016

Erős, T. (2017). Scaling fish meta-communities in stream networks: Synthesis and future research avenues. *Community Ecology, 18*(1), 72–86. https://doi.org/10.1556/168.2017.18.1.9

European Union. (2007). *Council Regulation (EC) No 1100/2007 of 18 September 2007 establishing measures for the recovery of the stock of European eel*. Official Journal of the European Union, L 248, 17–23. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32007R1100

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics, 34*(1), 487-515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fan, H. P., Lin, Y., Zhong, Q. F., Liao, B. C., Zhuo, Y. C., & Xin, Z. M. (2008). Studies on culture biology characteristics of *Anguilla mossambica* Peters. *Journal of Fujian Fisheries, 4*, 7–12.

Fargallo, J. A., Navarro-López, J., Cantalapiedra, J. L., Pelegrin, J. S., & Hernández Fernández, M. (2022). Trophic niche breadth of Falconidae species predicts biomic specialisation but not range size. *Biology, 11*(4), 522. https://doi.org/10.3390/biology11040522

Feinsinger, P., Spears, E. E., & Poole, R. W. (1981). A simple measure of niche breadth. *Ecology, 62*(1), 27-32. https://doi.org/10.2307/1936703

Ferrante, M., Conti, G. O., Zuccarello, P., Cristaldi, A., Grasso, A., Copat, C., & Fiore, M. (2010). Evaluation of the heavy metals and organochlorine pesticides in European eel (*Anguilla anguilla*) from Sicily. *Environmental Research, 110*(8), 721–726. https://doi.org/10.1016/j.envres.2010.08.001

Ferrer, N., Folch, A., Lane, M., Olago, D., Odida, J., & Custodio, E. (2019). Groundwater hydrodynamics of an Eastern Africa coastal aquifer, including La Niña 2016–17 droughts. *Science of the Total Environment, 661*, 575-597. https://doi.org/10.1016/j.scitotenv.2019.01.125

Feunteun, E. (2002). Management and restoration of European eel population (*Anguilla anguilla*): An impossible bargain. *Ecological Engineering, 18*(5), 575–591. https://doi.org/10.1016/S0925-8574(02)00018-5

Fiedler, P. C., & Lavín, M. F. (2017). Oceanographic conditions of the eastern tropical Pacific. In P. W. Glynn, D. P. Manzello, & I. C. Enochs (Eds.), *Coral reefs of the eastern tropical Pacific: Persistence and loss in a dynamic environment* (pp. 59–83). Springer. https://doi.org/10.1007/978-94-017-7499-4_3

Fish-Info Service. (2000). *Eel market report: Consumption of eel in Japan from 1991 to 1999*. https://www.fis.com

Food and Agriculture Organization of the United Nations (FAO). (2015). Global exports and imports of *Anguilla* spp. 1976–2011. *FISHSTAT Fisheries Commodities Production and Trade Database: Data extracted December 2019*. FAO.

Foulds, W. L., & Lucas, M. C. (2013). Migration barriers for eels in modified rivers. *Hydrobiologia*, *703*(1), 117–131. https://doi.org/10.1007/s10750-012-1353-9

Friedland, K. D., Miller, M. J., & Knights, B. (2007). Oceanic changes in the Sargasso Sea and declines in recruitment of the European eel. *ICES Journal of Marine Science*, *64*(3), 519–530. https://doi.org/10.1093/icesjms/fsm022

Friedland, K. D., Miller, M. J., & Knights, B. (2021). Oceanic changes in the Sargasso Sea and declining eel recruitment. *ICES Journal of Marine Science, 78*(2), 456–468. https://doi.org/10.1093/icesjms/fsaa189

Froehlicher, H., Kaifu, K., Rambonilaza, T., & Daverat, F. (2023). Eel translocation from a conservation perspective: A coupled systematic and narrative review. *Global Ecology and Conservation, 46*, e02635. https://doi.org/10.1016/j.gecco.2023.e02635

Froese, R. (2006). Cube law, condition factor and weight–length relationships: History, meta-analysis and recommendations. *Journal of Applied Ichthyology, 22*(4), 241-253. https://doi.org/10.1111/j.1439-0426.2006.00805.x

Froese, R., & Pauly, D. (Eds.). (2016). *Fish-Base* [World Wide Web electronic publication]. Retrieved May 2023, from https://www.fishbase.org

Froese, R., Zeller, D., Kleisner, K., & Pauly, D. (2012). What catch data can tell us about the status of global fisheries. *Marine Biology, 159*, 1283-1292. https://doi.org/10.1007/s00227-012-1916-3

Frost, H. (2001). *Socioeconomic cost-benefit analysis of the use of glass eel* [Unpublished study].

Fulton, T. W. (1904). On the spawning of the cod (*Gadus morrhua* L.) in autumn in the North Sea. *Sea, 6*(6.51), 6-53.

Gagnaire, P. A., Normandeau, E., Côté, C., Møller Hansen, M., & Bernatchez, L. (2012). The genetic consequences of spatially varying selection in the panmictic American eel (*Anguilla rostrata*). *Genetics, 190*(2), 725–736. https://doi.org/10.1534/genetics.111.134825

Galib, S. M., Rashid, M. A., Chaki, N., Mohsin, A. B. M., & Joadder, M. A. R. (2018). Seasonal variation and community structure of fishes in the Mahananda River with special reference to conservation issues. *Journal of Fisheries, 4*(1), 325–334. https://doi.org/10.17017/jfish.v4i1.2018.264

Gallagher, C. A., Chimienti, M., Grimm, V., & Nabe-Nielsen, J. (2022). Energy-mediated responses to changing prey size and distribution in marine top predator movements and population dynamics. *Journal of Animal Ecology, 91*(1), 241–254. https://doi.org/10.1111/1365-2656.13627

Gebrekiros, S. T. (2016). Factors affecting stream fish community composition and habitat suitability. *Journal of Aquaculture and Marine Biology, 4*(2), 00076. https://doi.org/10.15406/jamb.2016.04.00076

Geeraerts, C., & Belpaire, C. (2010). The effects of contaminants on European eel health. *Science of the Total Environment, 408*(21), 4169–4177. https://doi.org/10.1016/j.scitotenv.2010.05.011

Geist, J., & Hawkins, S. J. (2020). Riverbed restoration as a conservation tool for freshwater species. *Conservation Biology, 34*(6), 1503–1515. https://doi.org/10.1111/cobi.13524

Gkenas, C., Oikonomou, A., Economou, A. N., & Leonardos, I. D. (2012). Fish invasions in the Mediterranean: How do non-native species respond to habitat characteristics? *Aquatic Biology, 17*(2), 123–134. https://doi.org/10.3354/ab00473

Glova, G. J., Jellyman, D. J., & Bonnett, M. L. (1998). Factors associated with the distribution and habitat of eels (*Anguilla* spp.) in three New Zealand lowland streams. *New Zealand Journal of Marine and Freshwater Research, 32*(2), 255-269. https://doi.org/10.1080/00288330.1998.9516837

Godínez-Ortega, J. L., Cuatlán-Cortés, J. V., López-Bautista, J. M., & van Tussenbroek, B. I. (2021). A natural history of floating *Sargassum* species (*Sargasso*) from Mexico. In *Natural history and ecology of Mexico and Central America* (pp. 59–94).

Gollock, M., Curnick, D., & Debney, A. (2011). Recent recruitment trends of juvenile eels in tributaries of the River Thames. *Hydrobiologia, 672*, 33–37. https://doi.org/10.1007/s10750-011-0734-y

Government of Kenya (GoK). (2017). Reliable update of Kenya's emission baseline projections and impact on NDC target.

Gracey, E. O., & Verones, F. (2016). Impacts from hydropower production on biodiversity in an LCA framework—review and recommendations. *The International Journal of Life Cycle Assessment, 21*(3), 412–428. https://doi.org/10.1007/s11367-016-1039-3

Gregory, S. V., Swanson, F. J., McKee, W. A., & Cummins, K. W. (1991). An ecosystem perspective of riparian zones. *BioScience, 41*(8), 540–551. https://doi.org/10.2307/1312252

Grill, G., Lehner, B., Thieme, M., Geenen, B., & Zarfl, C. (2019). Mapping the world's free-flowing rivers. *Nature, 569*(7755), 215–221. https://doi.org/10.1038/s41586-019-1111-9

Guangwei, D., & Shishun, L. (1999). *Analysis of impetuses to change of agricultural land resources in China* (Bulletin No. 13[1]). Chinese Academy of Sciences.

Guareschi, S., & Wood, P. J. (2019). Taxonomic changes and non-native species: An overview of constraints and new challenges for macroinvertebrate-based indices calculation in river ecosystems. *Science of the Total Environment, 660,* 40–46. https://doi.org/10.1016/j.scitotenv.2019.01.008

Guimaraes, L., Gravato, C., Santos, J., Monteiro, L. S., & Guilhermino, L. (2009). Yellow eel (*Anguilla anguilla*) development in NW Portuguese estuaries with different contamination levels. *Ecotoxicology, 18,* 385-402. https://doi.org/10.1007/s10646-009-0391-5

Guo, H., Zhang, X., Zhang, Y., Tang, W., & Liu, K. (2024). Recruitment patterns and environmental sensitivity of glass eels of *Anguilla japonica* in the Yangtze Estuary, China. *Biology, 13*(1), 56. https://doi.org/10.3390/biology13010056

Guo, J. Z., Chen, Z. Z., & Xu, S. S. (2017). Advances in fish particle size spectra study. *Marine Fisheries, 39*(5), 582-591.

Guo, J., Venier, M., Salamova, A., & Hites, R. A. (2017). Bioaccumulation of Dechloranes, organophosphate esters, and other flame retardants in Great Lakes fish. *Science of the Total Environment, 583,* 1–9. https://doi.org/10.1016/j.scitotenv.2017.01.032

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., & Fujita, R. (2008). A global map of human impact on marine ecosystems. *Science, 319*(5865), 948–952. https://doi.org/10.1126/science.1149345

Haro, A. (2014). Anguillidae: Freshwater eels. In M. L. Warren, Jr. & B. M. Burr (Eds.), *Freshwater fishes of North America* (Vol. 1, pp. 313–331). Johns Hopkins University Press.

Haro, A., Richkus, W., Whalen, K., Hoar, A., Busch, W. D., Lary, S., Brush, T., & Dixon, D. (2000). Population decline of the American eel: Implications for research and management. *Fisheries, 25*(9), 7–16. https://doi.org/10.1577/1548-8446(2000)025<0007:PDOTAE>2.0.CO;2

Harrison, A. J., Walker, A. M., Pinder, A. C., Briand, C., & Aprahamian, M. W. (2014). A review of glass eel migratory behaviour, sampling techniques and abundance estimates in estuaries: Implications for assessing recruitment, local production and exploitation. *Reviews in Fish Biology and Fisheries*, *24*, 967–983. https://doi.org/10.1007/s11160-014-9365-9

Harrison, I. J., Stiassny, M. L. J., & Winemiller, K. O. (2021). Anthropogenic impacts on freshwater ecosystems: A global synthesis. *BioScience, 71*(9), 823–834. https://doi.org/10.1093/biosci/biab056

Hartanto, F., Bataragoa, N. E., & Lohoo, A. V. (2016). Longitudinal distribution and morphometric character of eel at the downstream site of Kabur River, East Likupang, North Minahasa. *Scientific Journal Platax*, *3*(2), 54-62.

Hartmann, D. L., Tank, A. M. G. K., & Rusticucci, M. (2013). Observations: Atmosphere and surface. In T. F. Stocker, D. Qin, G.-K. Plattner, M. Tignor, S. K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex, & P. M. Midgley (Eds.), *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 31–39). Cambridge University Press. https://doi.org/10.1017/CBO9781107415324.008

Hatakeyama, R., Sudo, R., Yatabe, T., Yamano, K., & Nomura, K. (2022). Developmental features of Japanese eels, *Anguilla japonica*, from the late leptocephalus to the yellow eel stages: An early metamorphosis to the eel-like form and a prolonged transition to the juvenile. *Journal of Fish Biology, 100*(2), 454–473. https://doi.org/10.1111/jfb.14971

Hayden, B., Palomares, M. L. D., Smith, B. E., & Poelen, J. H. (2019). Biological and environmental drivers of trophic ecology in marine fishes—a global perspective. *Scientific Reports, 9,* Article 11415. https://doi.org/10.1038/s41598-019-47618-2

Heisey, P. G., Mathur, D., Phipps, J. L., Avalos, J. C., Hoffman, C. E., Adams, S. W., & De-Oliveira, E. (2019). Passage survival of European and American eels at Francis and propeller turbines. *Journal of Fish Biology, 95*(5), 1172–1183. https://doi.org/10.1111/jfb.14115

Helfman, G. S., Bozeman, E. L., & Brothers, E. B. (1987). Size, age, and sex of American eels in a Georgia river. *Transactions of the American Fisheries Society, 113*(2), 132-141. https://doi.org/10.1577/1548-8659(1984)113<132: SAASOA>2.0.CO;2

Hope, C. (2006). The marginal impact of CO2 from PAGE2002: An integrated assessment model incorporating the IPCC's five reasons for concern. *Integrated Assessment Journal, 6*(1), 1-22.

Houde, E. D., Able, K. W., Strydom, N. A., Wolanski, E., & Arula, T. (2022). Reproduction, ontogeny and recruitment. In A. K. Whitfield et al. (Eds.), *Fish and fisheries in estuaries: A global perspective* (pp. 60–187). John Wiley & Sons. https://doi.org/10.1002/9781119705345.ch3

Hoyle, S. D., Kim, D., Lee, S., Matsumoto, T., Satoh, K., & Yeh, Y. (2016). Collaborative study of tropical tuna CPUE from multiple Indian Ocean longline fleets in 2016. *IOTC WPTT18, Victoria.*

Hsiung, K. M., Ma, C., Ko, C. Y., Tseng, Y. H., Kuo, Y. C., & Han, Y. S. (2022). Effects of environmental factors within the spawning area and migration routes on the length of *Anguilla japonica* glass eels recruited to Taiwan. *Marine Ecology Progress Series, 683*, 109–121. https://doi.org/10.3354/meps13967

Hsu, H. Y., Chen, H. W., & Han, Y. S. (2019). Habitat partitioning and its possible genetic background between two sympatrically distributed eel species in Taiwan. *Zoological Studies, 58*(27). https://doi.org/10.6620/ZS.2019.58-27

Hsu, H. Y., Lin, Y. T., Huang, Y. C., & Han, Y. S. (2020). Skin coloration and habitat preference of the freshwater *Anguilla* eels. *International Journal of Aquaculture and Fishery Sciences, 6*(3), 96–101. https://doi.org/10.17352/2455-8400.000060

Huntingford, F., Hunter, W., & Braithwaite, V. (2012). Movement and orientation. In F. Huntingford, M. Jobling, & S. Kadri (Eds.), *Aquaculture and behaviour* (pp. 87–120). Wiley-Blackwell.

Ibbotson, A. T., Smith, J., Scarlett, P., & Aprahamian, M. W. (2002). Colonisation of freshwater habitats by the European eel *Anguilla anguilla*. *Freshwater Biology, 47*(9), 1696–1706. https://doi.org/10.1046/j.1365-2427.2002.00901.x

Itakura, H., & Wakiya, R. (2020). Habitat preference, movements and growth of giant mottled eels, *Anguilla marmorata,* in a small subtropical Amami-Oshima Island river. *Peer Journal, 8,* e10187. https://doi.org/10.7717/peerj.10187

Itakura, H., Kitagawa, T., Miller, M. J., & Kimura, S. (2015). Declines in catches of Japanese eels in rivers and lakes across Japan: Have river and lake modifications reduced fishery catches? *Landscape and Ecological Engineering, 11*(1), 147–160. https://doi.org/10.1007/s11355-014-0257-y

Jacoby, D. M. P., Casselman, J. M., Crook, V., DeLucia, M. B., Gollock, M., Kurwie, T., Maxey, M., Namba, M., Reynolds, J., & Robinet, T. (2015). Anguillid eels: Their status and management. *Fisheries and Aquaculture Journal, 46*(4), 411–421. https://doi.org/10.1111/faf.12075

Jacoby, D. M. P., Gollock, M. J., Casselman, J. M., Crook, V., DeLucia, M. B., & Robinet, T. (2015). The importance of conserving Anguillid eels. *Conservation Biology, 29*(2), 452–461. https://doi.org/10.1111/cobi.12414

Jacoby, D. M., Casselman, J. M., Crook, V., DeLucia, M. B., Ahn, H., Kaifu, K., & Gollock, M. J. (2015). Synergistic patterns of threat and the challenges facing global anguillid eel conservation. *Global Ecology and Conservation, 4*, 321–333. https://doi.org/10.1016/j.gecco.2015.07.009

Jacoby, D., & Gollock, M. (2014). *Anguilla anguilla*. The IUCN Red List of Threatened Species 2014. https://doi.org/10.2305/IUCN.UK.2014-1.RLTS.T60344A45833138.en

Jansen, H. M., Winter, H. V., Bruijs, M. C., & Polman, H. J. (2007). Just go with the flow? Route selection and mortality during downstream migration of silver eels in relation to river discharge. *ICES Journal of Marine Science, 64*(7), 1437–1443. https://doi.org/10.1093/icesjms/fsm152

Jellyman, D. J. (1989). Diet of two species of freshwater eel (*Anguilla* spp.) in Lake Pounui, New Zealand. *New Zealand Journal of Marine and Freshwater Research, 23*(1), 1-10. https://doi.org/10.1080/00288330.1989.9516455

Jellyman, D. J. (2007). Status of New Zealand freshwater eel stocks and management initiatives. *ICES Journal of Marine Science, 64*(7), 1379–1386. https://doi.org/10.1093/icesjms/fsm105

Jellyman, D. J. (2022). An enigma: How can freshwater eels (*Anguilla* spp.) be such a successful genus yet be universally threatened? *Reviews in Fish Biology and Fisheries, 32*(2), 701–718. https://doi.org/10.1007/s11160-022-09702-0

Jespersen, P. (1942). *Indo-Pacific leptocephalus of the genus Anguilla: Systematic and biological studies.* Carlsberg Foundation.

Jubb, R. A. (1964). The eels of South African rivers and observations on their ecology. In D. H. S. Davis (Ed.), *Ecological studies in Southern Africa* (pp. 186–206). Springer.

Junk, W. J., & Piedade, M. T. F. (2011). An introduction to South American wetland forests: Distribution, definitions and general characterization. In W. J. Junk, M. T. F. Piedade, F. Wittmann, J. Schöngart, & P. Parolin (Eds.), *Amazonian floodplain forests: Ecophysiology, biodiversity, and sustainable management* (pp. 3–25). Springer. https://doi.org/10.1007/978-90-481-8725-6_1

Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river-floodplain systems. *Canadian Special Publication of Fisheries and Aquatic Sciences, 106*, 110–127.

Kaifu, K., Aoyama, J., & Tsukamoto, K. (2013). Movements of the Japanese eel in natural habitats during the early growth phase. *Environmental Biology of Fishes, 96*(6), 735–747. https://doi.org/10.1007/s10641-012-0074-5

Kaifu, K., Stein, F., Dekker, W., Walker, N., Dolloff, C. A., Steele, K., Aguirre, A. A., Nijman, V., Siriwat, P., & Sasal, P. (2019). Global exploitation of freshwater eels (*Anguilla* spp.): Fisheries, stock status, and illegal trade. In P. Coulson & A. Don (Eds.), *Eels: Biology, monitoring, management, culture and exploitation* (pp. 377–422). CABI. https://doi.org/10.1079/9781800629097.0023

Kalogianni, E., Vourka, A., Karaouzas, I., Vardakas, L., Laschou, S., & Skoulikidis, N. T. (2017). Combined effects of water stress and pollution on macroinvertebrate and fish assemblages in a Mediterranean intermittent river. *Science of the Total Environment, 603*, 639–650. https://doi.org/10.1016/j.scitotenv.2017.06.078

Kanazawa, K., & Miyake, Y. (2006). Comparison of stream invertebrate assemblages between concrete and natural substrates. *Ecology and Civil Engineering, 9*(2), 141–150. https://doi.org/10.3825/ece.9.141

Kar, D. (2015). *Epizootic ulcerative fish disease syndrome*. Academic Press.

Karr, J. R., & Dudley, D. R. (1981). Ecological perspective on water quality goals. *Environmental Management, 5*, 55–68. https://doi.org/10.1007/BF01866609

Katselis, G. I., Papadopoulou, K. I., & Vassilopoulou, V. (2003). Feeding ecology of the European eel (*Anguilla anguilla* L.) in the Greek Ionian Sea. *Fisheries Research*, *64*(1), 1–10. https://doi.org/10.1016/S0165-7836(03)00035-8

Kellner, K. F., Duchamp, J. E., & Swihart, R. K. (2019). Niche breadth and vertebrate sensitivity to habitat modification: Signals from multiple taxa across replicated landscapes. *Biodiversity and Conservation, 28,* 2647-2667. https://doi.org/10.1007/s10531-019-01772-w

Kemp, P. S., Worthington, T. A., & Langford, T. E. (2022). Riverbed management for the conservation of migratory fish. *Aquatic Conservation: Marine and Freshwater Ecosystems, 32*(1), 89–102. https://doi.org/10.1002/aqc.3707

Kemp, P. S., Worthington, T. A., Langford, T. E. L., Tree, A. R. J., & Gaywood, M. J. (2015). The effects of river modification on eel populations. *Ecological Applications, 25*(3), 781–797. https://doi.org/10.1890/14-0276.1

Kenya National Bureau of Statistics. (2010). *The 2009 Kenya population and housing census* (Vol. 1). Kenya National Bureau of Statistics.

Kihia, C., O'Brien, G., Celine, H., & Mbaru, E. (2022, April). *WIO slippery eel: Slippery eel fishery along Kenyan east flowing rivers expose lacunae in Kenyan fisheries management; Preliminary results: Eel fishery and its management* [Conference presentation]. Egerton University International Conference, Egerton, Kenya. Unpublished manuscript.

Kimura, S., & Tsukamoto, K. (2006). The salinity front in the North Equatorial Current: A landmark for the spawning migration of the Japanese eel (*Anguilla japonica*) related to the stock recruitment. *Deep Sea Research Part II: Topical Studies in Oceanography, 53*(3–4), 315–325. https://doi.org/10.1016/j.dsr2.2006.01.009

Kingsford, R. T. (2000). Ecology of Australian freshwater wetlands: A review. *Australian Journal of Ecology, 25*(2), 173–189. https://doi.org/10.1046/j.1442-9993.2000.01000.x

Kirk, R. S. (2003). The impact of the nematode parasite *Anguillicola crassus* on European eels. *Fisheries Management and Ecology, 10*(6), 385–394. https://doi.org/10.1046/j.1365-2400.2003.00355.x

Kirk, R. S., & Lewis, J. W. (1993). An evaluation of pollutant-induced changes in the gills of rainbow trout using scanning electron microscopy. *Environmental Technology, 14*(6), 577–585. https://doi.org/10.1080/09593339309385338

Kirk, R. S., Lewis, J. W., & Kennedy, C. R. (2003). Survival and transmission of *Anguillicola crassus* Kuwahara, Niimi & Itagaki, 1974 (Nematoda) in seawater eels. *Parasitology, 120*(3), 289–295. https://doi.org/10.1017/S0031182002008728

Kiruba-Sankar, R., Raj, J. P., Saravanan, K., Kumar, K. L., Angel, J. R. J., Velmurugan, A., & Roy, S. D. (2018). Invasive species in freshwater ecosystems: Threats to ecosystem services. In C. Sivaperuman (Ed.), *Biodiversity and climate change adaptation in tropical islands* (pp. 257–296). Academic Press. https://doi.org/10.1016/B978-0-12-813064-3.00013-6

Kleynhans, C. J. (1996). A qualitative procedure for the assessment of the habitat integrity status of the Luvuvhu River (Limpopo system, South Africa). *Journal of Aquatic Ecosystem Health, 5,* 41-54. https://doi.org/10.1023/A:1009932011912

Knights, B. (2003). A review of the possible impacts of long-term oceanic and climate changes and fishing mortality on recruitment of anguillid eels of the Northern Hemisphere. *Science of the Total Environment, 310*(1–3), 237–244. https://doi.org/10.1016/S0048-9697(02)00644-7

Koei, N. (2013). *The Project on the Development of the National Water Master Plan 2030; Final Report; Volume V Sectoral Report (E)—Agriculture and Irrigation; The Republic of Kenya.* Water Resources Management Authority: Nairobi, Kenya.

Korkea-Aho, T. L., Partanen, J. M., Kiviniemi, V., Vainikka, A., & Taskinen, J. (2006). Association between environmental stress and epidermal papillomatosis of roach *Rutilus rutilus*. *Diseases of Aquatic Organisms, 72*(1), 1–8. https://doi.org/10.3354/dao072001

Koster, W. M., Dawson, D. R., Kitchingman, A., Moloney, P. D., & Hale, R. (2020). Habitat use, movement and activity of two large-bodied native riverine fishes in a regulated lowland weir pool. *Journal of Fish Biology, 96*(3), 782–794. https://doi.org/10.1111/jfb.14272

Koushlesh, S. K., Sinha, A., Kumari, K., Borah, S., Chanu, T. N., Baitha, R., & Das, B. K. (2018). Length- weight relationship and relative condition factor of five indigenous fish species from Torsa River, West Bengal, India. *Journal of Applied Ichthyology, 34*(1), 169-171. https://doi.org/10.1111/jai.13717

Kuehne, L. M., Dickens, C., Tickner, D., Messager, M. L., Olden, J. D., O'Brien, G., & Eriyagama, N. (2023). The future of global river health monitoring. *PLOS Water, 2*(9), e0000101. https://doi.org/10.1371/journal.pwat.0000101

Kültz, D. (2015). Physiological mechanisms used by fish to cope with salinity stress. *The Journal of Experimental Biology, 218*(12), 1907-1914. https://doi.org/10.1242/jeb.118210

Kumai, Y., Kuroki, M., & Morita, K. (2021). Influence of environmental parameters on habitat use by sympatric freshwater eels *Anguilla marmorata* and *Anguilla japonica* on Yakushima Island, Japan. *Canadian Journal of Zoology, 99*(12), 1020-1027. https://doi.org/10.1139/cjz-2021-0047

Kumai, Y., Kuroki, M., & Morita, K. (2021). Influence of environmental parameters on habitat use by sympatric freshwater eels *Anguilla marmorata* and *Anguilla japonica* on Yakushima Island, Japan. *Canadian Journal of Zoology, 99*(12), 1020–1027. https://doi.org/10.1139/cjz-2021-0034

Kumai, Y., Tsukamoto, K., & Kuroki, M. (2020). Growth and habitat use of two anguillid eels, *Anguilla marmorata* and *A. japonica*, on Yakushima Island, Japan. *Ichthyological Research, 67,* 375–384. https://doi.org/10.1007/s10228-020-00732-y

Kume, M., Terashima, Y., Wada, T., & Yamashita, Y. (2019). Longitudinal distribution and microhabitat use of young Japanese eel *Anguilla japonica* in a small river flowing through paddy areas. *Journal of Applied Ichthyology, 35*(4), 876–883. https://doi.org/10.1111/jai.13917

Kuroki, M. (2023). Life history. In K. Tsukamoto & M. J. Miller (Eds.), *Eel science* (pp. 37–50). Springer Nature Singapore. https://doi.org/10.1007/978-981-99-5692-0_3

Kuroki, M., Miller, M. J., & Tsukamoto, K. (2014). Diversity of early life-history traits in freshwater eels and the evolution of their oceanic migrations. *Canadian Journal of Zoology, 92*(9), 749–770. https://doi.org/10.1139/cjz-2014-0038

Laffaille, P., Feunteun, E., Baisez, A., Robinet, T., Acou, A., Legault, A., & Lek, S. (2003). Spatial organization of European eel (*Anguilla anguilla* L.) in a small catchment. *Ecology of Freshwater Fish, 12*(4), 254–264. https://doi.org/10.1046/j.1600-0633.2003.00029.x

Lagarde, R., Peyre, J., Amilhat, E., Bourrin, F., Prellwitz, F., Perrault, A., Simon, G., & Faliex, E. (2022). Phenology and environmental drivers of glass eel entrance in a Mediterranean lagoon. *ICES Journal of Marine Science, 79*(7), 2107–2116. https://doi.org/10.1093/icesjms/fsac144

Lambert, P., Beaulaton, L., Daverat, F., & Rigaud, R. (2006). Assessment of eel stock status in Garonne and Dordogne water bodies by analysing length structures. In *ICES Annual Science Conference Maastricht, NLD, 19-23 September 2006* (p. 19).

Lambert, T., & D'Adamo, N. (2011). The effects of habitat modification on the ecology and population dynamics of freshwater eels (*Anguilla* spp.): A review. *Ecological Applications, 21*(1), 66-76. https://doi.org/10.1890/09-0606.1

Landis, W. G., & Thomas, J. F. (2009). Regional risk assessment as a part of the long-term receiving water study. *Integrated Environmental Assessment and Management, 5*(2), 234–247. https://doi.org/10.1897/IEAM_2008-062.1

Landis, W. G., & Wiegers, J. A. (1997). Design considerations and a suggested approach for regional and comparative ecological risk assessment. *Human and Ecological Risk Assessment: An International Journal, 3*(3), 287–297. https://doi.org/10.1080/10807039709383685

Landis, W. G., & Wiegers, J. K. (2005). Introduction to the regional risk assessment using the relative risk model. In W. G. Landis (Ed.), *Regional scale ecological risk assessment using the relative risk model* (pp. 11–36). CRC Press. https://doi.org/10.1201/9781420037410

Landis, W. G., & Wiegers, J. K. (2007). Ten years of the relative risk model and regional scale ecological risk assessment. *Human and Ecological Risk Assessment: An International Journal, 13*(1), 25–38. https://doi.org/10.1080/10807030601107536

Landis, W., Sofield, R., Yu, M. H., & Landis, W. G. (2003). *Introduction to environmental toxicology: Impacts of chemicals upon ecological systems*. CRC Press.

Landsberg, J., Van Dolah, F., & Doucette, G. (2005). Marine and estuarine harmful algal blooms: Impacts on human and animal health. In S. L. Belkin & R. R. Colwell (Eds.), *Oceans and health: Pathogens in the marine environment* (pp. 165–215). Springer. https://doi.org/10.1007/0-387-23709-7_7

Langerhans, R. B. (2008). Predictability of phenotypic differentiation across flow regimes in fishes. *Integrative and Comparative Biology, 48*(6), 750-768. https://doi.org/10.1093/icb/icn068

Lanszki, J., Hayward, M. W., Ranc, N., & Zalewski, A. (2022). Dietary flexibility promotes range expansion: The case of golden jackals in Eurasia. *Journal of Biogeography, 49*(6), 993-1005. https://doi.org/10.1111/jbi.14256

Le Cren, E. D. (1951). The length-weight relationship and seasonal cycle in gonad weight and condition in the perch (*Perca fluviatilis*). *The Journal of Animal Ecology, 20*, 201-219. https://doi.org/10.2307/1540

Le, D. Q., Nguyen, D. C., Harino, H., Kakutani, N., Chino, N., & Arai, T. (2010). Distribution of trace metals and methylmercury in soft tissues of the freshwater eel *Anguilla marmorata* in Vietnam. *Archives of Environmental Contamination and Toxicology, 59*(2), 282–290. https://doi.org/10.1007/s00244-010-9481-1

Lee, W. C., Chen, Y. H., Lee, Y. C., & Liao, I. C. (2003). The competitiveness of the eel aquaculture in Taiwan, Japan, and China. *Aquaculture, 221*(1–4), 115–124. https://doi.org/10.1016/S0044-8486(03)00053-4

Lefebvre, F., Fazio, G., Mounaix, B., & Crivelli, A. J. (2013). Is the continental life of the European eel *Anguilla anguilla* affected by the parasitic invader *Anguillicoloides crassus*? *Proceedings of the Royal Society B: Biological Sciences, 280*(1754), 20122916. https://doi.org/10.1098/rspb.2012.2916

Leone, C., Zucchetta, M., Capoccioni, F., Gravina, M. F., Franzoi, P., & Ciccotti, E. (2016). Stage-specific distribution models can predict eel (*Anguilla anguilla*) occurrence during settlement in coastal lagoons. *Estuarine, Coastal and Shelf Science, 170*, 123–133. https://doi.org/10.1016/j.ecss.2016.01.010

Ligeiro, R., Hughes, R. M., Kaufmann, P. R., Macedo, D. R., Firmiano, K. R., Ferreira, W. R., & Callisto, M. (2013). Defining quantitative stream disturbance gradients and the additive role of habitat variation to explain macroinvertebrate taxa richness. *Ecological Indicators*, *25*, 45–57. https://doi.org/10.1016/j.ecolind.2012.09.004

Lima, J. D. F., Garcia, J. D. S., & Silva, T. C. D. (2014). Natural diet and feeding habits of a freshwater prawn (*Macrobrachium carcinus*: Crustacea, Decapoda) in the estuary of the Amazon River. *Acta Amazonica, 44*, 235-244. https://doi.org/10.1590/S0044-59672014000200004

Limburg, K. E., & Waldman, J. R. (2018). Eels and people: A global perspective. *Fisheries, 43*(8), 397–407. https://doi.org/10.1002/fsh.10172

Lin, K., Lin, Y., Liu, P., He, Y., & Tu, X. (2016). Considering the order and symmetry to improve the traditional RVA for the evaluation of hydrologic alteration of river systems. *Water Resources Management, 30*(14), 5501–5516. https://doi.org/10.1007/s11269-016-1502-8

Lin, Y. J., Jessop, B. M., Weyl, O. L. F., Iizuka, Y., Lin, S. H., Tzeng, W. N., & Sun, C. L. (2012). Regional variation in otolith Sr: Ca ratios of African long-finned eel *Anguilla mossambica* and mottled eel *Anguilla marmorata*: A challenge to the classic tool for reconstructing migratory histories of fishes. *Journal of Fish Biology, 81*(2), 427–441. https://doi.org/10.1111/j.1095-8649.2012.03357.x

Lin, Y. J., Jessop, B. M., Weyl, O. L. F., Iizuka, Y., Lin, S. H., & Tzeng, W. N. (2015). Migratory history of African long-finned eel *Anguilla mossambica* from Maningory River, Madagascar: Discovery of a unique pattern in otolith Sr: Ca ratios. *Environmental Biology of Fishes, 98*(2), 457–468. https://doi.org/10.1007/s10641-014-0285-0

Lin, Y. J., Jessop, B. M., Weyl, O. L. F., Iizuka, Y., Lin, S. H., Tzeng, W. N., & Sun, C. L. (2013). Regional variation in otolith Sr: Ca ratios of African long-finned eel *Anguilla mossambica* and mottled eel *Anguilla marmorata*: A challenge to the classic tool for reconstructing migratory histories of fishes. *Journal of Fish Biology, 81*(2), 427–441. https://doi.org/10.1111/jfb.12178

Liu, H., Labonne, J., Coste, P., Huchet, E., Plagnes-Juan, E., Rives, J., & Bolliet, V. (2019). Looking at the complex relationships between migration behaviour and conditional strategy based on energy metabolism in the European glass eel (*Anguilla anguilla*). *Science of the Total Environment, 696*, 134039. https://doi.org/10.1016/j.scitotenv.2019.134039

Lombardini, E. D., Hard, G. C., & Harshbarger, J. C. (2014). Neoplasms of the urinary tract in fish. *Veterinary Pathology, 51*(5), 1000–1012. https://doi.org/10.1177/0300985813506054

Lotze, H. K., Lenihan, H. S., Bourque, B. J., Bradbury, R. H., Cooke, R. G., Kay, M. C., & Jackson, J. B. (2006). Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science*, *312*(5781), 1806–1809. https://doi.org/10.1126/science.1128035

Lucy, F. E., Roy, H., Simpson, A., Carlton, J. T., Hanson, J. M., Magellan, K., & McDonald, J. (2016). INVASIVESNET towards an international association for open knowledge on invasive alien species. *Management of Biological Invasions, 7*(4), 289–294. https://doi.org/10.3391/mbi.2016.7.4.01

MacGregor, R., Mathers, A., Thompson, P., Casselman, J. M., Dettmers, J. M., LaPan, S., & Allen, B. (2008). Declines of American eel in North America: Complexities associated with bi-national management. In R. J. Beamish & B. J. Rothschild (Eds.), *International governance of fisheries ecosystems: Learning from the past, finding solutions for the future* (pp. 357–381). American Fisheries Society.

Maes, G. E., Pujolar, J. M., Hellemans, B., & Volckaert, F. A. (2006). Evidence for isolation by time in the European eel (*Anguilla anguilla* L.). *Molecular Ecology, 15*(8), 2095-2107. https://doi.org/10.1111/j.1365-294X.2006.02956.x

Maestre, F. T., Eldridge, D. J., Soliveres, S., Kéfi, S., Delgado-Baquerizo, M., Bowker, M. A., & Berdugo, M. (2016). Structure and functioning of dryland ecosystems in a changing world. *Annual Review of Ecology, Evolution, and Systematics, 47*(1), 215–237. https://doi.org/10.1146/annurev-ecolsys-121415-032311

Maitland, P. S., & Craig, J. F. (2010). Freshwater fishes: A declining resource. *Silent Summer: The State of Wildlife in Britain and Ireland* (pp. 383-384). Royal Society of Biology.

Marini, M., Pedrosa-Gerasmio, I. R., Santos, M. D., Shibuno, T., Daryani, A., Romana-Eguia, M. R. R., & Wibowo, A. (2021). Genetic diversity, population structure and demographic history of the tropical eel *Anguilla bicolor pacifica* in Southeast Asia using mitochondrial DNA control region sequences. *Global Ecology and Conservation, 26*, e01493. https://doi.org/10.1016/j.gecco.2021.e01493

Martínez-Gómez, C., Fernández, B., Barcala, E., García-Aparicio, V., Jumilla, E., Gea-Pacheco, Á., & León, V. M. (2023). The impact of chemical pollution on the European eel (*Anguilla anguilla*) from a Mediterranean hypersaline coastal lagoon. *Environmental Science and Pollution Research, 30*(33), 80106–80122. https://doi.org/10.1007/s11356-023-27871-9

Masese, F. O., Kitaka, N., Kipkemboi, J., Gettel, G. M., Irvine, K., & McClain, M. E. (2018). Effects of land use on water quality and ecosystem health in Kenyan rivers. *Water, 10*(12), 1627. https://doi.org/10.3390/w10121627

Mateo, M., Lambert, P., Tétard, S., & Drouineau, H. (2017). Impacts that cause the highest direct mortality of individuals do not necessarily have the greatest influence on temperate eel escapement. *Fisheries Research, 193*, 51–59. https://doi.org/10.1016/j.fishres.2017.04.004

Mbaka, J. G., & Mwaniki, M. W. (2015). A review of hydrological alterations and their ecological implications in Kenya. *Hydrobiologia, 768*(1), 5–23. https://doi.org/10.1007/s10750-015-2543-6

McIntyre, P. B., Reidy Liermann, C., Childress, E., Hamann, E. J., Hogan, J. D., Januchowski-Hartley, S. R., & Pracheil, B. M. (2016). Conservation of migratory fishes in freshwater ecosystems. In G. P. Closs, M. Krkosek, & J. D. Olden (Eds.), *Conservation of freshwater fishes* (pp. 324–360). Cambridge University Press. https://doi.org/10.1017/CBO9781139627085.014

Meister, A. L., & Flagg, L. N. (1997). Recent developments in the American eel fisheries of eastern North America. *Focus, 22*(1), 25–26.

Menzel, A., Sparks, T. H., Estrella, N., Koch, E., Aasa, A., Ahas, R., & Chmielewski, F. M. (2006). European phenological response to climate change matches the warming pattern.

Global Change Biology, 12(10), 1969–1976. https://doi.org/10.1111/j.1365-2486.2006.01193.x

Merk, B., Höhne, L., Freese, M., Marohn, L., Hanel, R., & Pohlmann, J. D. (2023). To hear or not to hear: Selective tidal stream transport can interfere with the detectability of migrating silver eels in a tidal river. *Animal Biotelemetry, 11*(1), 44. https://doi.org/10.1186/s40317-023-00353-y

Miller, M. J., & Tsukamoto, K. (2016). The ecology of oceanic dispersal and survival of anguillid leptocephali. *Canadian Journal of Fisheries and Aquatic Sciences, 73*(2), 233–244. https://doi.org/10.1139/cjfas-2015-0340

Miller, M. J., Bonhommeau, S., Munk, P., Castonguay, M., Hanel, R., & McCleave, J. D. (2015). A century of research on the larval distributions of the Atlantic eels: A re-examination of the data. *Biological Reviews, 90*(4), 1035–1064. https://doi.org/10.1111/brv.12144

Miller, M. J., Casselman, J. M., & Secor, D. H. (2020). Habitat fragmentation and the decline of Anguillid eels. *Canadian Journal of Fisheries and Aquatic Sciences, 77*(9), 1412–1425. https://doi.org/10.1139/cjfas-2019-0383

Miller, M. J., Feunteun, E., & Tsukamoto, K. (2016). Did a “perfect storm” of oceanic changes and continental anthropogenic impacts cause northern hemisphere anguillid recruitment reductions? *ICES Journal of Marine Science, 73*(1), 43–56. https://doi.org/10.1093/icesjms/fsv063

Miller, M. J., Shimizu, M., Aoyama, J., Watanabe, S., Kuroki, M., Feunteun, E., & Tsukamoto, K. (2022). Distribution and abundance of leptocephali in the western South Pacific region during two large-scale sampling surveys. *Progress in Oceanography, 206*, 102853. https://doi.org/10.1016/j.pocean.2022.102853

Milly, P. C., Dunne, K. A., & Vecchia, A. V. (2005). Global pattern of trends in streamflow and water availability in a changing climate. *Nature, 438*(7066), 347–350. https://doi.org/10.1038/nature04312

Milošević, D., Mrdak, D., & Causevic, D. (2022). Length-weight relationship and condition factor of silver stage of European eel, *Anguilla anguilla* (Linnaeus, 1758) from Lake Skadar (Montenegro). *Regional Studies in Marine Science, 35,* 5-11. https://doi.org/10.1016/j.rsma.2020.101319

Monteiro, R. M., Domingos, I., Almeida, P. R., Costa, J. L., Pereira, E., Belo, A. F., Portela, T., Telhado, A., & Quintella, B. R. (2023). Upstream movement of juvenile eels

(*Anguilla anguilla* L.) in a south-western European river. *Environmental Biology of Fishes, 106*(6), 1313–1325. https://doi.org/10.1007/s10641-023-01417-x

Moore, C. J. (2008). Synthetic polymers in the marine environment: A rapidly increasing, long-term threat. *Environmental Research, 108*(2), 131–139. https://doi.org/10.1016/j.envres.2008.07.025

Moyle, P. B., & Marchetti, M. P. (2019). *Environmental flow assessment: Methods and applications*. Wiley

Mumma, A., Lane, M., Kairu, E., Tuinhof, A., & Hirji, R. (2011). *Kenya groundwater governance case study*.

Naiman, R. J., & Décamps, H. (1997). The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics, 28,* 621–658. https://doi.org/10.1146/annurev.ecolsys.28.1.621

Naisbett-Jones, L. C., & Lohmann, K. J. (2022). Magneto reception and magnetic navigation in fishes: A half century of discovery. *Journal of Comparative Physiology A, 208*(1), 19–40. https://doi.org/10.1007/s00359-021-01477-w

New, A. L., Alderson, S. G., Smeed, D. A., & Stansfield, K. L. (2007). On the circulation of water masses across the Mascarene Plateau in the South Indian Ocean. *Deep Sea Research Part I: Oceanographic Research Papers, 54*(1), 42–74. https://doi.org/10.1016/j.dsr.2006.09.003

Nguyen, A. T., Tsukamoto, K., & Lokman, P. M. (2018). Composition and distribution of freshwater eels *Anguilla* spp. in Vietnam. *Fisheries Science, 84*, 987-994. https://doi.org/10.1007/s12562-018-1189-3

Nijman, V. (2015). CITES-listings, EU eel trade bans and the increase of export of tropical eels out of Indonesia. *Marine Policy, 58,* 36–41. https://doi.org/10.1016/j.marpol.2015.04.009

Nijman, V., & Stein, F. M. (2022). Meta-analyses of molecular seafood studies identify the global distribution of legal and illegal trade in CITES-regulated European eels. *Current Research in Food Science, 5,* 191-195. https://doi.org/10.1016/j.crfs.2022.02.002

Nishimoto, A., Iida, M., Yokouchi, K., Fukuda, N., & Yamamoto, T. (2023). Eels as natural samplers highlight spatial heterogeneity in energy flow in an estuary. *Estuarine, Coastal and Shelf Science, 281,* 108215. https://doi.org/10.1016/j.ecss.2023.108215

Njiru, M., Nyamweya, C. S., Owiti, H., & Makori, A. J. (2021). Water quality assessment of East African rivers. *African Journal of Aquatic Science, 46*(2), 159–172. https://doi.org/10.2989/16085914.2021.1881222

O’Brien, G. C., & Wepener, V. (2012). Regional-scale risk assessment methodology using the Relative Risk Model (RRM) for surface freshwater aquatic ecosystems in South Africa. *Water SA, 38*(2), 153-166. https://doi.org/10.4314/wsa.v38i2.6

O'Brien, G., Smit, N. J., & Wepener, V. (2021). Regional scale risk to the ecological sustainability and ecosystem services of an African floodplain system. *Risk Analysis, 41*, 1925-1952. https://doi.org/10.1111/risa.13740

Ogello, E. O., Mbodze, M. M., Outa, N. O., Ajode, M. Z., Ojwando, W., & Mwaruwa, M. M. (2024). Characterizing anthropogenic litter pollutants in African freshwater ecosystems: A case study of Dunga Beach in Lake Victoria, Kenya. *Lakes & Reservoirs: Science, Policy and Management for Sustainable Use, 29*(1), e12454. https://doi.org/10.1111/lre.12454

Ogutu-Ohwayo, R., Hecky, R. E., Cohen, A. S., & Kaufman, L. (2016). Human impacts on African basin lakes: Implications for ecosystems, livelihoods and management. *Global Environmental Change*, *39*, 1–11. https://doi.org/10.1016/j.gloenvcha.2016.03.001

Okeyo, D. O. (1998). Updating names, distribution and ecology of riverine fish of Kenya in the Athi-Galana-Sabaki river drainage system. *Naga, the ICLARM Quarterly, 21*, 44–53. https://bit.ly/2Z1qsi8

Onikura, N., Nakajima, J., Eguchi, K., Miyake, T., Nishida, T., Inui, R., & Kawamura, K. (2007). Relationships between presence and populations of freshwater fishes and revetment conditions in creeks around Sea of Ariake, northwestern Kyushu, Japan. *Journal of Japan Society on Water Environment, 30*(5), 277–282.

Overton, A. S., & Rulifson, R. A. (2009). Annual variability in upstream migration of glass eels in a southern USA coastal watershed. *Environmental Biology of Fishes, 84*, 29–37. https://doi.org/10.1007/s10641-008-9394-6

Ovidio, M., Seredynski, A. L., Philippart, J. C., & Nzau Matondo, B. (2013). A bit of quiet between the migrations: The resting life of the European eel during their freshwater growth phase in a small stream. *Aquatic Ecology, 47*, 291-301. https://doi.org/10.1007/s10452-013-9443-3

Panfili, J., Ximénès, M. C., & Crivelli, A. J. (1994). Sources of variation in growth of the European eel (*Anguilla anguilla*) estimated from otoliths. *Canadian Journal of Fisheries and Aquatic Sciences, 51*(3), 506-515. https://doi.org/10.1139/f94-053

Pangerang, U. K., Lawelle, S. A., Fekri, L., Idris, M., & Marthen, J. L. (2022). Diversity of eel (glass eel) based on morphometric measurements in the Konaweha River, Southeast Sulawesi. In S. Sutirman, D. S. Damopolii, & A. Rauf (Eds.), *Proceedings of the*

International Conference on Improving Tropical Animal Production for Food Security (ITAPS 2021) (pp. 495–501). Atlantis Press. https://doi.org/10.2991/abs.2022.501

Patey, G., Couillard, C. M., Pierron, F., Baudrimont, M., & Couture, P. (2017). Biotransformation, antioxidant and histopathological biomarker responses to contaminants in European and American yellow eels from the Gironde and St. Lawrence estuaries. *Chemosphere*, *188*, 292–303. https://doi.org/10.1016/j.chemosphere.2017.09.020

Pervin, M. R., & Mortuza, M. G. (2008). Notes on length-weight relationship and condition factor of freshwater fish, *Labeo boga* (Hamilton) (Cypriniformes: Cyprinidae). *University Journal of Zoology, Rajshahi University, 27*, 97-98.

Pianka, E. R. (1981). Resource acquisition and allocation among animals. In C. R. Townsend & P. Calow (Eds.), *Physiological ecology: An evolutionary approach to resource use* (pp. 300–314). Harvard University Press.

Pike, C., Crook, V., & Gollock, M. (2020). *Anguilla anguilla*. *The IUCN Red List of Threatened Species, 2020*, e.T60344A152845178. International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.UK.2020-2.RLTS.T60344A152845178.en

Piper, A. T., Righton, D., Aarestrup, K., Gollock, M., & Piper, A. (2021). Important questions to progress science and sustainable management of anguillid eels. *Fish and Fisheries*, 22(3), 652–667. https://doi.org/10.1111/faf.12541

Plass-Johnson, J. G., Taylor, M. H., Husain, A. A., Teichberg, M. C., & Ferse, S. C. (2016). Non-random variability in functional composition of coral reef fish communities along an environmental gradient. *Public Library of Science ONE, 11*(4), e0154014. https://doi.org/10.1371/journal.pone.0154014

Plieninger, T., Av Rana, H. Á., Fagerholm, N., Ellingsgaard, G. F., Magnussen, E., Raymond, C. M., & Verbrugge, L. N. (2018). Identifying and assessing the potential for conflict between landscape values and development preferences on the Faroe Islands. *Global Environmental Change, 52*, 162–180. https://doi.org/10.1016/j.gloenvcha.2018.07.006

Podgorniak, T., Angelini, A., Blanchet, S., De Oliveira, E., Pierron, F., & Daverat, F. (2016). Climbing experience in glass eels: A cognitive task or a matter of physical capacities? *Physiology & Behavior, 151*, 448–455. https://doi.org/10.1016/j.physbeh.2015.08.043

Poff, N. L., & Zimmerman, J. K. H. (2017). Ecological responses to altered flow regimes: A literature review to inform environmental flows science and management. *Ecohydrology, 10*(1), e1817. https://doi.org/10.1002/eco.1817

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E., & Stromberg, J. C. (2017). The natural flow regime paradigm and river conservation. *BioScience, 67*(8), 759–772. https://doi.org/10.1093/biosci/bix058

Popova, E., Vousden, D., Sauer, W. H., Mohammed, E. Y., Allain, V., Downey-Breedt, N., & Yool, A. (2019). Ecological connectivity between the areas beyond national jurisdiction and coastal waters: Safeguarding interests of coastal communities in developing countries. *Marine Policy, 104*, 90–102. https://doi.org/10.1016/j.marpol.2019.02.050

Porceddu, R., Podda, C., Mulas, G., Palmas, F., Picci, L., Scano, C., & Sabatini, A. (2022). Changes in dendritic spine morphology and density of granule cells in the olfactory bulb of *Anguilla anguilla* (L., 1758): A possible way to understand orientation and migratory behaviour. *Biology*, *11*(8), 1244. https://doi.org/10.3390/biology11081244

Postel, S., & Richter, B. (2003). *Rivers for life: Managing water for people and nature*. Island Press.

Pratama, I., Prayitno, S. B., & Syakuri, H. (2019). Identification and prevalence of parasites isolated from eels (*Anguilla bicolor*) along the migration pathway at Serayu River, Central Java. *Omni-Akuatika*, *15*(1), 81–92. https://doi.org/10.20884/1.oa.2019.15.1.689

Pujolar, J. M., & Maes, G. E. (2016). Evolutionary genomics of North Atlantic eels: Current status and perspectives. In T. Arai (Ed.), *Biology and ecology of anguillid eels* (pp. 36–51). CRC Press. https://doi.org/10.1201/b19925-4

Quatrain, A. M., McClain- Counts, J., Artabane, S. J., Roa- Varón, A., McIver, T. C., Rhode, M., & Ross, S. W. (2019). Assemblage structure, vertical distributions, and stable-isotope compositions of anguilliform leptocephali in the Gulf of Mexico. *Journal of Fish Biology*, *94*(4), 621–647.

Radke, R. J., & Eckmann, R. (1996). Piscivorous eels in Lake Constance: Can they influence year class strength of perch? *Annales Zoologici Fennici, 33*(1), 489–494.

Rajeshkumar, S., & Li, X. (2018). Bioaccumulation of heavy metals in fish species from the Meiliang Bay, Taihu Lake, China. *Toxicology Reports*, *5*, 288–295. https://doi.org/10.1016/j.toxrep.2018.01.007

Rattan, R. K., Datta, S. P., Chandra, S. C., & Saharan, N. S. (2002). Heavy metals and environmental quality—Indian scenario. *Fertiliser News, 47*, 21–40. Reeve, C., Rowsey, L. E., & Speers-Roesch, B. (2022). Inactivity and the passive slowing effect of cold on resting metabolism as the primary drivers of energy savings in overwintering fishes. *Journal of Experimental Biology, 225*(8), Article jeb243407. https://doi.org/10.1242/jeb.243407

Richkus, W. A., & Whalen, K. (2000). Evidence for a decline in the abundance of the American eel, *Anguilla rostrata* (LeSueur), in North America since the early 1980s. *Dana, 12*, 83–97.

Righton, D., Westerberg, H., Feunteun, E., Økland, F., Gargan, P., Amilhat, E., Metcalfe, J., Lobon-Cervia, J., Sjöberg, N., Simon, J., Acou, A., Vedor, M., Walker, A., Trancart, T., Brämick, U., Aarestrup, K., & McCarthy, T. (2021). Empirical observations of the spawning migration of European eels: The long and dangerous road to the Sargasso Sea. *Science Advances*, *7*(26), eabg9804. https://doi.org/10.1126/sciadv.abg9804

Ringuet, S., Muto, F., & Raymakers, C. (2002). Eels: Their harvest and trade in Europe and Asia. *Traffic Bulletin, 19*(2), 80–106.

Robinet, T., Feunteun, E., Keith, P., Marquet, G., Olivier, J. M., Réveillac, E., & Valade, P. (2007). Eel community structure, fluvial recruitment of *Anguilla marmorata* and indication for a weak local production of spawners from rivers of Réunion and Mauritius islands. *Environmental Biology of Fishes*, *78*, 93-105. https://doi.org/10.1007/s10641-006-9094-7

Robinet, T., Lecomte-Finiger, R., Escoubeyrou, K., & Feunteun, E. (2003). Tropical eels *Anguilla* spp. recruiting to Réunion Island in the Indian Ocean: Taxonomy, patterns of recruitment and early life histories. *Marine Ecology Progress Series, 259*, 263-272. https://doi.org/10.3354/meps259263

Robinet, T., Réveillac, E., Kuroki, M., Aoyama, J., Tsukamoto, K., Rabenevanana, M. W., & Feunteun, E. (2008). New clues for freshwater eels (*Anguilla* spp.) migration routes to eastern Madagascar and surrounding islands. *Marine Biology, 154*, 453-463. https://doi.org/10.1007/s00227-008-1007-7

Rochman, C. M., Hoh, E., Hentschel, B. T., & Kaye, S. (2013). Long-term field measurement of sorption of organic contaminants to five types of plastic pellets: Implications for plastic marine debris. *Environmental Science & Technology, 47*(3), 1646–1654. https://doi.org/10.1021/es303700s

Rosen, D. A. S., & Worthy, G. A. J. (2018). Nutrition and energetics. In F. M. D. Gulland, L. A. Dierauf, & K. L. Whitman (Eds.), *CRC Handbook of Marine Mammal Medicine* (3rd ed., pp. 695–738). CRC Press. https://doi.org/10.1201/9781315144931-29

Rosgen, D. L. (1996). *Applied river morphology*. Wildland Hydrology

Rousseau, K., Lafont, A. G., Pasquier, J., Maugars, G., Jolly, C., Sébert, M. E., & Dufour, S. (2013). Advances in eel reproductive physiology and endocrinology. In *Eel Physiology* (pp. 1-43). Science Publishers.

Rupasinghe, H., & Attygalle, M. V. E. (2006). Food and feeding of brown-stage eels of *Anguilla bicolor* in the Bolgoda Estuary. *Simanjuntak C. P. H., Rahardjo M. F. 2001 Journal Iktiologi Indonesia*, 1(2), 11–16. *(Unpublished study)*

Sarre, G. A., Pegg, G. G., & Ruscoe, W. A. (2015). The response of freshwater eels (*Anguilla* spp.) to habitat modifications: Implications for conservation management. *Aquatic Conservation: Marine and Freshwater Ecosystems, 25*(3), 363-375. https://doi.org/10.1002/aqc.2524

Schabetsberger, R., Chang, Y. L. K., & Miller, M. J. (2021). Spawning migration and larval dispersal of tropical Pacific eels (*Anguilla* spp.) in the centre of their distribution ranges. *Marine Ecology Progress Series, 670*, 167–184. https://doi.org/10.3354/meps13767

Schabetsberger, R., Miller, M. J., Olmo, G. D., Kaiser, R., Økland, F., Watanabe, S., & Tsukamoto, K. (2016). Hydrographic features of anguillid spawning areas: Potential signposts for migrating eels. *Marine Ecology Progress Series, 554*, 141–155. https://doi.org/10.3354/meps11811

Schaeffer, D. J., Herricks, E. E., & Kerster, H. W. (1988). Ecosystem health: I. Measuring ecosystem health. *Environmental Management, 12*(4), 445–455. https://doi.org/10.1007/BF01873258

Schmidt, R. E., Kahn, P., Helfrich, L. A., & Limburg, K. E. (2009). Impacts of habitat degradation on the distribution and abundance of American eels (*Anguilla rostrata*) in tributaries of the Hudson River, New York. *American Fisheries Society Symposium, 58,* 107–120.

Schmidt, R. E., Miller, D., & Limburg, K. E. (2020). Impacts of anthropogenic stressors on American eel populations. *Aquatic Conservation, 30*(6), 1147–1161. https://doi.org/10.1002/aqc.3276

Seegers, L., De Vos, L., & Okeyo, D. O. (2003). Annotated checklist of the freshwater fishes of Kenya (excluding the lacustrine haplochromines from Lake Victoria). *Journal of East*

African Natural History, 92(1), 11–47. https://doi.org/10.2982/0012-8317(2003)92[11:ACOTFF]2.0.CO;2

Shalovenkov, N. (2019). Alien species invasion: Case study of the Black Sea. In *Coasts and Estuaries* (pp. 547–568). https://doi.org/10.1016/B978-0-12-814003-1.00030-3

Shanmughan, A., Dahanukar, N., Harrison, A., Pinder, A. C., Ranjeet, K., & Raghavan, R. (2022). Demographics and exploitation of two Near Threatened freshwater eels, *Anguilla bengalensis* and *Anguilla bicolor*, in small-scale subsistence fisheries and implications for conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems, 32*(2), 269–281. https://doi.org/10.1002/aqc.3765

Shiraishi, H., & Crook, V. (2015). *Eel market dynamics: An analysis of Anguilla production, trade and consumption in East Asia* (TRAFFIC Report). TRAFFIC, Tokyo, Japan. https://doi.org/10.13140/RG.2.1.4426.2487

Shuai, F., Li, J., Yu, S., & Yang, J. (2023). Temporal pattern of the occurrence of Japanese glass eels (*Anguilla japonica*) in the Pearl River Estuary. *Fishes, 8*(5), 256. https://doi.org/10.3390/fishes8050256

Sidqi, M., Sarong, M. A., Batubara, A. S., & Muchlisin, Z. A. (2018). Feeding habit, length-weight relationships and condition factors of the tropical shortfin eel *Anguilla bicolor bicolor* in Banda Aceh waters, Indonesia. In *IOP Conference Series: Earth and Environmental Science, 216*(1), 012049. IOP Publishing. https://doi.org/10.1088/1755-1315/216/1/012049

Silva, S., Gage, M., Costa, M. J., & Correia, A. T. (2018). The impact of altered flow regimes on fish migration. *Aquatic Conservation, 28*(4), 745–758. https://doi.org/10.1002/aqc.2914

Simeoni, C., Furlan, E., Pham, H. V., Critto, A., de Juan, S., Trégarot, E., & Marcomini, A. (2023). Evaluating the combined effect of climate and anthropogenic stressors on marine coastal ecosystems: Insights from a systematic review of cumulative impact assessment approaches. *Science of the Total Environment, 861*, 160687. https://doi.org/10.1016/j.scitotenv.2023.160687

Steendam, C., Verhelst, P., Van Wassenbergh, S., & De Meyer, J. (2020). Burrowing behaviour of the European eel (*Anguilla anguilla*): Effects of life stage. *Journal of Fish Biology, 97*(5), 1332–1342. https://doi.org/10.1111/jfb.14481

Stein, F., Doering-Arjes, P., Fladung, E., Brämick, U., Bendall, B., & Schröder, B. (2016). Downstream migration of the European eel (*Anguilla anguilla*) in the Elbe River, Germany: Movement patterns and the potential impact of environmental factors. *River Research and Applications*, *32*(4), 666–676. https://doi.org/10.1002/rra.2881

Strayer, D. L. (2010). Alien species in fresh waters: Ecological effects, interactions with other stressors, and prospects for the future. *Freshwater Biology, 55*, 152–174. https://doi.org/10.1111/j.1365-2427.2009.02210.x

Stuart-Smith, R. D., Bates, A. E., Lefcheck, J. S., Duffy, J. E., Baker, S. C., Thomson, R. J., & Edgar, G. J. (2013). Integrating abundance and functional traits reveals new global hotspots of fish diversity. *Nature, 501*(7468), 539-542. https://doi.org/10.1038/nature12529

Sugeha, H. Y. (2010). Recruitment mechanism of the tropical glass eels' genus *Anguilla* in the Poso Estuary, Central Sulawesi Island, Indonesia. *(Journal of Fisheries Sciences), 12*(2), 86–100. https://doi.org/10.22146/jfs.2943

Sugeha, H. Y., Aoyama, J., Miller, M. J., & Tsukamoto, K. (2001). Notes on the diet of tropical freshwater eels from Indonesian waters. *Coastal Marine Science*, *25*(1), 91–98.

Sullivan, S. M. P., Watzin, M. C., & Hession, W. C. (2006). Influence of stream geomorphic condition on fish communities in Vermont, USA. *Freshwater Biology, 51*(8), 1612–1628. https://doi.org/10.1111/j.1365-2427.2006.01610.x

Sures, B., & Knopf, K. (2004). Individual and combined effects of cadmium and 3,3′,4,4′,5-pentachlorobiphenyl (PCB 126) on the humoral immune response in European eel (*Anguilla anguilla*) experimentally infected with larvae of *Anguillicola crassus* (Nematoda). *Parasitology, 128*(4), 445–454. https://doi.org/10.1017/S0031182004004768

Suter, G., Barnthouse, S., Bartell, T., Mill, D. M., & Paterson, S. (1993). *Ecological Risk Assessment*. Lewis Publishers.

Sweeney, B. W., & Newbold, J. D. (2014). Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: A literature review. *Journal of the American Water Resources Association, 50*(3), 560–584. https://doi.org/10.1111/jawr.12141

Tamam, O. A. (2014). The aetiology, histopathology, and ultrastructural features of perianal erythema (Red Anus Syndrome) in the European eel (*Anguilla anguilla*). *PLOS ONE, 9*(3), e92015. https://doi.org/10.1371/journal.pone.0092015

Tamario, C., Sunde, J., Petersson, E., Tibblin, P., & Forsman, A. (2019). Ecological and evolutionary consequences of environmental change and management actions for migrating fish. *Frontiers in Ecology and Evolution, 7*, 271. https://doi.org/10.3389/fevo.2019.00271

Tatsukawa, K. (2003). Eel resources in East Asia. In K. Aida, K. Tsukamoto, & K. Yamauchi (Eds.), *Eel Biology* (pp. 293–298). Springer, Tokyo. https://doi.org/10.1007/978-4-431-65907-5_27

Teichert, N., Borja, A., Chust, G., Uriarte, A., & Lepage, M. (2016). Restoring fish ecological quality in estuaries: Implication of interactive and cumulative effects among anthropogenic stressors. *Science of the Total Environment, 542*, 383–393. https://doi.org/10.1016/j.scitotenv.2015.10.068

Tembo, J. K., Kihia, C. M., Kitaka, N. K., O'Brien, G., Hanzen, C., Mbaru, E., & Wanjiru, K. C. (2023). Diet and trophic interactions between catadromous eels and sympatric fish in Kenyan east-flowing river systems. *Western Indian Ocean Journal of Marine Science, 22*(2), 87–106. https://doi.org/10.4314/wiojms.v22i2.7

Tesch, F. W., & Rohlf, N. (2003). Migration from continental waters to the spawning grounds. In K. Aida, K. Tsukamoto, & K. Yamauchi (Eds.), *Eel Biology* (pp. 223–234). Springer, Tokyo. https://doi.org/10.1007/978-4-431-65907-5_21

Thomaz, S. M., & Cunha, E. R. (2010). The role of macrophytes in habitat structuring in aquatic ecosystems: Methods of measurement, causes, and consequences on animal assemblages' composition and abundance. *Acta Limnologica Brasiliensia, 22*(2), 218–236. https://doi.org/10.1590/S1679-97572010000200011

Thorp, J. H., & Rogers, D. C. (Eds.). (2014). *Thorp and Covich's freshwater invertebrates: Ecology and general biology* (Vol. 1). Elsevier.

Tockner, K., & Stanford, J. A. (2002). Riverine flood plains: Present state and future trends. *Environmental Conservation, 29*(3), 308–330. https://doi.org/10.1017/S0376892902000222

Tóth, R., Czeglédi, I., Kern, B., & Erős, T. (2019). Land use effects in riverscapes: Diversity and environmental drivers of stream fish communities in protected, agricultural, and urban landscapes. *Ecological Indicators, 101*, 742–748. https://doi.org/10.1016/j.ecolind.2019.01.057

Trancart, T., Acou, A., De Oliveira, E., & Feunteun, E. (2013). Forecasting animal migration using SARIMAX: An efficient means of reducing silver eel mortality caused by turbines. *Endangered Species Research, 21*(2), 181–190. https://doi.org/10.3354/esr00515

Trancart, T., Drouineau, H., Laffaille, P., & Bégout, M. L. (2020). Growth responses of eels in modified riverine habitats. *Fisheries Research, 228*, 105596. https://doi.org/10.1016/j.fishres.2020.105596

Trischitta, F., Takei, Y., & Sébert, P. (Eds.). (2013). *Eel physiology*. CRC Press.

Tsukamoto, K. (1992). Discovery of the spawning area for the Japanese eel. *Nature, 356*(6372), 789–791. https://doi.org/10.1038/356789a0

Tsukamoto, K. (2006). Spawning of eels near a seamount. *Nature, 439*(7079), 929. https://doi.org/10.1038/439929a

Tsukamoto, K. (2009). Oceanic migration and spawning of anguillid eels. *Journal of Fish Biology*, *74*(9), 1833–1852. https://doi.org/10.1111/j.1095-8649.2009.02242.x

Tsukamoto, K., & Arai, T. (2001). Facultative catadromy of the eel *Anguilla japonica* between fresh and saline water habitats. *Marine Ecology Progress Series, 220*, 265–276. https://doi.org/10.3354/meps220265

Tsukamoto, K., & Miller, M. J. (2021). The mysterious feeding ecology of leptocephali: A unique strategy of consuming marine snow materials. *Fisheries Science*, *87*(1), 11-29. https://doi.org/10.1007/s12562-020-01369-1

Tsukamoto, K., Aoyama, J., & Miller, M. J. (2009). Present status of the Japanese eel: Resources and recent research. In J. M. Casselman & D. K. Cairns (Eds.), *Eels at the Edge: American Fisheries Society Symposium* (Vol. 58, pp. 21–35). Bethesda, MD: American Fisheries Society.

Tsukamoto, K., Kuroki, M., & Watanabe, S. (2020). Common names for all species and subspecies of the genus *Anguilla*. *Environmental Biology of Fishes*, *103*(8), 985–991. https://doi.org/10.1007/s10641-020-01001-2

Tsukamoto, K., Nakai, I., & Tesch, W. V. (1998). Do all freshwater eels migrate? *Nature, 396*, 635-636. https://go.nature.com/354deVB

Twardochleb, L. A., Olden, J. D., & Larson, E. R. (2013). Quantitative global synthesis of the effects of invasive crayfish on freshwater ecosystems. *Ecology, 94*(7), 1775–1782. https://doi.org/10.1890/12-0973.1

Tzeng, W. N. (2016). Fisheries, stock decline, and conservation of anguillid eel. In W. Dekker & H. Wickström (Eds.), *Biology and Ecology of Anguillid Eels* (pp. 291–324). CRC Press.

United States Environmental Protection Agency (USEPA). (1998). *Guidelines for ecological risk assessment; Notice*. *Federal Register, 63*(26846–26924). Environmental Monitoring

Systems Laboratory, Office of Research and Development, US Environmental Protection Agency, Cincinnati, Ohio.

Unmack, P. J. (2013). Biogeography. In P. Humphries & K. Walker (Eds.), *Ecology of Australian freshwater fishes* (pp. 25–48). CSIRO Publishing. https://doi.org/10.1071/9780643097445

Van Appledorn, M., De Jager, N. R., & Rohweder, J. J. (2024). Low-complexity floodplain inundation model performs well for ecological and management applications in a large river ecosystem. *Journal of the American Water Resources Association, 60*(1), 9–26. https://doi.org/10.1111/1752-1688.13152

Van Beurden, S. J., Engelsma, M. Y., Roozenburg, I., Voorbergen-Laarman, M. A., van Tulden, P. W., Kerkhoff, S., & Haenen, O. L. (2012). Viral diseases of wild and farmed European eel *Anguilla anguilla* with particular reference to the Netherlands. *Diseases of Aquatic Organisms, 101*(1), 69–86. https://doi.org/10.3354/dao02513

Van den Brink, P. J., Choung, C. B., Landis, W., Mayer-Pinto, M., Pettigrove, V., Scanes, P., & Stauber, J. (2016). New approaches to the ecological risk assessment of multiple stressors. *Marine and Freshwater Research, 67*(4), 429-439. https://doi.org/10.1071/MF15351

Van Ginneken, V., Bruijs, M., Murk, T., Palstra, A., & van den Thillart, G. (2009). The effect of PCBs on the spawning migration of European silver eel (*Anguilla anguilla* L.). In K. Tsukamoto, M. J. Miller, & J. Aoyama (Eds.), *Spawning migration of the European eel* (pp. 365–386). Springer, Dordrecht. https://doi.org/10.1007/978-1-4020-9561-2_19

Van Wichelen, J., Verhelst, P., Buysse, D., Belpaire, C., Vlietinck, K., & Coeck, J. (2021). Glass eel (*Anguilla anguilla* L.) behaviour after artificial intake by adjusted tidal barrage management. *Estuarine, Coastal and Shelf Science, 249*, 107127. https://doi.org/10.1016/j.ecss.2020.107127

Van Wichelen, J., Verhelst, P., Perneel, M., Van Driessche, C., Buysse, D., Belpaire, C., & De Troch, M. (2022). Glass eel (*Anguilla anguilla* L. 1758) feeding behaviour during upstream migration in an artificial waterway. *Journal of Fish Biology, 101*(4), 1047-1057. https://doi.org/10.1111/jfb.15093

Verhelst, P., Bruneel, S., Reubens, J., Coeck, J., Goethals, P., Oldoni, D., & Mouton, A. (2018). Selective tidal stream transport in silver European eel (*Anguilla anguilla* L.) – Migration behaviour in a dynamic estuary. *Estuarine, Coastal and Shelf Science, 213*, 260–268. https://doi.org/10.1016/j.ecss.2018.08.030

Verhelst, P., Vandeputte, M., Larmuseau, M. H. D., & Volckaert, F. A. M. (2018). Migration barriers and their effects on eel recruitment. *Journal of Applied Ecology, 55*(2), 497–507. https://doi.org/10.1111/1365-2664.12997

Vezi, M., Downs, C. T., Wepener, V., & O'Brien, G. (2019). Response of zooplankton communities to altered water quality and seasonal flow changes in selected river-dominated estuaries in Kwa-Zulu-Natal, South Africa. *Ecohydrology & Hydrobiology, 19*(3), 393-406. https://doi.org/10.1016/j.ecohyd.2019.05.003

Villamagna, A. M., & Murphy, B. R. (2010). Ecological and socio-economic impacts of invasive water hyacinth (*Eichhornia crassipes*): A review. *Freshwater Biology, 55*(2), 282–298. https://doi.org/10.1111/j.1365-2427.2009.02272.x

Wade, M., O'Brien, G. C., Wepener, V., & Jewitt, G. (2020). Risk assessment of water quantity and quality stressors to balance the use and protection of vulnerable water resources. *Integrated Environmental Assessment and Management, 17*, 110–130. https://doi.org/10.1002/ieam.4350

Wakiya, R., & Mochioka, N. (2021). Contrasting diets of the Japanese eel, *Anguilla japonica*, in the upper and lower areas of Tsuchikawa-gawa River, Kagoshima, Japan. *Ichthyological Research, 68*(1), 145–151. https://doi.org/10.1007/s10228-020-00779-0

Wakiya, R., Itakura, H., & Kaifu, K. (2019). Age, growth, and sex ratios of the giant mottled eel, *Anguilla marmorata*, in freshwater habitats near its northern geographic limit: A comparison to tropical regions. *Zoological Studies, 58*, 34. https://doi.org/10.6620/ZS.2019.58-34

Walker, R. H., & Jones, J. D. (2018). Land use impacts on sedimentation and stream health in temperate watersheds. *Journal of Environmental Management*, 220, 65–75. https://doi.org/10.1016/j.jenvman.2018.05.012:contentReference[oaicite:1]{index=1}

Walsh, L. L., & Tucker, P. K. (2020). Isotopic niche breadth of a generalist mesopredator increases with habitat heterogeneity across its range. *Ecosphere, 11*(12), e03314. https://doi.org/10.1002/ecs2.3314

Ward, J. V., Tockner, K., Arscott, D. B., & Claret, C. (2002**).** Riverine landscape diversity. *Freshwater Biology, 47*(4), 517–539. https://doi.org/10.1046/j.1365-2427.2002.00913.x

Watson, J. E. M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., Thompson, I., Ray, J. C., Murray, K., Salazar, A., McAlpine, C., Potapov, P., Walston, J., Robinson, J. G., Painter, M., Wilkie, D., Filardi, C., Laurance, W. F., Houghton, R. A., Maxwell, S., Grantham, H., Samper, C., Wang, S., Laestadius, L., Runting, R. K., Silva-Chávez, G.

A., Ervin, J., & Lindenmayer, D. (2018). The exceptional value of intact forest ecosystems. *Nature Ecology & Evolution, 2*(4), 599–610. https://doi.org/10.1038/s41559-018-0490-x

Westerberg, H., Miller, M. J., Wysujack, K., Marohn, L., Freese, M., Pohlmann, J. D., & Hanel, R. (2018). Larval abundance across the European eel spawning area: An analysis of recent and historic data. *Fish and Fisheries, 19*(5), 890–902. https://doi.org/10.1111/faf.12271

Wiegers, J. K., Feder, H. M., Mortensen, L. S., Shaw, D. G., Wilson, V. J., & Landis, W. G. (1998). A regional multiple-stressor rank-based ecological risk assessment for the fjord of Port Valdez, Alaska. *Human and Ecological Risk Assessment, 4*, 1125–1173. https://doi.org/10.1080/10807039809383077

Windsor, F. M., Thompson, R. C., & Bannister, R. C. A. (2019). Microplastic ingestion by riverine macroinvertebrates. *Science of the Total Environment, 646*, 68–74. https://doi.org/10.1016/j.scitotenv.2018.07.375

Woolnough, D. A., Downing, J. A., & Newton, T. J. (2009). Fish movement and habitat use depends on water body size and shape. *Ecology of Freshwater Fish, 18*(1), 83-91. https://doi.org/10.1111/j.1600-0633.2008.00349.x

Wootton, R. J. (2012). *Ecology of teleost fishes* (Vol. 1). Springer Science & Business Media.

Wright, A., Brooks, C., Gooseff, M., Howkins, A., & Chignell, S. (2024). An analysis of McMurdo Dry Valleys' lotic habitats within Antarctica's protected area network and addressing gaps in biodiversity protection. *Arctic, Antarctic, and Alpine Research, 56*(1), 2375176. https://doi.org/10.1080/15230430.2024.2375176

Wu, J., & Zhao, F. (2024). Genomic and developmental approaches to enhance reproductive success and growth in eel (*Anguilla* spp.). *International Journal of Aquaculture, 14*, 154–164. https://doi.org/10.5376/ija.2024.14.0016

Wuenschel, M. J., & Able, K. W. (2008). Swimming ability of eels (*Anguilla rostrata*, *Conger oceanicus*) at estuarine ingress: Contrasting patterns of cross-shelf transport? *Marine Biology, 154*, 775–786. https://doi.org/10.1007/s00227-008-1052-4

Wuenschel, M. J., Jugovich, A. R., & Hare, J. A. (2006). Estimating the energy density of fish: The importance of ontogeny. *Transactions of the American Fisheries Society, 135*(2), 379-385. https://doi.org/10.1577/T05-186.1

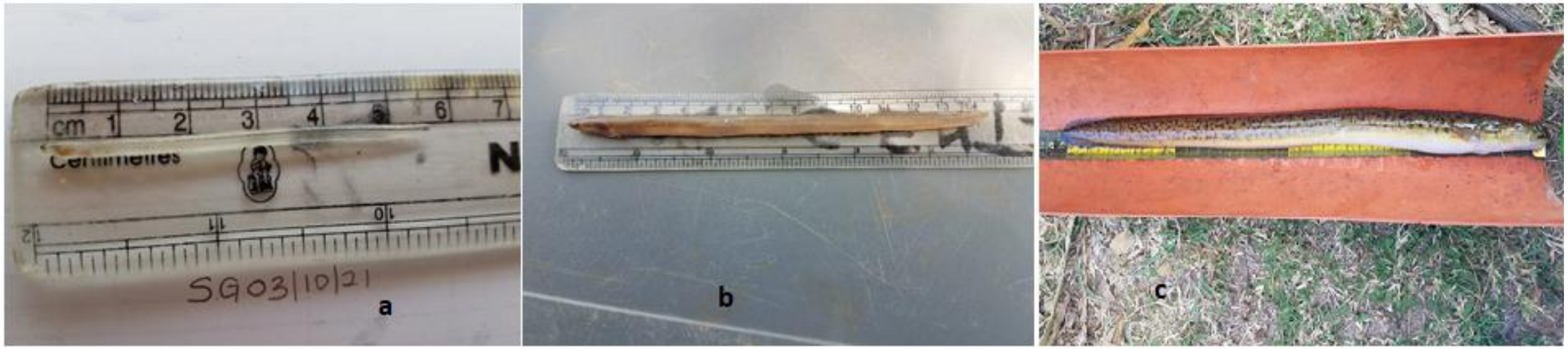
Yeh, S. (2002). *The aquaculture status and its sustainability in Taiwan* [Unpublished manuscript, Department of Aquaculture, National Pingtung University of Science and Technology]. Yuan, Y., Yuan, Y., Dai, Y., Gong, Y., & Yuan, Y. (2022). Development status and trends in the eel farming industry in Asia. *North American Journal of Aquaculture*, *84*(1), 3–17. https://doi.org/10.1002/naaq.10187

Zenimoto, K., Kitagawa, T., Miyazaki, S., Sasai, Y., Sasaki, H., & Kimura, S. (2009). The effects of seasonal and interannual variability of oceanic structure in the western Pacific North Equatorial Current on larval transport of the Japanese eel *Anguilla japonica*. *Journal of Fish Biology*, *74*(9), 1878–1890. https://doi.org/10.1111/j.1095-8649.2009.02247.x

Zumbrägel, C. (2024). Eel on the move: Fish migration and the construction of "flowing spaces" on the Rhine and Weser Rivers (1880–1930). *Water History*, 1–25. https://doi.org/10.1007/s12685-024-00308-x

APPENDICES

Appendix A: Measurement of Glass Eel, Elver, and Yellow Eel



a) Glass eel, b) Elver, and c) Yellow eel caught at River AGS

Appendix B: Examination of Eels' Vomerine for Identification

Open mouth of the eels (a, c) for morphological identification of eels at Upper River Ramisi; (b) shows ocular measurements for identification of eel life stages. ©photo

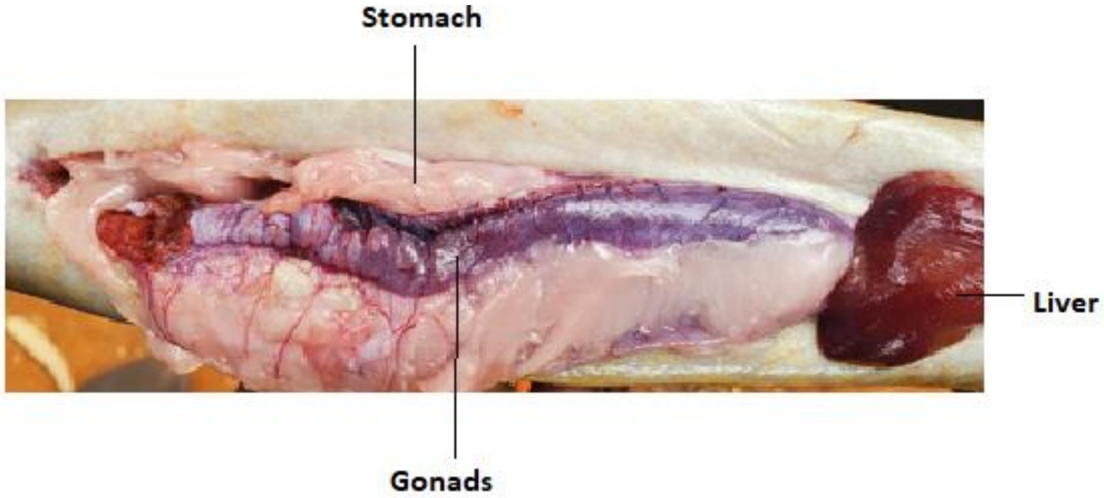
Appendix C: Photo of Silver Eel Stage of *A. bicolor* and Yellow Stage of *A. bengalensis*

(a)*A. bicolor* silver stage characterized by the silver colour, conspicuous lateral line, and a white underbelly caught at River Ramisi and;
(b)Yellow eel stage of *A. bengalensis* caught at Upper River AGS, Kenya. ©photo

Appendix D: Setting of Fyke Nets

(a) Setting a 3mm size- mesh- fyke net at the River Ramisi. (b) Retrieving the fyke net after an overnight stay, and c) retrieving a 10mm fyke net at the River Ramisi. ©photo

Appendix E: A Split Belly of Yellow Eel for Diet Studies, at River AGS



Appendix F: Node Justification for Risk Assessment Scoring

Node name	Node code	Rank (score)	Rank definition and measure for variable	Justification
Alien or invasive species predators on silver eels within the rivers	ALIEN_ESC_PRED	Zero (25)	Absence	Presence of alien fish that are invasive and actively predate on fishes including eels. E.g. Trout, Bass, *Clarias spp*.
		Low (50)	at least one Trout species/ 1 species	
		Moderate (75)	at least one Bass species/ 2 species	
		High (100)	>2 species present/ >2 species	
Alien or invasive species predators within the estuary	ALIEN_EST_PRED	Zero (25)	Absence	Presence of alien fish that are invasive and actively predate on fishes including eels. E.g. Trout, Bass, *Clarias spp*.
		Low (50)	at least one Trout species/ 1 species	
		Moderate (75)	at least one Bass species/ 2 species	
		High (100)	>2 species present/ >2 species	
Alien or invasive species predators within the river	ALIEN_RIV_PRED	Zero (25)	Absence	Presence of alien fish that are invasive and actively predate on fishes including eels. E.g. Trout, Bass, *Clarias spp*.
		Low (50)	at least one Trout species/ 1 species	
		Moderate (75)	at least one Bass species/ 2 species	
		High (100)	>2 species present/ >2 species	
Distance needed to travel to	DIST_SPAWNING	Zero (25)	Need to travel < 1000 km	The distance (km) that silver eels need to travel to the spawning ground from the mouth of the estuary.
		Low (50)	Need to travel > 1000 km but < 2000 km	

spawning grounds		Moderate (75)	Need to travel > 2000 km but < 3000 km	Calculated as the hypothetical straight line eels follow from the mouth of the estuary to the area of the SEC (15.4° - 17.0°S and 60.0°E).
		High (100)	Need to travel > 3000 km	
Distance needed to travel from spawning grounds	DISTANCE_SEC	Zero (25)	Need to travel < 1000 km	The distance (km) that larvae need to travel from the spawning ground to the mouth of the estuary. Calculated as the hypothetical straight line eels follow to the mouth of the estuary from the area of the SEC (15.4° - 17.0°S and 60.0°E).
		Low (50)	Need to travel > 1000 km but < 2000 km	
		Moderate (75)	Need to travel > 2000 km but < 3000 km	
		High (100)	Need to travel > 3000 km	
The number of barriers in the estuary	EST_BARRIERS	Zero (25)	There are no barriers preventing larvae and glass eels entering the rivers	Physical barriers like dam walls, gauging weirs and hydropower turbines can affect the migration of larvae and glass eels into the rivers. The size and the number of barriers will affect the migration of these eels life stage.
		Low (50)	There are some (1-3) barriers but they do not prevent larvae and glass eels entering the rivers	
		Moderate (75)	There are few barriers (4-6) preventing larvae and glass eels entering the rivers	
		High (100)	The barriers prevent the larvae and glass eels to enter the rivers	

The water quality of the estuarine environment that affect eels	EST_QUALITY	Zero (25)	If the eco-classification outcomes from the water quality eco-classification assessment is an A or B the risk zero as there is no potential of change to the variable from natural/ideal state.	Water quality in terms for eels in rivers are classified based on the outcomes for the eco-classification assessment for estuaries (water quality component) that describes the changes in water quality from a natural to a severely modified state.
		Low (50)	If the eco-classification outcomes from the water quality eco-classification assessment is an C the risk low as there is moderate potential of change but still in a suitable condition.	
		Moderate (75)	If the eco-classification outcomes from the water quality eco-classification assessment is an D the risk moderate as there is a threshold of potential for concern and water quality is in an unacceptable condition.	
		High (100)	If the eco-classification outcomes from the water quality eco-classification assessment is an E or F the risk high as the condition is in an	

			unacceptable and unsustainable state.	
Type of estuary mouth (open/closed).	EST_TYPE	Zero (25)	Year round open estuary	The timing/duration the mount of the estuary is open and accessible for eels to migrate into.
		Low (50)	Partially open-closed estuary, tidally.	
		Moderate (75)	Seasonally closed (low flow periods).	
		High (100)		
Fishing of elver and yellow eels	FISHING_ELV_YEL	Zero (25)	Zero kilograms per hectare is harvested.	Intensity of harvest based on actual yields given as kilograms (kg) per hectare per year. CPUE accounted for in the risk profile.
		Low (50)	Less than 18 kg/per hectare/per year is harvested.	
		Moderate (75)	Between 18 and 36 kg/per hectare/per year is harvested	
		High (100)	More than 36 kg/per hectare/per year is harvested.	
Fishing for larvae	FISHING_LARVAE	Zero (25)	Less than 10 tons per year	Intensity of harvest based on actual yields given as tons per year. CPUE accounted for in the risk profile.
		Low (50)	Between 100 and 6000 tons per year	
		Moderate (75)	Between 6000 and 10 000 tons/year	
		High (100)	More than 10 000 ton/year	
Fishing for silver eels	FISHING_SILVER	Zero (25)	no fishers per km	The fishing pressure from fisher using seine and fyke nets (less so gill nets and baited hooks and lure,
		Low (50)	< 3 fishers per km	
		Moderate (75)	3-15 fisher per km	

		High (100)	>15 fishers per km	because they don't feed). has on sliver eels migrating down river
The cover required for the maintenance of fish/invert communities (food for eels)	FOOD_COVER	Zero (25)	Less than 25% of the expected species are intolerant to changes in the cover conditions.	Variable represent the expected fish/invert communities available for eels as food that are cover specific or sensitive to changes the cover conditions as it could influence their life-cycle, feeding ecology and behavioural activities.
		Low (50)	More than 25% but less than 50% of the expected species are intolerant to changes in the cover conditions.	
		Moderate (75)	More than 50% but less than 75% of the expected species are intolerant to changes in the cover conditions.	
		High (100)	More than 75% of the expected species are intolerant to changes in the cover conditions.	
The water quality conditions for the maintenance of fish/invert communities (food for eels)	FOOD_RIV_QUALITY	Zero (25)	Less than 25% of the expected species are intolerant to changes in the water quality conditions.	Variable represent the expected fish/invert communities available for eels as food that are water quality specific or sensitive to changes the water quality conditions as it could influence their life-cycle,
		Low (50)	More than 25% but less than 50% of the expected species are intolerant to changes in	

			the water quality conditions.	feeding ecology and behavioural activities.
		Moderate (75)	More than 50% but less than 75% of the expected species are intolerant to changes in the water quality conditions.	
		High (100)	More than 75% of the expected species are intolerant to changes in the water quality conditions.	
The velocity-depth classes conditions for the maintenance of fish/invert communities (food for eels)	FOOD_VD	Zero (25)	Less than 25% of the expected species are intolerant to changes in the velocity-depth conditions.	Variable represent the expected fish/invert communities available for eels as food that are influenced or sensitive to changes in the velocity-depth conditions as it could influence their life-cycle, feeding ecology and behavioural activities.
		Low (50)	More than 25% but less than 50% of the expected species are intolerant to changes in the velocity-depth conditions.	
		Moderate (75)	More than 50% but less than 75% of the expected species are intolerant to changes in the velocity-depth conditions.	

		High (100)	More than 75% of the expected species are intolerant to changes in the velocity-depth conditions.	
The abundances of indigenous predators on silver eels in the river environment	IND_ESC_PRED	Zero (25)	No predation pressure	Predation pressure as a factor of predator per eel in the system (relative to size class and predator) on eels from any wildlife that has the ability to predate on eels as a potential victim. Excluding alien species.
		Low (50)	1 predator 4 eels	
		Moderate (75)	Moderate predator pressure	
		High (100)	1 predator per eel.	
The abundances of indigenous predators on larvae/glass eels	IND_EST_PRED	Zero (25)	No predation pressure	Predation pressure as a factor of predator per eel in the system (relative to size class and predator) on eels from any wildlife that has the ability to predate on eels as a potential victim. Excluding alien species.
		Low (50)	1 predator 4 eels	
		Moderate (75)	Moderate predator pressure	
		High (100)	1 predator per eel.	
The abundances of indigenous predators on yellow eels	IND_RIV_PRED	Zero (25)	No predation pressure	Predation pressure as a factor of predator per eel in the system (relative to size class and predator) on eels from any wildlife that has the ability to predate on eels as a potential victim. Excluding alien species.
		Low (50)	1 predator 4 eels	
		Moderate (75)	Moderate predator pressure	
		High (100)	1 predator per eel.	
The cover requirements	LARVAE_COV	Zero (25)	More than 75% of estuary (1 km) have	Based on the percentage of marginal vegetation (1km

(depth and plants) for larvae to survive within the estuary			sufficient cover for larvae	from mouth) that provides cover for eel larvae to enter the estuary.
		Low (50)	Between 50 - 75% of estuary (1km) have sufficient cover for larvae.	
		Moderate (75)	Between 30 - 50% of estuary (1km) have sufficient cover for larvae.	
		High (100)	Less than 30% of estuary provide sufficient cover for larvae.	
The velocity depth required for the larvae entering the estuary	LARVAE_VD	Zero (25)	Average depth available is more than 1.3 m deep.	Velocity depth classes provide habitat and connectivity between habitats for eels. No preferences/significant result found for velocity classes.
		Low (50)	Average depth available is between 0.7 and 1.3 m deep.	
		Moderate (75)	Average depth available is between 0.4 and 0.7 m deep.	
		High (100)	Average depth available is less than 0. 4m deep.	
The abundances of predators on silver eels in the marine environment	MAR_SIL_PRED (Marine silver eel predation)	Zero (25)	<10km	The exposed distance (km) within the continental shelf, as assumed predation numbers are higher in continental shelf.
		Low (50)	> 10 km but < 200 km	
		Moderate (75)	> 200 km but < 500 km	
		High (100)	> 500 km	
The water quality of	MARINE_QUALITY (Marine water quality)	Zero (25)	If the eco-classification outcomes from the water	Water quality in terms for eels in rivers are classified

marine environment that can affect spawning			quality eco-classification assessment is an A or B the risk zero as there is no potential of change to the variable from natural/ideal state.	based on the outcomes for the eco-classification assessment for estuaries (water quality component) that describes the changes in water quality from a natural to a severely modified state.
		Low (50)	If the eco-classification outcomes from the water quality eco-classification assessment is an C the risk low as there is moderate potential of change but still in a suitable condition.	
		Moderate (75)	If the eco-classification outcomes from the water quality eco-classification assessment is an D the risk moderate as there is a threshold of potential for concern and water quality is in an unacceptable condition.	
		High (100)	If the eco-classification outcomes from the water quality eco-classification assessment is an E or F the risk high as the condition is in an unacceptable and unsustainable state.	

The number of barriers in the river affecting upstream migration	RIV_BARRIERS (River barriers)	Zero (25)	There are no barriers preventing larvae and glass eels entering the rivers	Physical barriers like dam walls, gauging weirs and hydropower turbines can affect the migration of larvae and glass eels into the rivers. The size and the number of barriers will affect the migration of these eels life stage.
		Low (50)	There are some (1-3) barriers but they do not prevent larvae and glass eels entering the rivers	
		Moderate (75)	There are few barriers (4-6) preventing larvae and glass eels entering the rivers	
		High (100)	The barriers prevent the larvae and glass eels to enter the rivers	
The number of barriers in the river affecting downstream migration	RIV_ESC_BARRIER (River escapement barrier)	Zero (25)	There are no barriers affecting eel escapement	Physical barriers like dam walls, gauging weirs and hydropower turbines affect the escapement of silver eels back into the estuary. The higher the number of barriers the lower the success of eel escapement back into the estuaries.
		Low (50)	There are a few barriers but does not influences eel escapement	
		Moderate (75)	There are a number of barriers which effect more than 50 % of eel escapement	
		High (100)	The barriers prevent the eels to pass and thus no escapement takes place	
The water quality required for the	SIL_RIV_QUALITY (Silver eel River Quality)	Zero (25)	If the eco-classification outcomes from the water quality eco-classification assessment is an A or B	Water quality in terms for eels in rivers are classified based on the outcomes for the eco-classification

maintenances of silver eels in the river			the risk zero as there is no potential of change to the variable from natural/ideal state.	assessment (water quality component) that describes the changes in water quality from a natural to a severely modified state.
		Low (50)	If the eco-classification outcomes from the water quality eco-classification assessment is an C the risk low as there is moderate potential of change but still in a suitable condition.	
		Moderate (75)	If the eco-classification outcomes from the water quality eco-classification assessment is an D the risk moderate as there is a threshold of potential for concern and water quality is in an unacceptable condition.	
		High (100)	If the eco-classification outcomes from the water quality eco-classification assessment is an E or F the risk high as the condition is in an unacceptable and unsustainable state.	
The required depth for	SILVER_ESC_DEPTH (Silver eel escapement depth)	Zero (25)	Average depth available is more than 1.3 m deep.	Velocity depth classes provide habitat and connectivity between habitats

silver eel escapement		Low (50)	Average depth available is between 0.7 and 1.3 m deep.	for eels. No preferences/significant result found for velocity classes.
		Moderate (75)	Average depth available is between 0.4 and 0.7 m deep.	
		High (100)	Average depth available is less than 0. 4m deep.	
The Location in the Western Indian Ocean (tropical vs temperate)	WIO_LOCATION (West Indian Ocean location)	Zero (25)	Within 10 ° North or South from the equator	Tropical region near the equator are considered to have more preferred climate and environmental preferences for eels and the further the location mores away the greater the risk to the eels to have these conditions.
		Low (50)	Within 10° to 15° North or South from the equator	
		Moderate (75)	Within 15° to 25° North or South from the equator	
		High (100)	Further than 25° North or South from the equator	
The water quality required for the maintenances of yellow eels	YEL_RIV_QUALITY	Zero (25)	If the eco-classification outcomes from the water quality eco-classification assessment is an A or B the risk zero as there is no potential of change to the variable from natural/ideal state.	Water quality in terms for eels in rivers are classified based on the outcomes for the eco-classification assessment (water quality component) that describes the changes in water quality from a natural to a severely modified state.
		Low (50)	If the eco-classification outcomes from the water quality eco-classification assessment is an C the	

			risk low as there is moderate potential of change but still in a suitable condition.	
		Moderate (75)	If the eco-classification outcomes from the water quality eco-classification assessment is an D the risk moderate as there is a threshold of potential for concern and water quality is in an unacceptable condition.	
		High (100)	If the eco-classification outcomes from the water quality eco-classification assessment is an E or F the risk high as the condition is in an unacceptable and unsustainable state.	
The cover (including depth) required for yellow eels	YELLOW_COV (Yellow eel maintenance cover)	Zero (25)	More than 75% of river/risk region have sufficient cover for yellow eels	Based on the percentage of marginal vegetation in the river or risk region that provides cover for yellow eels.
		Low (50)	Between 50 - 75% of the river of risk region have sufficient cover for yellow eels.	
		Moderate (75)	Between 30 - 50% river of risk region have	

			sufficient cover for yellow eels.	
		High (100)	Less than 30% of river or risk region provide sufficient cover for yellow eels.	
The velocity depth requirement of yellow eels	YELLOW_VD (Yellow eel maintenance Velocity and Depth	Zero (25)	Average depth available is more than 1.3 m deep.	Velocity depth classes provide habitat and connectivity between habitats for eels. No preferences/significant result found for velocity classes.
		Low (50)	Average depth available is between 0.7 and 1.3 m deep.	
		Moderate (75)	Average depth available is between 0.4 and 0.7 m deep.	
		High (100)	Average depth available is less than 0. 4m deep.	
Fishing for silver eels in the marine environment	MAR_SIL_HARVEST (Marine Silver eel harvest)	Zero (25)	Zero kilograms per hectare is harvested.	Intensity of harvest based on actual yields given as kilograms (kg) per hectare per year. CPUE accounted for in the risk profile.
		Low (50)	Less than 2 kg/per hectare/per year is harvested.	
		Moderate (75)	Between 2 and 8 kg/per hectare/per year is harvested	
		High (100)	More than 8 kg/per hectare/per year is harvested.	
Altitude of the river	RIVER_ALT (River altitude)	Zero (25)	Average river altitude of catchment/risk region below 180m	The average altitude of the river/s the eels migrate into from the estuary.

		Low (50)	Average river altitude of catchment/risk region between 180m and 620m	
		Moderate (75)	Average river altitude of catchment/risk region between 620m and 1400m	
		High (100)	Average river altitude of catchment/risk region above 1400m	
Presence of Alien parasites in the elver yellow eels	ELV_YEL_PARA (Elver-Yellow eel parasites)	Zero (25)	no parasite	The intensity/severity of alien parasite infestation on anguillid eels
		Low (50)	Parasite presence but no mortalities	
		Moderate (75)	High infestation low mortalities	
		High (100)	Mass mortalities in eels	

Appendix G: Habitat Integrity Assessment Tool

The Kleynhans method

- Impacts along the River recorded in respect to specified criteria and impact categories
- Two assessments were carried out namely; In-stream and riparian modifiers
- Standard weightings were applied to each criteria
- Overall weighted scores were calculated
- Assessments were carried out at every 105 km distance.

Habitat Integrity Criteria

Instream and Riparian habitat modifiers		**Impact categories**
• Water abstraction • Flow modification • Bed modification • Channel modification • Water quality • Inundation	• Exotic macrophytes • Exotic fauna • Solid waste disposal • Vegetation removal • Exotic vegetation encroachment • Bank erosion	• None 0 • Small 1 - 5 • Moderate 6 - 10 • Large 11 - 15 • Serious 16 – 20 • Critical 21 - 25

Weights of modifiers

INSTREAM MODIFIER	WEIGHT	RIPARIAN MODIFIER	WEIGHT
Water abstraction	14	Bank erosion	14
Water quality	14	Indigenous vegetation removal	13
Bed modification	13	Water abstraction	13
Channel modification	13	Water quality	13
Flow modification	13	Exotic vegetation encroachment	12
Inundation	10	Channel modification	12
Exotic macrophytes	9	Flow modification	12
Exotic fauna	8	Inundation	11
Solid waste disposal	6		
TOTAL	**100**	**TOTAL**	**100**

Overall habitat integrity calculated using the formula;

$$100 - \left[\frac{\sum \text{Each score x Each weight}}{25}\right] \quad \text{………………………………………………………………}1$$

Classes for assessment of habitat integrity

CLASS	DESCRIPTION	SCORE (% OF TOTAL)
A	Unmodified, natural.	100
B	Largely natural with few modifications. A small change from natural in habitats and biotas may have taken place, but the ecosystem functions are essentially unchanged.	80-99
C	Moderately modified. A loss of and change from natural habitats and biotas has occurred, but the basic ecosystem functions are still predominantly unchanged.	60-79
D	Largely modified. A large loss of natural habitats, biotas and basic ecosystem functions has occurred.	40-59
E	The losses of natural habitats, biotas and basic ecosystem functions are extensive.	20-39
F	Modifications have reached a critical level and the lotic system has been completely modified, with an almost complete loss of natural habitats and biotas. In the worst instances, basic ecosystem functions have been destroyed and the changes are irreversible.	0-19

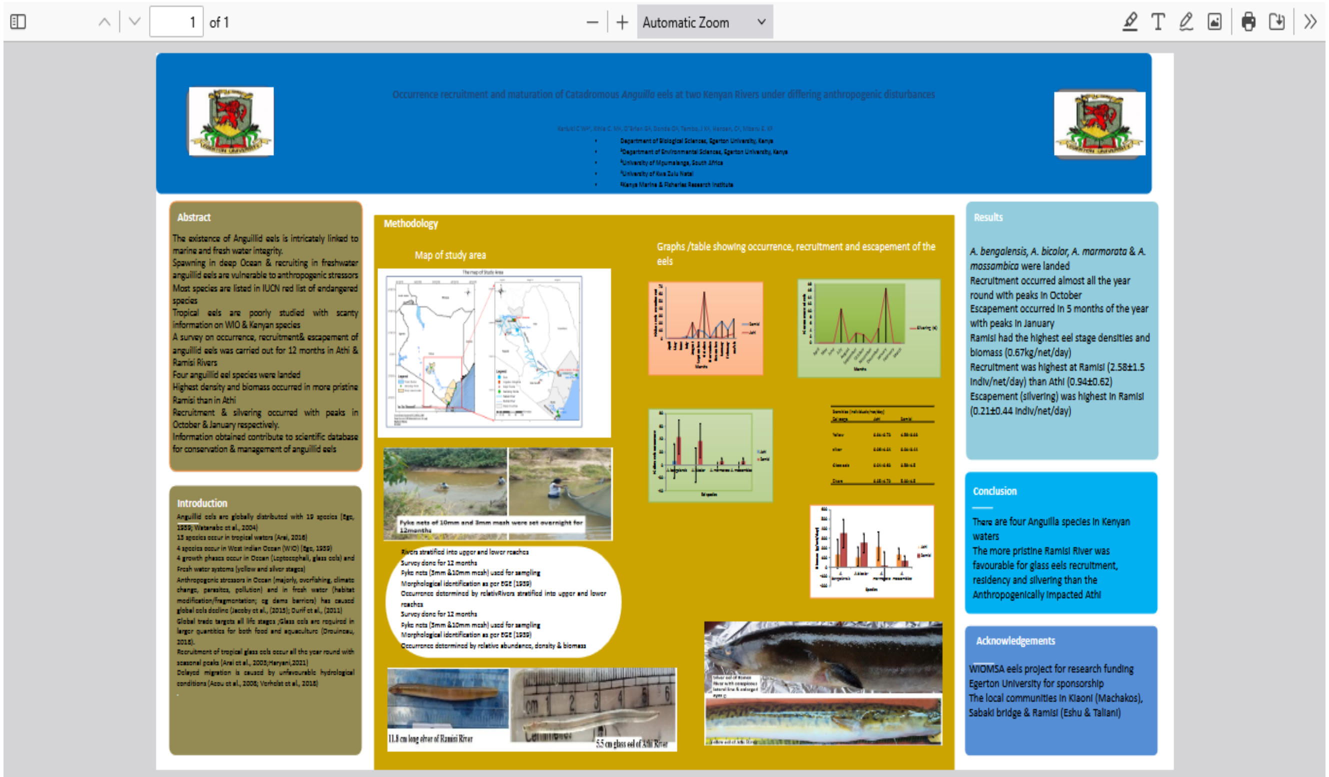
Sampling site guide

Habitat Parameter	Condition Category			
	Optimal	**Suboptimal**	**Marginal**	**Poor**
1. Epifaunal Substrate/ Available Cover	Greater than 70% of substrate favorable for epifaunal colonization and fish cover; mix of snags, submerged logs, undercut banks, cobble or other stable habitat and at stage to allow full colonization potential (i.e., logs/snags that are not new fall and not transient).	40-70% mix of stable habitat; well-suited for full colonization potential; adequate habitat for maintenance of populations; presence of additional substrate in the form of newfall, but not yet prepared for colonization (may rate at high end of scale).	20-40% mix of stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed.	Less than 20% stable habitat; lack of habitat is obvious; substrate unstable or lacking.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
2. Embeddedness	Gravel, cobble, and boulder particles are 0- 25% surrounded by fine sediment. Layering of cobble provides diversity of niche space.	Gravel, cobble, and boulder particles are 25- 50% surrounded by fine sediment.	Gravel, cobble, and boulder particles are 50- 75% surrounded by fine sediment.	Gravel, cobble, and boulder particles are more than 75% surrounded by fine sediment.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
3. Velocity/Depth Regime	All four velocity/depth regimes present (slowdeep, slow-shallow, fastdeep, fast-shallow).(Slow is < 0.3 m/s, deep is > 0.5 m.)	Only 3 of the 4 regimes present (if fast-shallow is missing, score lower than if missing other regimes).	Only 2 of the 4 habitat regimes present (if fastshallow or slow-shallow are missing, score low).	Dominated by 1 velocity/ depth regime (usually slow-deep).
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
4. Sediment Deposition	Little or no enlargement of islands or point bars and less than 5% of the bottom affected by sediment deposition.	Some new increase in bar formation, mostly from gravel, sand or fine sediment; 5-30% of the bottom affected; slight deposition in pools.	Moderate deposition of new gravel, sand or fine sediment on old and new bars; 30-50% of the bottom affected; sediment deposits at obstructions, constrictions, and bends; moderate deposition of pools prevalent.	Heavy deposits of fine material, increased bar development; more than 50% of the bottom changing frequently; pools almost absent due to substantial sediment deposition.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
5. Channel Flow Status	Water reaches base of both lower banks, and minimal amount of channel substrate is exposed.	Water fills >75% of the available channel; or <25% of channel substrate is exposed.	Water fills 25-75% of the available channel, and/or riffle substrates are mostly exposed.	Very little water in channel and mostly present as standing pools.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
6. Channel Alteration	Channelization or dredging absent or minimal; stream with normal pattern.	Some channelization present, usually in areas of bridge abutments; evidence of past channelization, i.e., dredging, (greater than past 20 yr) may be present, but recent channelization is not present.	Channelization may be extensive; embankments or shoring structures present on both banks; and 40 to 80% of stream reach channelized and disrupted.	Banks shored with gabion or cement; over 80% of the stream reach channelized and disrupted. Instream habitat greatly altered or removed entirely.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0

Habitat Parameter	Condition Category			
	Optimal	**Suboptimal**	**Marginal**	**Poor**
7. Frequency of Riffles (or bends)	Occurrence of riffles relatively frequent; ratio of distance between riffles divided by width of the stream <7:1 (generally 5 to 7); variety of habitat is key. In streams where riffles are continuous, placement of boulders or other large, natural obstruction is important.	Occurrence of riffles infrequent; distance between riffles divided by the width of the stream is between 7 to 15.	Occasional riffle or bend; bottom contours provide some habitat; distance between riffles divided by the width of the stream is between 15 to 25.	Generally all flat water or shallow riffles; poor habitat; distance between riffles divided by the width of the stream is a ratio of >25.
SCORE	20 19 18 17 16	15 14 13 12 11	10 9 8 7 6	5 4 3 2 1 0
8. Bank Stability (score each bank) Note: determine left or right side by facing downstream.	Banks stable; evidence of erosion or bank failure absent or minimal; little potential for future problems. <5% of bank affected.	Moderately stable; infrequent, small areas of erosion mostly healed over. 5-30% of bank in reach has areas of erosion.	Moderately unstable; 30-60% of bank in reach has areas of erosion; high erosion potential during floods.	Unstable; many eroded areas; "raw" areas frequent along straight sections and bends; obvious bank sloughing; 60-100% of bank has erosional scars.
SCORE ___ (LB)	Left Bank 10 9	8 7 6	5 4 3	2 1 0
SCORE ___ (RB)	Right Bank 10 9	8 7 6	5 4 3	2 1 0
9. Vegetative Protection (score each bank)	More than 90% of the streambank surfaces and immediate riparian zone covered by native vegetation, including trees, understory shrubs, or nonwoody macrophytes; vegetative disruption through grazing or mowing minimal or not evident; almost all plants allowed to grow naturally.	70-90% of the streambank surfaces covered by native vegetation, but one class of plants is not wellrepresented; disruption evident but not affecting full plant growth potential to any great extent; more than one-half of the potential plant stubble height remaining.	50-70% of the streambank surfaces covered by vegetation; disruption obvious; patches of bare soil or closely cropped vegetation common; less than onehalf of the potential plant stubble height remaining.	Less than 50% of the streambank surfaces covered by vegetation; disruption of streambank vegetation is very high; vegetation has been removed to 5 centimeters or less in average stubble height.
SCORE ___ (LB)	Left Bank 10 9	8 7 6	5 4 3	2 1 0
SCORE ___ (RB)	Right Bank 10 9	8 7 6	5 4 3	2 1 0
10. Riparian Vegetative Zone Width (score each bank riparian zone)	Width of riparian zone >18 meters; human activities (i.e., parking lots, roadbeds, clear-cuts, lawns, or crops) have not impacted zone.	Width of riparian zone 12-18 meters; human activities have impacted zone only minimally.	Width of riparian zone 6- 12 meters; human activities have impacted zone a great deal.	Width of riparian zone <6 meters: little or no riparian vegetation due to human activities.
SCORE ___ (LB)	Left Bank 10 9	8 7 6	5 4 3	2 1 0
SCORE ___ (RB)	Right Bank 10 9	8 7 6	5 4 3	2 1 0
Total Score ________				

Kleynhans CJ. 1996. A qualitative procedure for the assessment of the habitat integrity status of the Luvuvhu River (Limpopo system, South Africa). *Journal of Aquatic Ecosystem Health* 5: 41–54.

Appendix H: 12th WIOMSA Science Symposium Poster Presentation



Appendix I: Publication 1

Vol. 16(2), pp. 13-25, July-December 2024
DOI: 10.5897/IJFA2023.0855
Article Number: 04ED35972509
ISSN 2006-9839

http://www.academicjournals.org/IJFA

ACADEMIC JOURNALS

International Journal of Fisheries and Aquaculture

Full Length Research Paper

Population structure and habitat preference of four anguillid eels occurring in the River Athi-Galana-Sabaki (AGS) and the River Ramisi, Kenya

Catherine Wanjiru Kariuki[1]*, Charles Maina Kihia[2], Gordon O'brien[3], Oscar Omondi Donde[1], Japhet Kadzo Tembo[2] and Emmanuel Mbaru[4]

[1]Department of Environmental Sciences, Egerton University, 20115, Kenya.
[2]Department of Biological Sciences, Egerton University, 20115, Kenya.
[3]School of Biology and Environmental Science, University of Mpumalanga, 1201 South Africa.
[4]Kenya Marine and Fisheries Research Institute, 80101, Kenya.

Received 11 August, 2023; Accepted 27 December, 2023

Knowledge about the population structure of freshwater eels and their habitats is crucial for conservation and exploitation purposes, but this information is lacking for Kenyan anguillid eels. Therefore, there was a need to carry out the current study. A total of 304 eels were landed between May 2021 and June 2022, with the longer eels occurring in the River AGS compared to those of the River Ramisi (t-test; P < 0.01). The study revealed a significant contrast among species populations between the rivers (p < 0.05), with the highest relative abundance occurring in size class 50 to 60 cm (36%). Size class >80 cm had the heaviest eels, with River AGS having heavier eels than Ramisi. River Ramisi had a higher catch per unit effort (CPUE) in all eel growth stages than the River AGS. Biotopes of River Ramisi comprised a higher number of eels (61.6%) than those of the River AGS (38.4%). The study confirmed that Kenyan rivers consist of freshwater eels of different size classes that prefer different biotopes.

Key words: *Anguilla* eels, River AGS, River Ramisi, abundance, size class, habitat preference.

INTRODUCTION

Fish communities often comprise discrete and nonrandom species assemblages (Rahel and Nibbelink, 1999) that exhibit population characteristics determined to a large extent by a combination of abiotic (temperature, water depth, currents, bottom substrates, cover, oxygen levels, dissolved minerals, among others) and biotic (alien species, predators, and others) factors (Gilliam and Fraser, 2001). Species distribution and abundance within a particular predator, and others) factors (Gilliam and Fraser, 2001). Species distribution and abundance within a particular environment are determined both by tolerance to physical conditions environment are determined both by tolerance to physical conditions and interactions with other organisms (Gebrekiros, 2016).

Generally, larger water bodies are more suitable for growth and survival because they encompass more

*Corresponding author. E-mail: ckariuki@egerton.ac.ke.

Appendix J: Publication 2

Vol. 17(x), pp. xx-xx, xxx 2025
DOI: 10.5897/AJESTxxxx
Article Number: xxxxxx
ISSN: 1996-0786

http://www.academicjournals.org/AJEST

African Journal of Environmental Science and Technology

Full Length Research Paper

Growth response of anguillid eels in two Kenyan Rivers with varying levels of habitat integrity

Catherine Wanjiru Kariuki[1]*, Charles Maina Kihia[1], Gordon O'Brien[3], Oscar Omondi Donde[2], and Japhet Kadzo Tembo[1]

[1]Department of Biological Sciences, Faculty of Science, Egerton University, P. O. Box, 536, 20115, Egerton, Kenya.
[2]Department of Environmental Sciences, Faculty of Environment and Resources Development, Egerton University, P. O. Box, 536, 20115, Egerton, Kenya.
[3]Charles Sturt University, The Grange Chancellery, Panorama Avenue, Bathurst NSW Australia 2795 (ABN: 83 878 708 551).

Received 23 October, 2024; Accepted 15 January, 2025

Eels of the genus *Anguilla* have a unique life cycle that involves long migrations between freshwater and marine environments, making a significant contribution to biodiversity and fisheries. This study examined the growth and health of Anguillid eels in two Kenyan rivers with differing habitat integrity; the Ramisi and the Athi-Galana-Sabaki (AGS) Rivers. Four species which are the *Anguilla bengalensis*, *Anguilla bicolor*, *Anguilla marmorata*, and *Anguilla mossambica* were assessed for morphometric variations and growth patterns. A Kruskal-Wallis test ($p \leq 0.05$) revealed significant differences in habitat integrity between the rivers. Although *A. bengalensis* and *A. bicolor* showed no significant differences in mean lengths, *A. marmorata* and *A. mossambica* were significantly larger in the Ramisi River. Growth patterns varied: *A. bengalensis* and *A. marmorata* exhibited contrasting growth forms between rivers, while *A. bicolor* and *A. mossambica* displayed positive allometric growth in both. Hepatosomatic Index (HSI) and Condition factor (K) analyses indicated better liver health and overall condition of most species in the River Ramisi. These findings enhance our understanding of Anguillid eels' responses to habitat variations and provide essential insights for biodiversity conservation and fisheries management strategies in Kenya, where information on these species is limited.

Key words: Anguillid eels, habitat integrity, length-weight, hepatosomatic index (HSI), Condition factor (k).

INTRODUCTION

The genus *Anguilla* comprises 19 species of catadromous migratory eels distributed across temperate and tropical regions globally, inhabiting freshwater, brackish, and coastal ecosystems (Arai, 2016; Jacoby et al., 2015). These eels are characterized by a complex life cycle that spans both marine and freshwater habitats. While classified as freshwater fishes, Anguillid eels spend critical stages of their lives in marine environments (Righton et al., 2021). Eggs hatch in oceanic spawning grounds, producing leptocephalus larvae that drift with ocean currents toward continental shelves. Upon reaching estuaries, the larvae metamorphose into glass

*Corresponding author. E-mail: ckariuki@egerton.ac.ke.

Appendix K: Research Permit

REPUBLIC OF KENYA

NATIONAL COMMISSION FOR SCIENCE,TECHNOLOGY & INNOVATION

Ref No: **664051**

Date of Issue: **09/June/2025**

RESEARCH LICENSE

This is to Certify that Ms.. Catherine Wanjiru Kariuki of Egerton University, has been licensed to conduct research as per the provision of the Science, Technology and Innovation Act, 2013 (Rev.2014) in Kilifi, Kwale on the topic: ASSESSMENT OF ANGUILLID EELS' BIO-ECOLOGY AND ECOLOGICAL STATUS IN ATHI-GALANA-SABAKI AND RAMISI RIVERS for the period ending : 09/June/2026.

License No: **NACOSTI/P/25/4174702**

664051

Applicant Identification Number

Deputy Director
NATIONAL COMMISSION FOR SCIENCE,TECHNOLOGY & INNOVATION

Verification QR Code

NOTE: This is a computer generated License. To verify the authenticity of this document, Scan the QR Code using QR scanner application.

See overleaf for conditions