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Assessment of Teleost Diversity in Faro Lake (Natural Oriented Reserve "Capo Peloro Lagoon", Messina, Italy)

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I THANK NATURE AND ITS CREATOR FOR GIVING ME THE OPPORTUNITY TO LIVE AND SHARE THIS WORLD WITH FISH. I DEDICATE THIS WORK TO THOSE WHO, LIKE ME, ARE ABLE TO MARVEL AT THE HARMONY AND EXTRAORDINARY ADAPTABILITY OF FISH, SEEING THEM AS AN EXAMPLE OF INSPIRATION

TABLE OF CONTENTS

INDEX OF FIGURES

INDEX OF TABLES

ABSTRACT

Coastal transitional environments, such as coastal lagoon, are highly biodiverse ecosystems. These environments are particularly sensitive to environmental changes, which significantly influence the dynamics of the communities inhabiting them.

The present monitoring study, conducted in Faro Lake, utilized non-invasive techniques such as visual census and BRUVs (Baited Remote Underwater Video systems) with the aim of providing for the first time information on the distribution patterns and dynamics of the fish community in the area. During one year of monitoring, the visual census technique allowed to record 50,943 individuals belonging to 52 species, proving to be the most efficient method of data collection. BRUVs, on the other hand, recorded the presence of 18 species but did not allow for a reliable estimation of abundances.

Distribution patterns and seasonal fluctuations in species richness and abundance were analyzed in relation to chemical and physical environmental parameters. The results revealed significant variations in fish community composition, determining which environmental factors have a greater influence on community assemblages. The classification of species into functional ecological categories (residents, seasonal migrants, occasional visitors) allowed to discover differences in habitat utilization by the different categories and to analyze variations in the abundance of the most representative species.

Although several authors have contributed to the knowledge of the teleost fauna in Faro Lake over the years, the data available are dated and fragmentary. The data collected were compared with those from previous studies, with particular reference to the work of Potoschi et al. (2002), which recorded 38 species in common with the present study, using sampling methodologies rather than observation-based techniques.

The results confirm that non-invasive methods are an effective strategy for monitoring transitional environments such as Lake Faro. The information gathered provides a basis for future studies aimed at better understanding the dynamics of fish communities and the factors that influence them, offering valuable data for possible management and conservation strategies focused on preserving these delicate ecosystems.

1 INTRODUCTION

1.1 Geographical context: The Strait of Messina

The Strait of Messina (Fig. 1), located between northeastern Sicily and southern Calabria (Italy) is a sea area formed through tectonic processes and is currently known worldwide for its bio-geomorphological features. The characteristics of the Strait of Messina result from unique biotic and abiotic factors present in the area. The underwater profile of the Strait resembles a mountain, with markedly different slopes on opposite sides. In the Tyrrhenian Sea, the seabed gently slopes to around 1000 meters in the Gulf of Milazzo, reaching depths of 2000 meters near the Stromboli Island. In contrast, in the southern part (Ionian Sea), the slope is much steeper, and the depth between the cities of Messina and Reggio Calabria already exceeds 500 meters, reaching 2000 meters in the connecting line between Capo dell' Armi and Capo Taormina. The narrowest width (3150 meters) of the strait is found along the line connecting Ganzirri (Sicily) and Punta Pezzo (Calabria). This area corresponds to an underwater relief, named "Sella" with depths ranging from 80 to 115 meters. In this area, the bottom is irregular, with a maximum depth of 115 meters (Spanò & De Domenico, 2017). The Strait of Messina represents the geographic point of connection between the Ionian Sea and Tyrrhenian Sea. Although the two basins are geographically continuous, they are distinct in physical and chemical parameters and physiographic characteristics (De Domenico, 2016). The significant difference in height between surface water masses generates very strong currents at each tidal change. The flow moving from North to South is termed "*scendente*", while the flow from South to North is termed "*montante*". The alternation and pattern of these currents are intricately linked to lunar phases, influencing their duration and timing. Each current phase spans six hours, with a brief period of approximately 13 minutes between phases referred to as "*stanca*". With four current phases and four "*stanca*" intervals occurring daily, successive days exhibit a difference of approximately 52 minutes. Peak current intensity coincides with new moon and full moon days. This phenomenon derives from the robust gravitational attraction exerted by the Moon's alignment with or opposition to the Sun during these phases. Maximum currents are typically observed on the day preceding the new moon and persist for three subsequent days. Current intensity gradually diminishes until reaching a minimum two days after the first quarter moon, then escalates to its zenith on the full moon day and the subsequent two days. Another period of reduced intensity occurs two days after the last quarter moon. The "*scendente*" and "*montante*" currents follow distinct paths due to the variable coastline

configuration, contributing to the unpredictable nature of local current flows (Longo, 1882; Defant, 1940). The hydrodynamic characteristics and the unique ecological conditions of the strait are closely linked to its geological structure and the intense tectonic activity characterizing the area. The distribution of sediments in the Strait of Messina is influenced by seismic activity and intense hydrodynamic patterns, making this area a unique environment within the Mediterranean Sea and supporting considerable species diversity in both flora and fauna (Costa F. & Genovese L., 2009). Similarly, brackish lagoons in Italy, while sharing many common characteristics, exhibit a variety of unique features that, along with the ecological complexity of the Strait, contribute to a rich and complex landscape offering significant research opportunities.

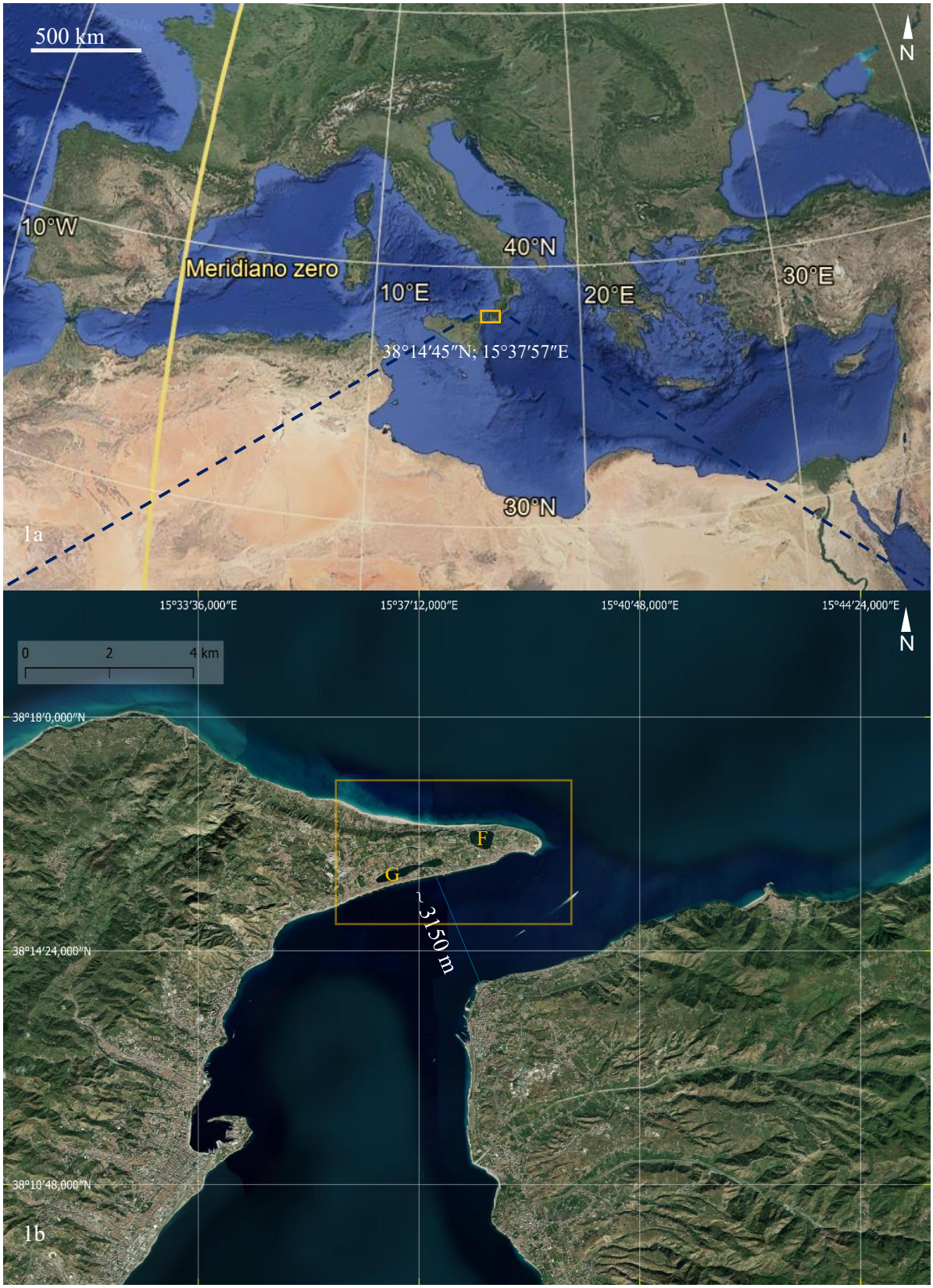

Figure 1; 1a, Map of the Mediterranean Sea with the Strait of Messina highlighted (orange box). 1b, Map of the Strait of Messina with the Natural Oriented Reserve "Laguna di Capo Peloro" indicated (orange box), the Lago di Ganzirri (G) and Lago di Faro (F) marked.

1.2 Semi-enclosed aquatic ecosystems and their importance

Coastal aquatic ecosystems are crucial for biodiversity conservation (Cognetti & Maltagliati, 2000) providing essential ecosystem services, ranging from nutrient cycling to climate regulation (Newton et al., 2018). Among these, semi-enclosed water basins, such as coastal lagoons, are distinguished by their peculiar ecological functioning, determined by a delicate balance between continental and marine influences (Sarno et al., 1993). Brackish environments are generally connected to the sea through natural or artificial canals (Fig. 2), often maintaining

Figure 2; An example of connecting canal: the English Canal, which periodically connects Faro Lake with the Tyrrhenian Sea.

a permanent connection that allows the water exchanges (López-Vila et al., 2021; Pérez-Ruzafa et al., 2007).

Coastal lagoons are transitional zones where water mass is influenced by the influx of freshwater from rainfall, runoff, springs, and other external sources, creating a dynamic and heterogeneous environment often characterized by shallow depths (Newton et al., 2018; Pérez-Ruzafa et al., 2011;Tiralongo & Baldacconi, 2014). A small aquatic ecosystem represents an environment in which complex biological processes take place, highlighting the interactions

between living organisms and their environment (Zweig, 1977). Coastal lagoons are ecologically relevant environments due to their high productivity and biodiversity (Simantiris et al., 2021; Lacoste et al., 2023). These aspects depend on the variety of habitats they offer. Another key factor is the type of substrate, which influences the distribution of species and determines the composition of biological assemblages (Canessa et al., 2021; Simantiris et al., 2021). Many coastal lagoons are dominated by muddy bottoms, while others are characterized by a mosaic of natural bottom including sandy, muddy, gravelly areas, and rocky outcrops with stones of different sizes. Additionally, man-made structures such as rock, cement, or wooden docks and jetties are common in many of these environments (Newton, 2018; Perez, 2007; Manzo, 2010; Lanzoni et al., 2021; Pérez-Ruzafa et al., 2019). Their habitats represent essential refuges and breeding grounds for various aquatic species, including fish, due to their high variability and the diversity of environments they host. This environmental diversity allows numerous species to find shelter, feed, reproduce and take advantage of development opportunities (Zamora-López et al., 2023; Cataudella et al., 2015; Verdiell-Cubedo et al., 2013).

Considering these factors, the establishment of a Natural Oriented Reserve (NOR) can provide significant benefits for the protection, management, and conservation of coastal lagoon ecosystems. A NOR is a protected reserve that permits specific sustainable activities, such as bivalve farming, to be integrated with environmental conservation goals. This type of reserve enables rigorous monitoring and active management of natural resources, ensuring that exploitation practices, such as fishing and aquaculture, are conducted sustainably and do not compromise the area's biodiversity and ecological functionality. Additionally, NORs facilitate scientific research and continuous monitoring, providing essential data to better understand ecological dynamics and anthropogenic impacts. The creation of an NOR also promotes environmental awareness and education, engaging local communities and stakeholders in the active protection of the ecosystem. Ultimately, NORs represent an effective tool for balancing conservation needs with sustainable resource use, ensuring the long-term protection of coastal lagoon ecosystems and the ecosystem services they provide. In Italy, the establishment and management of NORs are regulated by the “Framework Law on Protected Areas” (Legge 6 dicembre 1991, n. 394. Legge quadro sulle aree protette. Gazzetta Ufficiale della Repubblica Italiana, n. 292, 12 dicembre 1991.). This legislation provides a comprehensive framework for the protection and sustainable management of natural areas, including coastal lagoons and brackish environments. According to this law, NORs are designated to balance environmental

conservation with sustainable economic activities, such as mussel farming, ensuring that these practices do not compromise the ecological integrity of the protected areas.

1.3 Ecological and physical characteristics of semi-closed ecosystems

Sustainable management practices must consider nutrient dynamics, which are important for maintaining the ecological balance of brackish ecosystems. Nutrients play a fundamental role in shaping the biological structure of brackish ecosystems, such as coastal lagoons and brackish lakes (Jeppesen et al., 2007). The availability of nutrients, particularly nitrogen, phosphorus and carbon, regulates primary productivity and supports the growth of aquatic plants and algae, which form the base of the food chain (Guo et al., 2016). In brackish environments, nutrient dynamics are influenced by complex interactions between freshwater and marine influences. While freshwater inflows from rivers and precipitation introduce essential nutrients into these ecosystems, marine inputs also make significant contributions. This nutrient input fuels the growth of phytoplankton and macrophytes, which in turn support a variety of aquatic communities including zooplankton and benthic organisms. Seasonal changes in light, temperature, and salinity significantly influence nutrient availability and distribution within these ecosystems, shaping primary productivity and biodiversity. Moreover, the non-cyclic variations in dissolved nutrient inputs from tributaries and other sources into the lagoon influence the temporal variability observed within the ecosystem (Panigrahi et al., 2009). Previous research (Jeppesen et al., 1994) has demonstrated the profound influence of nutrient concentrations on fish biomass, zooplankton diversity, and the prevalence of keystone species in brackish lagoons.

Additionally, beyond the fundamental importance of nutrients, the role of keystone structures, keystone species and ecosystem engineers are essential in promoting diversity, complexity and heterogeneity ecosystems (Lawton & Jones, 1995; Jones et al., 1997; Tews et al., 2004; Valls et al., 2015). Specifically, marine seagrasses and algae provide a three-dimensional structure to habitats (Fig. 3), creating essential microhabitats for numerous aquatic organisms. These plants not only enhance oxygen availability and organic matter through photosynthesis but also stabilize sediments and provide shelters for benthic fauna and juvenile teleost species (Thiriet et al., 2014; Newell & Koch, 2004; Cabaço et al., 2010). Similarly, biogenic structures like bivalve mollusk shells add three-dimensional complexity to substrates such as mud, sand, and

hard bottoms, thereby enriching habitat diversity in these areas (Gutiérrez et al., 2003). Through these mechanisms, brackish environments can offer a wide range of microhabitats that support a diverse array of organisms, contributing significantly to the ecological resilience of these transitional ecosystems.

Figure 3; Portion of a Cymodocea nodosa *seagrass patch in the NOR "Laguna di Capo Peloro", with specimens of* Atherina hepsetus Linnaeus, 1758 *visible among the leaves. The seagrass bed contributes to sediment stabilization and provides shelter for numerous teleost and benthic species.*

In semi-enclosed aquatic ecosystems, connection canals are essential maintaining ecological balance. These canals facilitate the inflow of seawater, which brings fundamental nutrients and oxygen, helping to mitigate temperature variations of the lagoon. Additionally, the incoming water can redistribute muddy organic matter, significantly influencing the structure and composition of benthic communities (Magni et al., 2008). Over the course of the year, sediment accumulation can block these canals, limiting the effectiveness of water exchange and its associated benefits. Therefore, the periodic opening of these canals to remove sediments and restore water flow can be considered a beneficial intervention for the entire lagoon ecosystem (Zucchetta et al., 2021).

1.4 Teleost ecology in transitional environments: adaptations and dynamics

Teleost, which represent the largest and most diverse group of fishes, have developed a wide range of adaptive strategies that allow them to successfully colonize a variety of habitats, including coastal and brackish environments (Ravi & Venkatesh, 2008). These environments, characterized by variable salinity gradients, offer unique ecological conditions that have driven the evolution of specific physiological, behavioral, and morphological adaptations (Powers, 1980; Billard et al., 1981; D'Iglio et al., 2021). The selection of coastal and brackish environments by many teleost species is not random but results from selective pressures that have shaped their ecology and behavior to maximize reproductive success and survival. (Blaber & Blaber, 1980; Laverty & Skadhauge, 2012). Coastal and brackish environments, such as estuaries, lagoons, and river deltas, represent transitional ecosystems between marine and freshwater environments (Basset et al., 2006). The ability to tolerate wide variations in salinity, known as euryhalinity, is one of the most relevant physiological adaptations in teleost inhabiting these environments. Euryhaline species are capable of effectively regulating their osmotic balance in waters with varying salinity levels, enabling smooth transitions between marine and freshwater environments, thereby allowing them to exploit a wide range of habitats (Nordlie & Haney, 1998, Wedderburn et al., 2016). Considering the ecological advantages offered by these environments, many teleost species choose them as preferred habitats for certain phases of their life cycle. In particular, brackish environments are often used as nursery areas, where young specimens find refuge from marine predators thanks to the structural complexity of the habitat, such as seagrass beds, muddy bottoms, or mangroves and shallow waters (Whitfield, 2017). Furthermore, the nutrient richness resulting from the influx of freshwater rich in organic matter and sediment creates favorable conditions for the accumulation of primary biomass, making these habitats particularly attractive to juvenile species (Nixon et al., 1986; Heck Jr et al., 2000). Coastal lagoons, estuaries and bays constitute essential habitats for teleost, providing optimal conditions for the growth and development of their juvenile stages. These environments are characterized by separation from the intense currents and waves typical of adjacent open seas, achieved through narrow and elongated channels. This separation reduces the impact of dynamic forces (Kjerfve, 1994), creating a relatively unthreatening and protected refuge for juveniles. These natural conditions not only promote the survival and healthy growth of juvenile fish but also contribute significantly to the stability of fish populations in such environments. The shallow depths of lagoons, estuaries, and bays reduce predation, which is typically more intense in deeper environments, thereby

provide additional protection to juvenile fish (Ryer et al., 2010). Moreover, in these environments, floristic communities and phytoplankton are particularly abundant, constituting a fundamental trophic resource for small teleost (Napiórkowska-Krzebietke, 2017).

Coastal and brackish habitats act as ecological transition zones, where marine and freshwater species can coexist and interact (Garcia et al., 2003). For instance, several teleost species develop migratory life cycles that exploit the resources offered by these environments at different times of the year. Such ecological plasticity not only increases the chances of survival in changing environments but also allows these species to fully exploit the available resources at different stages of their life cycle (de Pontual et al., 2023; Chang & Iizuka, 2012; Prado et al., 2014). The ability of some species to adapt to environmental changes, coupled with the protection provided by more complex and resource-rich areas, enables their presence in these habitats despite increasing anthropogenic pressures that often threaten the integrity of these ecosystems (Bella et al., 2017; Capillo et al., 2018; Caliani et al., 2019)

The ability of teleost to exploit coastal and brackish environments as an adaptive strategy may lead to greater species richness and ecological diversification, which is crucial for the sustainability of these ecosystems and for maintaining global biodiversity (de Moura et al., 2012).

Considering the above-mentioned factors, it is clear how these transitional environments present distinct ecological conditions which make it easier to observe certain species compared to marine environments. Notably, species such as the *Aphanius fasciatus*, can be considered nearly exclusive of these habitats, highlighting the specialized nature of the fauna in these ecosystems (Maltagliati, 1999).

In semi-enclosed aquatic ecosystems, such as lagoons or coastal lakes, teleost species can be classified based on their patterns of residence and migration. For this reason, in the present thesis it has been chosen the classification proposed by Pauly and Yáñez-Arancibia, 1994, which distinguishes three main categories of teleost based on their presence in a specific area: **Sedentary Species** (SS), which complete their entire life cycle within these environments and are present throughout the year; **Seasonal Migrants** (SM), which inhabit these ecosystems during specific seasons for essential activities such as feeding, reproduction, or refuge, before leaving during other seasons; and **Occasional Visitors** (OV), whose presence is rare and generally attributable to specific ecological events or unique environmental conditions. This

classification can provide more detailed information, allowing a clearer and more specific view of the population dynamics of species residing in transitional ecosystems and a better understanding of how and under what circumstances migratory species choose to use these habitats.

1.5 Monitoring techniques and applications to these environments

Monitoring coastal and brackish ecosystems is essential for biodiversity conservation and sustainable resource management. The choice of techniques directly influences the quality of collected data and the impact on the ecological balance of these environments.

Traditional sampling methods, such as using nets or active capture techniques, are commonly used to assess teleost populations (Adao et al., 2022; Franco et al., 2012). While they provide valuable data on population size and diversity, they also have significant drawbacks. Physically captures of the specimens can be highly invasive, leading to stress, injury, or mortality not only in target species but also in non-target species (by-catch). Additionally, the use of nets can disrupt benthic habitats, altering the local communities and impacting the environment necessary for many species to thrive. (Barnette, 2001).

To mitigate the negative impacts of traditional methods, several non-invasive or minimally invasive techniques have been developed and increasingly adopted. These methods offer significant advantages in terms of reducing harm to fish populations and their habitats while providing accurate and comprehensive data.

- **Visual Census**: This technique involves divers or remotely operated vehicles (ROVs) for conduct direct observations of teleost populations. Visual census allows the identification and the count of the species in their natural habitats without physical capture. This method is particularly useful in clear water environments and can be employed repeatedly without causing disturbance (Gaeta et al., 2011; Pelletier et al., 2011).
- **Baited Remote Underwater Video Systems** (BRUVs): BRUVs utilize underwater cameras baited to attract fish into the field of view. This method captures video footage, which can then be analyzed to identify species and estimate population sizes. BRUVs are advantageous because they do not require physical contact with the species, thus minimizing stress and mortality. Additionally, they can be deployed in a range of habitats and depths, providing a comprehensive overview of teleost assemblages (Leonetti et al., 2024)

  Both the methods can be used in protected areas, as natural reserves, marine protected areas (MPAs) and similar ones thanks to their low to null impact on biota.

1.5.1 Visual census

Visual census is a well-established monitoring technique widely used today to record sightings of animal organisms in both terrestrial and aquatic environments. It is particularly prevalent for quantitative and qualitative analysis of fish communities and the observation of elusive species (Pierucci & Cózar, 2015; Yalgın & Türker, 2023; Ntouni et al., 2023; Pierucci & Suaria, 2023; Fleuré et al., 2024; Kovačić et al., 2024; Baiju et al., 2024). The origins of this method date back to the 1950s when Brock (1954) introduced the concept for teleost population surveys.

Subsequently, the use of Visual Census in the Mediterranean Sea was further developed in the 1970s, with significant contributions by Harmelin-Vivien (Harmelin-Vivien & Harmelin, 1975; Harmelin-Vivien et al., 1985) who standardized its application for marine biodiversity assessment and monitoring of fish population dynamics. This technique has established itself as a versatile tool in various scientific contexts due to its ability to provide direct and detailed data on populations of organisms. In ecological research, Visual Census is used to assess community structure (Bohnsack & Bannerot, 1986; Mayo & Jackson, 2006; Keller et al., 2024) and species distribution in various habitats (De Girolamo & Mazzoldi, 2001; Mayo, 2004; Baker & Sheaves, 2006; Brotto et al., 2007; Gomis et al., 2024; Lee et al., 2024), significantly contributing to studies on population ecology and species interactions. Furthermore, in marine biodiversity assessment, Visual Census is essential for monitoring ecosystem health, especially in MPAs, where it allows the detection of changes in the composition of teleost communities and other aquatic species, supporting the evaluation of the effectiveness of conservation measures adopted (Tunesi & Salvati, 2003; Daw et al., 2011; Prato et al., 2017; Ventura et al., 2024). According to the model proposed by Harmelin-Vivien (1985), Visual Census follows a series of fundamental steps that ensure systematic and accurate data collection on organisms present in a given area. The process begins with the delimitation of a specific study area, whose extent may vary depending on the habitat characteristics (Gilbert et al., 2005) and the research objectives. To facilitate detailed and standardized observation, this area is commonly divided into transects. Although various spatial subdivision methodologies exist (such as random movements, squares, and fixed points), the transect approach is the most commonly adopted. During field surveys, operators perform visual identification of the organisms present in the study area based on direct observation and experience (Samoilys & Carlos, 2000; Edgar et al., 2004). Each specimen observed within the transect is recorded and cataloged, noting the abundance of each species. Additionally, depending on the ecological context and the study

objectives, supplementary data can be collected, such as individual sizes (Grane-Feliu et al., 2019) and spatial position (Beldade & Gonçalves, 2007). Over the years, the use of transects has been progressively adapted to the specific characteristics of the habitats studied. In extensive contexts, using small transects can be ineffective, necessitating longer transects to ensure adequate spatial coverage representative of the entire survey area (Thresher & Gunn, 1986; Mapstone & Ayling, 1998). The Visual Census method, therefore, encompasses various application ways that vary according to habitat peculiarities; transects can also be configured differently to suit specific ecological variables (Sale and Sharp, 1983) and optimize the observation of target species (de la Guardia et al., 2021). Visual Census employs a range of tools and technologies tailored to the study environment and the specific census objectives. In marine contexts, underwater cameras are among the main technologies used to acquire high-resolution images and videos of fish communities (EBnEr et al., 2017; Bacheler et al., 2017; Sugimatsu et al., 2018; Wong et al., 2018; Wetz et al., 2020). Action cams, in particular, have found numerous applications in ecological research. Integrating these technologies into Visual Census allows a more comprehensive view of environmental dynamics and the habitats under study. Recent technological advancements in action cameras, such as improved waterproofing, real-time Wi-Fi communication, GPS integration, and the inclusion of sensors, are amplifying their applicability in aquatic research, enabling more detailed studies of fish community structure (Struthers et al., 2015). This approach allows for detailed and systematic review of collected data post-dive, minimizing potential errors due to operator and environmental parameters limitations. These devices have also been used to document fish behavior (Davis et al., 2017; Claassens and Hodgson, 2018) and foraging behavior (Hughes et al., 2003). Furthermore, cameras have facilitated the estimation of fish sizes (Harvey et al., 2002; Costa et al., 2006). Digital technologies, such as tablets and recognition software, play a crucial role in facilitating data collection, processing, and storage, increasing the efficiency of the census process. Additionally, these technologies allow for easy integration of data with other information sources, enhancing analysis and interpretation capabilities (Shimura et al., 2023). Visual Census is widely recognized for its non-invasive nature. This characteristic is essential to minimize stress on individuals and avoid altering their natural behavior. Furthermore, the non-invasiveness of Visual Census preserves habitat integrity, reducing anthropogenic impact on ecological communities and maintaining environmental conditions unaltered during surveys (Lipej et al., 2024; Muchanga et al., 2024; García-Salines & Sanchez-Jerez, 2024). Visual Census stands out for its relatively low cost compared to other ecological monitoring techniques, such as capture and marking or the use of advanced technologies. Being primarily

based on direct observation, it requires basic equipment and limited human resources, reducing operational expenses (Soldo & Glavičić, 2020). Visual Census is subject to limitations related to environmental conditions. In aquatic contexts, visibility can be significantly compromised by water turbidity caused by suspended particles, algae, or sediments. Such turbidity reduces the range of observation and may lead to an underestimation of organism density and diversity (Figueroa-Pico et al., 2020). Observer bias represents a limitation of Visual Census. Human errors can result from variability in observer competence (Pierucci & Suaria, 2023). To mitigate observer bias, it is essential to implement standardized observation protocols and ensure adequate training for observers, although some levels of error remain inevitable (Kilfoil J., 2020). Visual Census continues to represent a fundamental tool for ecological monitoring, with wide applicability ranging from shallow coastal waters to brackish semi-enclosed basins. Its effectiveness is closely linked to the proper application of methodologies and adaptation to specific environmental conditions, elements that determine the quality and reliability of the data collected.

1.5.2 BRUVs

Baited Remote Underwater Video systems are an advanced method for monitoring fish communities, allowing direct observation of organisms without physical capture (Brooks et al., 2011; Grimmel et al., 2020). These systems consist of underwater cameras installed on fixed frames, equipped with bait containers designed to attract fish towards the camera's field of view. The use of baits induces target species to converge on the monitored area, enabling the collection of data on fish populations while minimizing anthropogenic impact on the studied habitats.

The origins of BRUVs can be traced back to the 1950s, when underwater video systems were first introduced to study marine communities. However, the modern and standardized version of BRUVs was developed in the 2000s (Mallet & Pelletier, 2014). The BRUVs technique has been widely applied in several contexts such us coral reef ecosystems (Dorman et al., 2012; Watson et al., 2010) and marine protected areas (De Vos et al., 2014). Recent studies have contributed to refining the methodology, confirming BRUVs as a fundamental tool for monitoring coastal (Gladstone et al., 2012; Mateos-Molina et al., 2024) and deep-sea environments (Mathon et al., 2024; Motsenbocker et al., 2024), particularly in areas where invasive techniques could be damaging or impractical. Due to its capacity to gather data on elusive species (Torres et al., 2020) or those difficult to sample through conventional methods, BRUVs has become a robust and reliable technique for analyzing fish communities (Ghazilou et al., 2019).

The operation of BRUVs is based on a simple but effective structure: an underwater camera mounted on a metal chassis, accompanied by a baited arm that ends with a bait container (usually filled with minced fish or other attractants). This system is placed on the sea bottom, where the bait attracts fish species into the camera’s field of view. The video captures can be later analyzed in the laboratory, allowing species identification and population quantification.
The applications of BRUVs encompass a wide range of functions, including the monitoring of marine biodiversity, the analysis of fish community composition, and the examination of feeding behaviors (Parton et al., 2023). This technique is valuable not only for monitoring fish biodiversity but also for examining habitat utilization and migratory patterns (Sherman et al., 2020), supporting sustainable marine resource management and aquatic ecosystem conservation.

As previously mentioned BRUVs offer significant advantages due to their autonomous nature and ability to operate without operator efforts during video recording. This feature allows for continuous and long-term monitoring of fish communities without requiring the physical presence of operators on-site, thereby reducing the risk of habitat disturbance.

Additionally, BRUVs can be utilized in habitats that are difficult to access by both underwater operators and traditional methods, such as nets or diving (Cappo et al., 2006). This capability enables comprehensive data collection in areas where other sampling techniques may be limited. The adaptability of BRUVs to various environmental conditions makes them valuable tools for studying ecological relationships in transitional environments.

Despite their advantages, BRUVs have some limitations. One potential challenge is the selectivity of the bait, which may attract only certain species while excluding others, leading to biased data on fish community composition (Ghazilou et al., 2016). This selectivity can complicate the interpretation of results and may require careful consideration during study design.

Another limitation occurs in conditions of poor visibility, such as in turbid waters or during periods of high sediment suspension, where the effectiveness of BRUVs may be reduced. In such cases, adaptive strategies, such as the use of supplemental lighting or optical filters, are necessary to enhance the quality of footage and ensure more accurate data collection (Jones et al., 2019). Reduced visibility can hinder the effectiveness of BRUVs, potentially leading to an underestimation of fish abundance and diversity. The quality of the video material may also be compromised, making species identification more difficult (Esmaeili et al., 2021).

Additionally, another bias could arise from the behavior of certain species that tend to swim near the surface, avoiding the area captured by the cameras and thereby reducing their likelihood of being recorded (French et al., 2021). This can further skew the data, making it important to account for such behavioral patterns when designing studies and interpreting results.

1.6 Conservation and management

Coastal lagoons serve as vital ecosystems that support both biodiversity and the coastal fisheries industries. They provide essential spawning, feeding, and nursery grounds for commercially important fish species (Lanzoni et al., 2021). However, despite their ecological significance, coastal lagoons are subjected to increasing threats from anthropogenic activities, including urbanization, pollution, overfishing, habitat destruction, and modifications (Duck & da Silva, 2012; Zhang et al., 2020). The delicate balance of these ecosystems is further compounded by the effects of climate change, such as sea-level rise and altered precipitation patterns, which can aggravate existing problems and interrupt natural processes that preserve the environment (Pérez-Ruzafa et al., 2007). In consideration of these challenges, effective management and conservation strategies are imperative to safeguard the integrity and functionality of coastal lagoon ecosystems. This necessitates a holistic approach that integrates scientific research, stakeholder engagement, and policy intervention to mitigate anthropogenic impacts and promote sustainable resource management (Lanzoni et al., 2021).

Effective conservation and management of semi-enclosed brackish ecosystems, as Faro Lake, require continuous monitoring of environmental and biological variables. The dynamic nature of these environments, influenced by monthly and seasonal factors such as salinity, temperature, and oxygen level, has a direct impact on the fish community and the dynamics of their populations. Understanding these variations through consistent data collection is fundamental for maintaining ecological balance and preserving biodiversity in these habitats (Petersen et al., 2021).

One of the key insights emerging from the monitoring techniques employed in this study is the necessity to consider not only the richness and abundance of species but also the monthly and seasonal variations within this community. Temperature and salinity fluctuations, for instance, can influence the distribution and behavior of species, altering the community structure at different times of the year. These changes must be factored into management strategies to ensure that conservation efforts are adaptive and responsive to the natural dynamics of the ecosystem.

Furthermore, the data collected on environmental factors, such as the opening and closing of canals, influence the behavior of fish species, which may not have constant access to the lake, thereby restricting migratory movements in and out exclusively to periods when the canals are

open. The management of water flow through these canals could be a key tool for maintaining the ecological functionality of Faro Lake, ensuring adequate nutrient exchange and stability of the aquatic environment. For example, periodic opening of the canals to restore water flow can improve oxygen levels, water quality, and the health of benthic communities, consequently benefiting fish populations.

The results obtained from this study provide a valuable foundation for proposing sustainable management practices. By mapping species distributions and understanding the relationships between the fish community and environmental variables, targeted conservation measures can be designed. These may include strengthening controls by the relevant authorities, habitat restoration interventions, and the protection of nursery areas, which are essential for safeguarding critical habitats. Such measures will be vital for preserving the biodiversity of Faro Lake and ensuring the long-term sustainability of the ecosystem.

In conclusion, the integration of non-invasive monitoring techniques with a comprehensive understanding of environmental dynamics is essential for developing effective conservation and management strategies in transitional ecosystems like Faro Lake.

1.7 Current knowledge

Current knowledge about the teleost communities in Faro Lake is fragmentary and derives mainly from dated studies carried out over the years. Among these, the most significant were those of Cavaliere (1967), and Potoschi et al. (2002), which provided an overview of the fish fauna of the lake, highlighting the presence of 98 and 43 species respectively, many of which were also observed in the present study. Cavaliere and Potoschi et al., classified the species surveyed into three categories: permanent, periodic and occasional species, based on their patterns of presence and use of the lake. For the present study, the classification proposed by Pauly & Yáñez-Arancibia (1994) was adopted, which distinguishes species into sedentary, seasonal migrants and occasional visitors. In recent decades, climate change in the Mediterranean Sea has induced significant changes in aquatic ecosystems, affecting the life cycles and dynamics of species (Pörtner & Peck, 2010; Zucchetta et al., 2012). This context highlights the necessity for new research to update the existing knowledge. Several recent studies have focused on individual species that are rare or infrequent (Giacobbe et al., 2018; Spinelli et al., 2017, 2020), without providing a comprehensive and periodic overview of the fish communities in the lake. The absence of continuous and systematic monitoring could be an obstacle for the development of management and conservation strategies by the competent authorities. The objective of this study is to investigate the dynamics of teleost communities in order to promote targeted interventions to preserve the biodiversity and ecological functionality of this important transitional ecosystem.

1.8 Purpose and objectives

The principal aim of this doctoral research is to investigate the dynamics of teleost community in Faro Lake, focusing on species abundance and richness. This study aims to understand the fluctuations occurring throughout the year and to identify the environmental factors influencing these variations. Given the significant importance of Faro Lake as a transitional zone between marine and lagoon environments, the present study has focused on the seasonal and periodic teleost assemblages and their variation in relation to environmental parameters and habitat characteristics. Understanding how these species respond to seasonal changes and environmental variations can provide valuable insights for the conservation and management of these ecosystems.

The specific objectives of this thesis include the adoption of an integrated approach for monitoring teleost community, using different methodologies that allow the collection of useful and comparable data. The techniques employed include Underwater Visual Census (UVC) and the Baited Remote Underwater Video system (BRUVs). These tools, used in parallel, have enabled the collection of detailed data on species presence and distribution, allowing for the comparison of the techniques utilized. One of the main objectives was to evaluate which of these methods proved to be the most efficient in the specific context of transitional environments like Faro Lake, or whether the synergistic combination of both two represents the best approach for comprehensive and accurate monitoring.

Through continuous monitoring over the course of the year, it was possible to study how several fish species inhabited Faro Lake in different periods, highlighting potential seasonal preferences and migratory behaviors. From the monitoring of species presence and abundances, the presence of a particular behavior of a sedentary species was noted and focused on; in fact, *Gobius niger* Linnaeus, 1758, that is an abundant species in Faro Lake, has been noted to have a strictly relationship with the presence of marine litter objects (mainly glass and plastic bottles). This part of the thesis evaluates which factors affect this kind of behaviour.

The final goal of this study was to provide a solid scientific basis for the sustainable management of the lake's biodiversity, contributing to the development of conservation strategies that consider the observed teleost dynamics. In this way, the research aims to fill existing knowledge gaps offering new perspectives for the management and protection of local

teleost community, preserving the delicate balances of Faro Lake within the NOR of Capo Peloro.

One of the most significant features of the area is the complex coastal geomorphology, including two brackish water systems: Faro Lake and Ganzirri Lake. These coastal areas support the life of numerous aquatic species, which either complete their entire life cycle within the two lakes or exploit their characteristics, such as excellent nutrient availability and waters with less pronounced hydrodynamics than those in the Strait, to seek refuge and/or carry out reproductive functions. Particularly, Faro Lake is connected both to the Tyrrhenian Sea and the Ionian Sea, as well as to Ganzirri Lake through artificial canals. This unique characteristic results in a remarkable abundance and diversity of fish species within the lake and along the canals. For this reason, the monitoring of the teleost assemblages of Faro Lake has been conducted.

2 MATERIALS AND METHODS

2.1 Study area

Figure 4; (a): Map of the Capo Peloro Natural Oriented Reserve, showing the main areas: "1", English canal, "2", Faro canal, "3", Margi canal, "F", Faro Lake, "G", Ganzirri Lake. (b): Detailed representation of Faro Lake, highlighting the distinction between conglomerates and pebble zone (on the left) and the area used for mussel farming (on the right).

The Faro Lake (38°16'07"N 15°38'13"E) (Fig. 4b), located in the central Mediterranean Sea, in the north-eastern corner of Sicily, is a coastal brackish lake of hemispherical irregular shape, with a diameter of approximately 665 m, connected to the Tyrrhenian Sea and the Ionian Sea through two canals (Fig. 4a): the "English Canal" which flows into the Tyrrhenian Sea and it is periodically opened to ensure the exchange of surface waters with the sea, and the "Faro Canal" that permanently flows into the Ionian Sea (Ferrarin et al., 2013). The Margi Canal, connects the Faro Lake to the Ganzirri Lake (Capillo et al., 2018). The Faro Lake in its vertical profile has a funnel shape with a maximum depth of around 29 m (Bottari & Carveni, 2009). Particularly interesting is the internal geomorphology of the basin; pebbles and conglomerates are present in the northern and eastern part, while the southern and especially the eastern part are characterized by fine sediment. Furthermore, in the eastern part there is a reed thicket separating the lake from the road. In the eastern half of the basin, the depth changes rapidly due to the steep gradient of the bottom. The upper part of the water column of the deeper area is used for mussel relaying and growth by the local bivalve distribution plants, present in the area (Fig. 4b).

The lake has a vertical stratification, with the chemocline located around 12 to 13 meters depth. At this depth, oxygen levels drop sharply, while going further down, there's a gradual increase

in H_2S, caused by bacteria breaking down organic matter (Sorokin & Donato, 1975). These conditions do not allow fauna to live below this layer and characterize some parts of the Faro Lake as a biodiversity hotspot. Over the years, several authors have shown interest in this condition for knowledge and conservation purposes (Cavaliere, 1967; Potoschi et al., 2002) but knowledge remains fragmented and outdated to date.

The Faro Lake is included within the Natural Oriented Reserve (NOR) "Laguna di Capo Peloro", established by the Sicilian Region with the Decree of June 21, 2001, pursuant to Article 4 of Regional Law No. 14/88 (Regione Siciliana, 2001). The reserve encompasses a large area that includes Faro Lake, Ganzirri Lake including the canals connected to the sea. The primary objective of the reserve is to conserve transitional ecosystems, coastal habitats, and wildlife.

The Habitat 1150 “Coastal Lagoons” is recognized as a Site of Community Interest (SCI) under the (Habitat Directive 92/43/CEE) and as a Special Protection Area (SPA) under the (Birds Directive 79/409/CEE). It includes transitional environments, characterized by brackish waters and a dynamic equilibrium with the marine environment. Faro Lake is included in the Natura 2000 Network highlighting the importance for biodiversity conservation, both for endemic and for migratory species. Coastal lagoons are essential for the life cycle of numerous fish species, which utilize these areas as breeding, growth, and feeding zones. Furthermore, Faro Lake and the entire reserve, serve as a significant stopover site for over 180 species of avifauna, many of which are listed in Annex I of the Birds Directive. The reserve also hosts a variety of psammophilous and halophilous plant species, typical of coastal environments.

The management of the reserve is entrusted to the Metropolitan City of Messina, through the Office for Management Plans and Protection of Protected Areas, which is obligated to draft management plans aimed at ensuring the conservation of the habitats and species present. Among the interventions planned are the regulation of anthropogenic activities, the maintenance of the lake's hydrological regime, the prevention of wildfires, and the protection of biodiversity. The reserve is also subject of continuous monitoring projects of environmental conditions and species, to ensure sustainable management of the protected area.

(https://www.mase.gov.it/pagina/rete-natura-2000)
(https://orbs.regione.sicilia.it/images/natura2000/misureconservazione/Misure_di_Conservazione_ZSC_ITA030008_-_Format_sintetico.pdf)

Considering the status of the reserve, the present study it has been decided to use two different non-invasive monitoring techniques. This made it possible to compare the results between the techniques, highlighting the respective advantages and disadvantages. The techniques applied were Visual Census, BRUVs.

2.2 Instruments and Applied Techniques

2.2.1 Monitoring of environmental parameters

HANNA Instrument HI9829 multi-parameter probe was used to measure the chemical and physical parameters of the water. Surveys were carried out at nine stations strategically selected to represent different areas of the lake, including shallow and deeper water areas, both in areas distant from the canals connecting with the sea and in the immediate proximity. This approach allowed averages to be calculated and any significant differences between areas to be assessed. The parameters measured included temperature, pH, dissolved oxygen, salinity. Measurements were carried out following the periodicity of censuses, in fact periodic monitoring is essential to capture temporal variations in environmental parameters and teleost populations (de Andrade-Tubino et al., 2020; Machado et al., 2021). In order to avoid alterations in habitat conditions, such as sediment resuspension and anoxic sediment uplift due to trampling, the surveys were carried out before the application of monitoring techniques. The probe calibration was checked before each measurement session, using the standard solutions provided by the manufacturer for each chemical and physical parameter measured. Measurements were carried out with the maximum care to minimise turbulence and interference in the readings. At the end of each measurement session, the collected data was transferred to the University of Messina for processing. The processing of the results was performed using the dedicated software provided by the probe manufacturer, which allowed the visualisation and interpretation of the measured parameters.

2.2.2 Visual census

After several preliminary investigations, selected transects (Fig. 7) were chosen for the Faro Lake teleost monitoring study to cover the different types of environments occurring in the area, including muddy bottoms, pebble bottoms, shallow perimeter waters of the Lake, and deep waters. In addition, to improve the accuracy of the data, a perimeter survey was conducted, adapting transects to follow the irregular shape of the study area. This approach allowed a more comprehensive assessment of the fish community. Two techniques were applied: the linear transect technique (Fig. 5), applied for shallow water environments, and the circular transect technique, performed from the surface to the deepest points. The exact location

of the transects was established by GPS geographic coordinates. Furthermore, perimetral oriented transects (Fig. 7) were carried out considering the ecological traits of the target species and fish movement, method already used by (Watson & Quinn Ii, 1997). Monitoring was carried out biweekly, with one monitoring conducted in the first half of the month and the next in the second half of the month, covering an annual cycle, starting from the beginning of October 2022 and ending at the end of September 2023. This made it possible to carry out 24 monitoring sessions.

For the execution of Visual Census with linear transect, 5 transects of 300 meters each spaced 50 meters apart were chosen (Fig. 7). For each transect, the starting and ending point was chosen through geographic coordinates and the use of floating buoys anchored to the bottom. Visual census was conducted at a constant speed of about 15 m/min traveling in a straight line, covering an observation width of 2.5 meters on each side. A back-up camera was used to avoid double counting, to follow the fish throughout the transect and thus having the ability to discard any false replicates in accordance with the literature (Wartenberg & Booth, 2015).

The circular transect technique was conducted in 5 locations (Fig. 7), where full 360-degree observations were carried out, using floating buoys as fixed reference points.

Censuses were always conducted at the same moment of the day starting from 09:00). Video support allowed individuals to be counted multiple times and enabled more accurate identification of species. It was chosen to count individuals three times to minimize errors. An underwater slate was used during preliminary analysis, but the conspicuous abundance of individuals made image capture more suitable. During the transect survey individuals were videotaped to remotely annotate species abundance. Equipment consisting of a wetsuit, gloves, socks, belt with sinkers, fins, mask and snorkel was used.

A “GoPro Hero 4 Black Edition” camera was used for video documentation, using a video resolution of 1080p, 120 FPS, “Ultra-Wide” camera angle (FOV) and a screen resolution of 1920×1080, 16:9. The geographic coordinates of the start and end points of each transect were recorded by GPS GARMIN GPS MAP 64st and then plotted on a map made with QGIS 3.22 Białowieża.

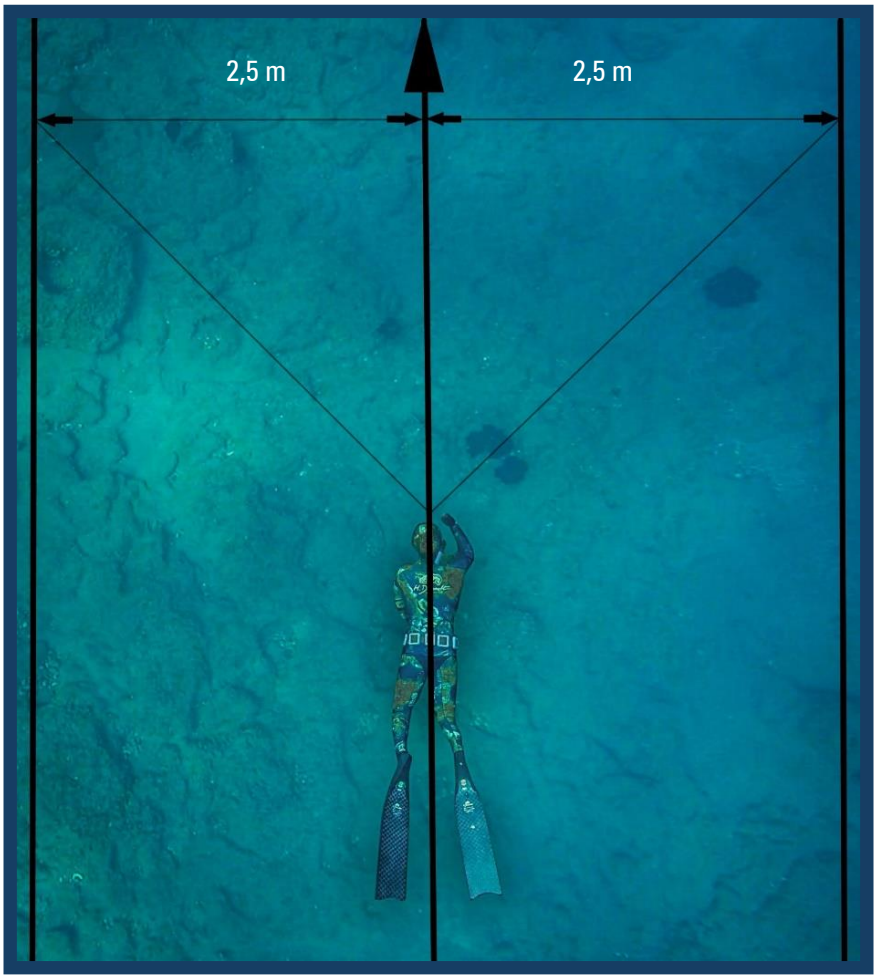

Figure 5; Technical representation of the linear transect VC method.

2.2.3 BRUVs

After preliminary analysis, considering variables such as wave motion, variable depth, turbidity, and sediment stability, it was possible to choose instrument placement stations (Fig. 7) for the application of the technique, in accordance with a recent study (Gomes et al., 2024). The monitoring tool (Fig. 6) was placed in three different stations of the lake (Fig. 7), selected according to the direction of the current (related to the movements of water masses in the Strait of Messina), to minimize the dispersion of the bait along the canals that flow into the sea, thus mitigating one of the limitations of the technique. In fact, it has been observed (Whitmarsh et al., 2018) that fishes tend to swim toward the BRUVs when they are in the opposite current and perceive the smell of the bait heading toward them. Therefore, also related to preliminary tests and field observations, sites characterized by higher abundance of species and individuals were chosen.

The instrument was made with a metal structure consisting of a heavy base coated to prevent corrosion, and an aluminum frame to provide strength and lightness for movement. The structure measured 80 cm in length, 70 cm in width and 110 cm in height. The system had a bait basket placed on the support bar and a bracket at the top for positioning the camera. Preliminary analysis showed that some baits exclusively attracted certain species. As previously noted in the literature (Harvey et al., 2007; Jones et al., 2020), the type of bait can influence the abundance of individuals and diversity. At the same time, it was observed that there is no one bait better than the others. For these circumstances, in order to avoid species selectivity at the monitoring stations and maximize the number of species, it was decided to use a bait mix placed inside the basket. One of the factors that may affect the information obtained from the BRUVs technique is the duration of execution, for this reason, in accordance with the literature (Unsworth et al., 2014) a video duration of 2 hours was chosen for each monitoring. All the measures were taken to ensure the data collected is as complete and accurate as possible.

Monitoring was carried out placing a bait inside the basket. Subsequently, the camera was placed on the bracket and recording was started. The camera was positioned to cover a camera angle of 5 meters. Following this, the operator moved away from the monitoring area to avoid influencing the behavior of the fish. After a two-hours recording period, the camera was picked up and transported to the laboratory for video analysis.

Each species was remotely annotated, marking the “first appearance time”. The first appearance time indicates the time interval from the placement of the bait until the first specimen of the species is visible in the videos. This approach allows for the observation of behavioral characteristics, as already applied by (Currey-Randall et al., 2020; Gore et al., 2020; Kilfoil et al., 2024; Whitmarsh et al., 2017). It was not possible to count the abundance of individuals, as specimens could repeatedly enter and leave the camera's field of view. The same camera used during the application of the visual census technique was used to record the videos.

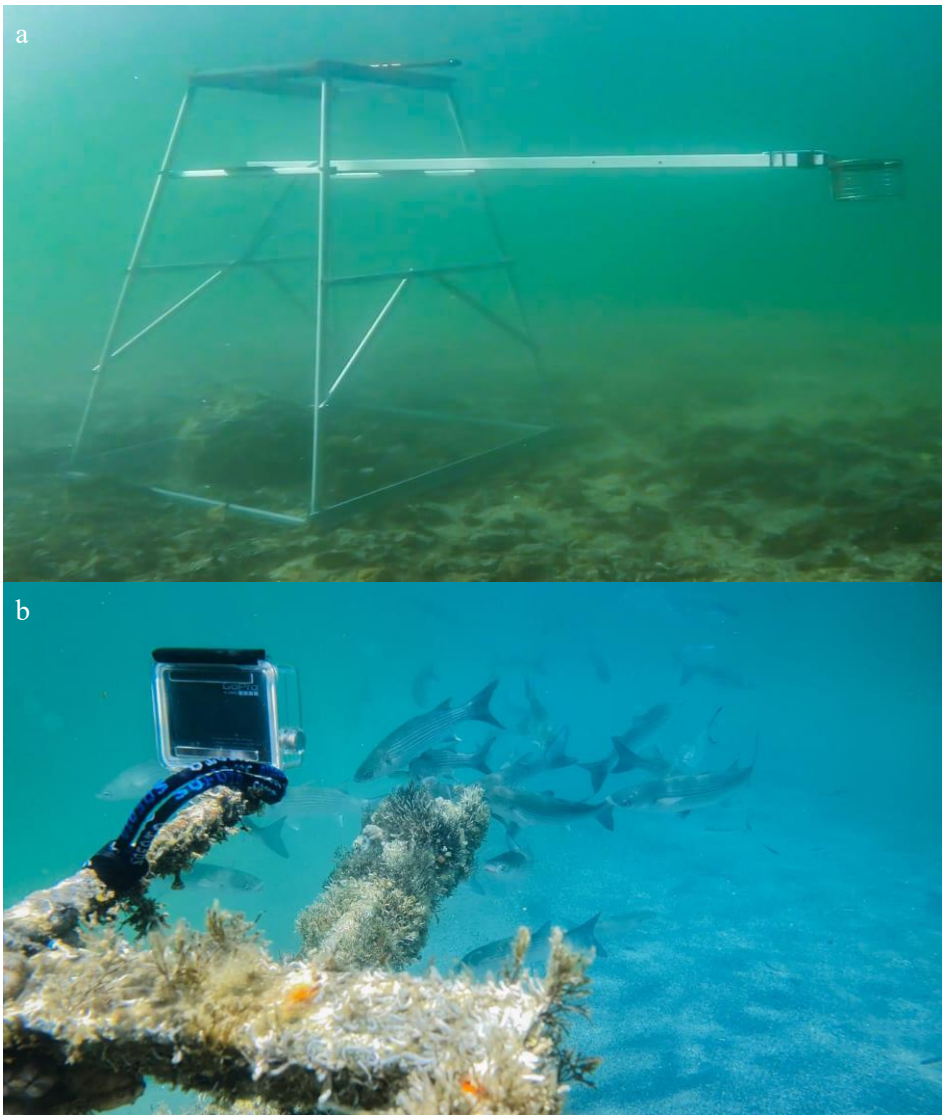

Figure 6; (a): Image of the BRUVs instrument used for monitoring; (b): Recording of teleost using the BRUVs technique.

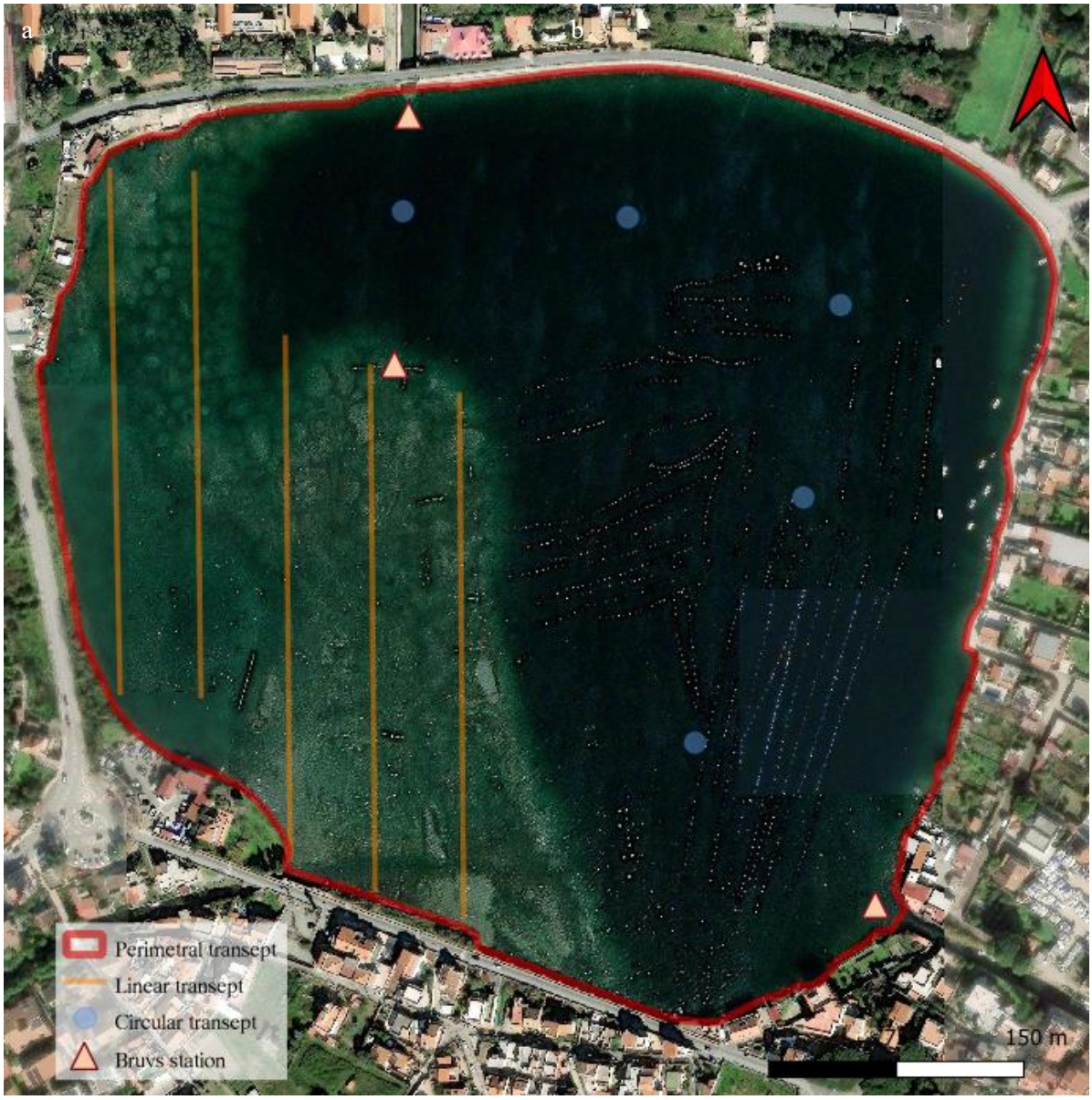

Figure 7; Graphical representation of transects traveled for obtaining data related to the visual census and BRUVs techniques. Orange line for linear transects, blue circles for circular transects, red for perimeter transect and pink triangles for BRUVs stations.

2.3 Gobius niger interaction with seafloor litter

Field observation and sampling were conducted using underwater visual census (UVCs) techniques between April and July 2024 covering the whole coastline perimeter of Faro Lake. UVCs were done through 250 m transects with a buffer of 2.5 m, reaching a maximum depth of 6 m. The sampling areas have been identified after a preliminary survey on abundance and distribution of black goby, choosing the area with a higher presence of the species. Subsequently, the shoreline zone was divided into four sites based on the characteristics of different level of anthropogenic pressures present that could be affect abundance and distribution of seafloor litter. Site 1 and site 3 were located along a street and shallow canal linking Strait of Messina and Faro Lake respectively. Instead, site 2 was positioned across boat mooring, cemented bank and close to anthropogenic activities (restaurants), while site 4 along reed cane habitat. In all sampling transects empty or inhabit ML were observed and photographed using an underwater action camera (GoPro Hero 4 Black edition) without altering its position and orientation. The abundance, size, material, color and fouling degree by macro-benthic organisms and vegetal taxa of each marine item were collected. When individuals were observed inside seafloor litter, attempts were made to photograph the species avoiding excessive stress. During sampling activities some individuals of the different species were caught, identified by peculiar characters of this species and afterwards released immediately (Sefc et al., 2020). In the same date of UVCs, water parameters were collected through multiparametric probe in four different stations selected as a part of sampling transect.

2.4 Data analysis

Data analysis was performed by using a combination of univariate and multivariate statistical methods.

Regarding environmental parameters, One-way ANOVA analysis was performed to assess temporal significant differences.

The teleost community structure was analyzed by considering number of taxa (S), fish abundance (N) (individuals/m^2), Pielou's evenness index (J′) and Shannon-Wiener's species diversity index (H loge'), Simpson index (1-Lambda'). These biological parameters were calculated for each station and season by using PRIMER-e V6. The seasonal differences of indices were tested through one-way ANOVA, performing a post-hoc Tukey's Honestly Significant Difference (HSD) test when significant differences were found ($p < 0.05$) (Prism V.8.2.1. Graph-pad Software Ltd., La Jolla, CA 92037, USA).

For the statistical analysis, the significance of the results was determined on the basis of p-value and R-squared (R^2). A p-value lower than 0.05 was considered statistically significant, indicating a strong probability that the observed results were not due to random chance. Furthermore, R-squared values were used to assess the adequacy of model fit, with higher R^2 values (0 to 1) suggesting a stronger relationship between the variables under investigation.

Information on fish community was extracted through multivariate data analysis. A square root transformation was applied to the abundance matrix, then the Bray-Curtis similarities were calculated. The dendrogram was created by means of the average linkage clustering method.

Non-parametric multi-dimensional scaling (nMDS) ordination was applied to the abundance matrix in order to detect temporal changes in the structure of fish community. The SIMilarity PERcentages (SIMPER) routine was used to establish which species contributed most to the observed differences in the data. Environmental variables which best correlated with the multivariate pattern of the fish community were identified by means of BIO-ENV procedure. All multivariate statistical analyses were performed by using PRIMER6-E.

On the interaction between *G. niger* and seafloor litter the following analyses have been performed.

To investigate the fouling degree of ML, coverage rate has been used. The coverage rate was ranked on a scale (expressed in %) ranging between zero (no coverage) and five (coverage

between 81% and 100%). Therefore, one corresponded to a coverage between 1 and 20%, two between 21 and 40%, three between 41 and 60% and four between 61 and 80% (Gallitelli et al., 2023). The coverage rate was not assessed in case seafloor litter was made with opaque material which does not allow light penetration inside the item. Additionally, inhabit rate (%IR) as a proportion between the number of ML occupied by fish and total number of ML was estimated for each site.

All statistical analyses were performed using R version 4.1.3 (Team R 2022a) in RStudio and R package “ggplot2” were used to create statistical figures. Generalized linear model GLM and generalized linear mixed models GLMMs (R function “glmmTMB” in package “glmmTMB”) were run to investigate a possible difference between empty and inhabit seafloor litter in the four different sites and to understand if individuals were attracted by a specific material, color or coverage rate of marine items collected. Site, material, color and coverage rate were included as fixed effect. All tests were followed by a post hoc test (Tukey's test) to reveal significative difference among sites. We set significant p-value levels to $p < 0.05$.

3 RESULTS

3.1 Results overview

The results of this study are presented in a structured approach, starting from the analysis of environmental data, which provides an overview of the physico-chemical conditions of the studied ecosystem. This is followed by the analysis of the teleost community, using univariate and multivariate statistical approaches to describe the community composition and its variations. The focus then shifts to teleost assemblages, with distribution maps generated through Visual Census showing the spatial patterns observed. A section is devoted to a comparison with the results obtained through BRUVs, assessing the contribution of each method to species detection. Finally, an ecological perspective is introduced, highlighting the added value provided by BRUVs in exploring specific aspects related to the ecology of the community studied.

3.2 Environmental data analysis

3.2.1 Temperature

The water temperature of the Faro Lake ranged from a minimum of 12.5 °C (27 January 2023) to a maximum 29.1 °C (20 July 2023 – 10 August 2023). Temperature values exhibited a typical annual variation, characterized by an increase during the spring and summer months and a decrease during the autumn and winter months (Tab. 1; Fig. 8). Data analysis revealed moderate significant temperature variation between autumn and winter ($p = 0.0166$), with highly significant differences observed between winter and spring ($p = 0.0072$) as well as winter and summer ($p = 0.001$) (Tab. 2; Fig. 9).

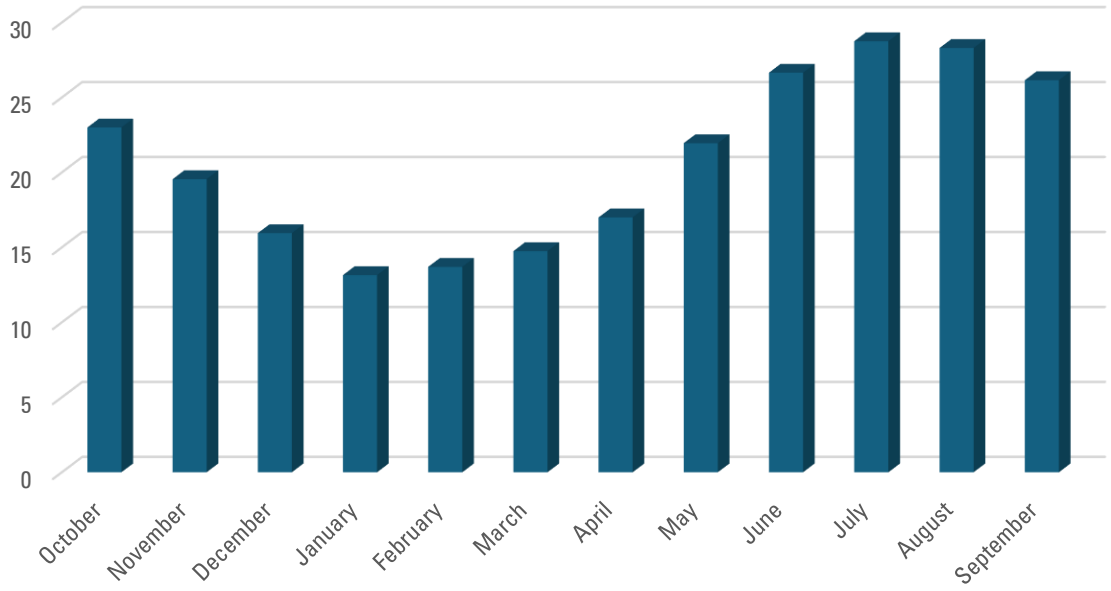

Figure 8; Annual Trends in Water Temperature of Faro Lake from October 2022 to September 2023.

Table 1; Descriptive statistic of water temperature data collected seasonally in the Faro Lake.

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	19.55	13.15	14.75	26.65
Maximum	26.15	15.95	21.95	28.75
Range	6.6	2.8	7.2	2.1
Mean	22.9	14.27	17.9	27.9
Std. Deviation	3.301	1.484	3.683	1.106
Std. Error of Mean	1.906	0.8565	2.127	0.6384
Lower 95% CI of mean	14.7	10.58	8.75	25.15
Upper 95% CI of mean	31.1	17.95	27.05	30.65

Table 2; ANOVA conducted on seasonal water temperature data collected in the Faro Lake.

ANOVA summary	
F	15.19
P value	0.0011
P value summary	**
Significant diff. among means (P < 0.05)?	Yes
R squared	0.8507

Tukey's multiple comparisons test	Mean Diff,	95,00% CI of diff,	Summary	Adjusted P Value
Autumn vs. Winter	8.633	1.729 to 15.54	*	0.0166
Autumn vs. Spring	5	-1.904 to 11.90	ns	0.1725
Autumn vs. Summer	-5	-11.90 to 1.904	ns	0.1725
Winter vs. Spring	-3,633	-10.54 to 3.271	ns	0.3897
Winter vs. Summer	-13,63	-20.54 to -6.729	**	0.001
Spring vs. Summer	-10	-16.90 to -3.096	**	0.0072

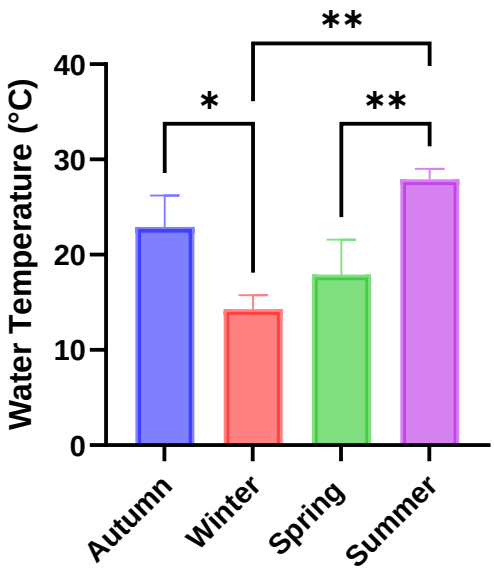

Figure 9; Graphical representation of seasonal temperature trends, specifying moderate significant variation between autumn and winter and high significance between winter and spring and winter and summer.

3.2.2 pH

pH value ranged from a minimum of 7.91 in December to a maximum of 8.20 in April maintaining a tendency towards a slight alkalinity typical of brackish waters (Fig. 10). The pH showed a stable seasonal trend, with slight increases in Spring and Summer (Tab. 3), but without showing significant changes between seasons ($p>0.05$) (Tab. 4, Fig. 11).

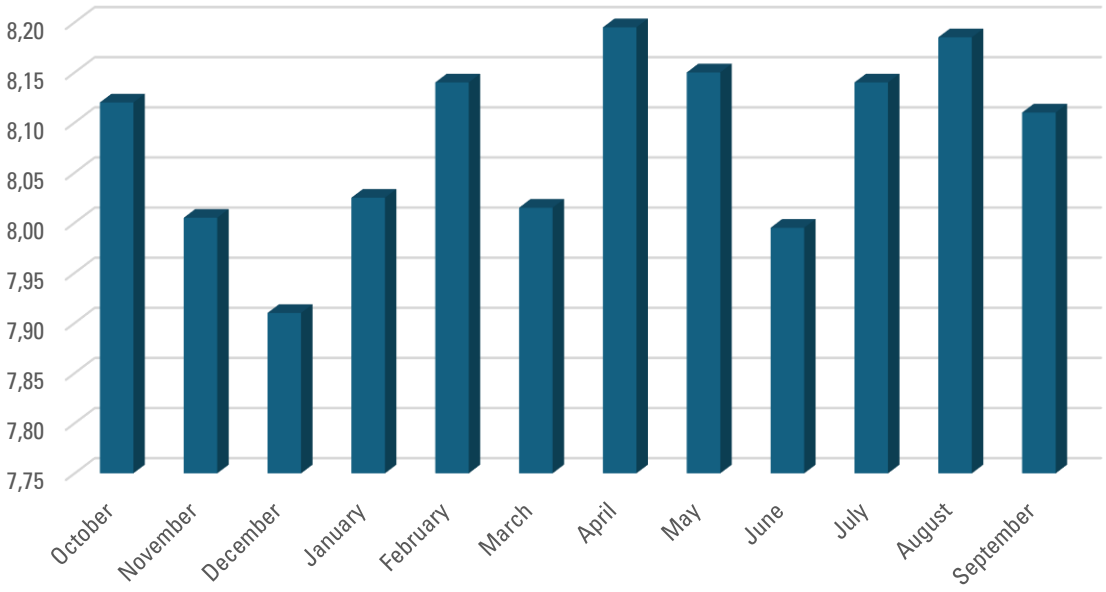

Figure 10; Annual pH Fluctuations in Faro Lake from October 2022 to September 2023.

Table 3; Descriptive statistic of pH data collected in the Faro Lake

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	8.005	7.91	8.015	7.995
Maximum	8.12	8.14	8.195	8.185
Range	0.115	0.23	0.18	0.19
Mean	8.078	8.025	8.12	8.107
Std. Deviation	0.06371	0.115	0.09367	0.09929
Std. Error of Mean	0.03678	0.0664	0.05408	0.05732
Lower 95% CI of mean	7.92	7.739	7.887	7.86
Upper 95% CI of mean	8.237	8.311	8.353	8.353

Table 4; ANOVA conducted on seasonal pH data collected in the Faro Lake.

ANOVA summary	
F	0.5918
P value	0.6376
P value summary	ns
Significant diff. among means (P < 0.05)?	No
R squared	0.1816

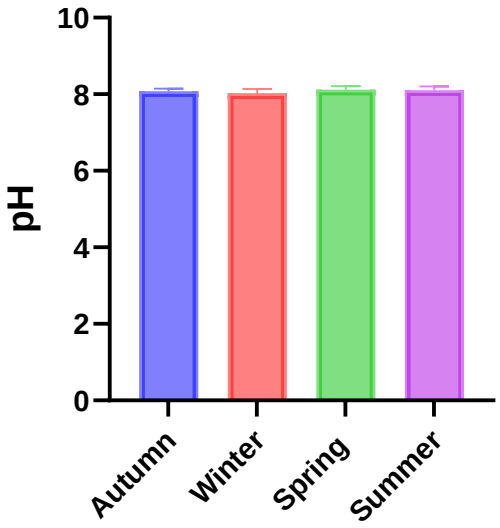

Figure 11; Graphical representation of seasonal pH trends showing no significant variations between seasons.

3.2.3 Salinity

Salinity varied throughout the year, with values ranging from a minimum of 30.25 PSU in May to a maximum of 36.25 PSU in August. During the winter months, salinity remained relatively stable, with values around 31.25 - 33.75 PSU. From the spring, a slight increase was observed, reaching the highest peaks in the summer months, particularly between July and September (Fig. 12). Salinity showed a mean oscillation between 32.15 during Winter and 34.87 during Summer (Tab. 5), without varying statistically significantly between seasons ($p>0.05$) (Tab. 6, Fig. 13).

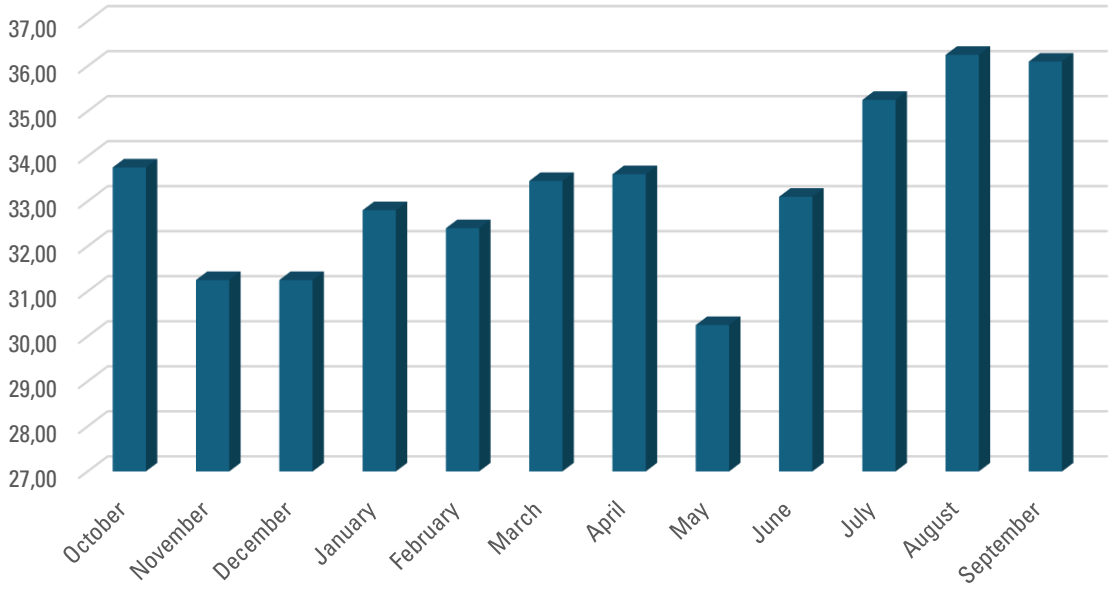

Figure 12; Annual salinity (PSU) variation in Faro Lake from October 2022 to September 2023.

Table 5; Descriptive statistic of salinity data collected in the Faro Lake.

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	31.25	31.25	30.25	33.1
Maximum	36.1	32.8	33.6	36.25
Range	4.85	1.55	3.35	3.15
Mean	33.7	32.15	32.43	34.87
Std. Deviation	2.425	0.8047	1.892	1.61
Std. Error of Mean	1.4	0.4646	1.093	0.9293
Lower 95% CI of mean	27.68	30.15	27.73	30.87
Upper 95% CI of mean	39.72	34.15	37.13	38.87

Table 6; ANOVA conducted on seasonal salinity data collected in the Faro Lake.

ANOVA summary	
F	1.476
P value	0.2926
P value summary	ns
Significant diff. among means (P < 0.05)?	No
R squared	0.3563

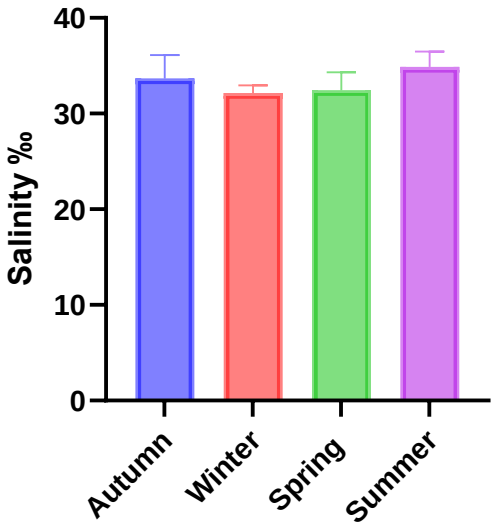

Figure 13; Graphical representation of seasonal salinity trends shows no statistically significant variation between seasons.

3.2.4 Dissolved oxygen

Dissolved oxygen levels showed a typical trend variation ranging from a minimum of 4.53 mg/L in July to a maximum of 6.71 mg/L in March. During the autumn and winter months, values gradually increased, reaching a spring peak in March. Dissolved oxygen levels decreased during the summer months, with the lowest values between June and August, and then showed a slight increase in September (Fig. 14). The variation of oxygen values followed a seasonal pattern, with a mean of 6.27 during spring and 4.7 during summer (Tab. 7). Significant variations in oxygen were highlighted between Autumn vs Spring ($p = 0.02$), Winter vs Summer ($p = 0.03$) and Spring vs Summer ($p = 0.005$) (Tab. 8, Fig. 15).

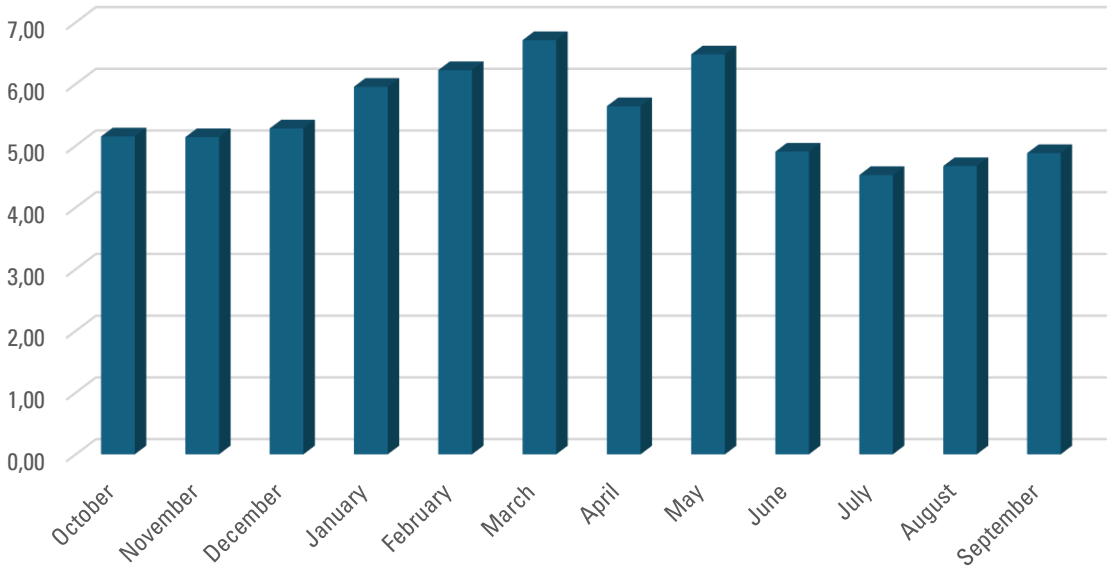

Figure 14; Annual trend of dissolved oxygen concentration (mg/L) of Faro Lake from October 2022 to September 2023.

Table 7; Descriptive statistic of oxygen data collected in the Faro Lake.

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	4.88	5.28	5.64	4.525
Maximum	5.15	6.225	6.71	4.905
Range	0.27	0.945	1.07	0.38
Mean	5.057	5.82	6.277	4.7
Std. Deviation	0.1531	0.4867	0.5632	0.1918
Std. Error of Mean	0.08838	0.281	0.3252	0.1107
Lower 95% CI of mean	4.676	4.611	4.878	4.224
Upper 95% CI of mean	5.437	7.029	7.676	5.176

Tukey's multiple comparisons test	Mean Diff,	95,00% CI of diff,	Summary	Adjusted P Value
Autumn vs. Winter	-0.7633	-1.788 to 0.2614	ns	0.1575
Autumn vs. Spring	-1.22	-2.245 to -0.1953	*	0.0215
Autumn vs. Summer	0.3567	-0.6681 to 1.381	ns	0.6913
Winter vs. Spring	-0.4567	-1.481 to 0.5681	ns	0.5183
Winter vs. Summer	1.12	0.09528 to 2.145	*	0.033
Spring vs. Summer	1.577	0.5519 to 2.601	**	0.0051

Table 8; ANOVA conducted on seasonal oxygen data collected in the Faro Lake.

ANOVA summary	
F	10.01
P value	0.0044
P value summary	**
Significant diff. among means (P < 0.05)?	Yes
R squared	0.7896

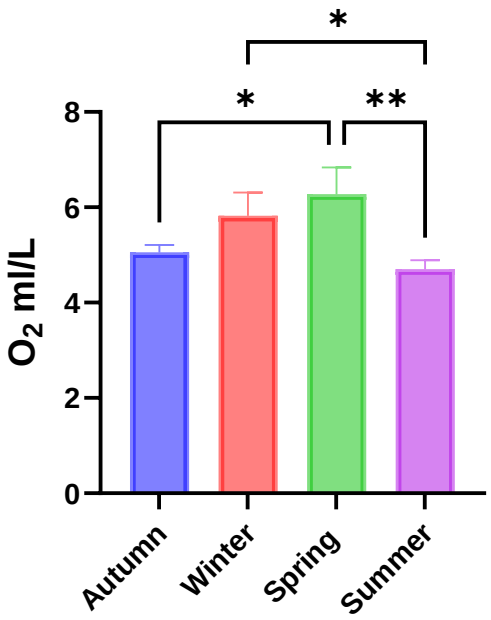

Figure 15; Graphical representation of dissolved oxigen seasonal trend showing significant variations between autumn and spring, winter and summer and highly significant differences observed between spring vs summer.

3.3 Teleost community data analysis

Seasonal trends in the ecological descriptors S (Species richness), N (Abundance), d (Margalef's index), J′ (Pielouj's index), H(LOGE) (Shannon Wiener index), 1-Lambda' (Simpson's index) are shown below.

3.3.1 Species richness

Fish diversity parameters showed a wide range of variability. The highest number of species (43) was observed during Summer, while the lowest (15) during Winter (Fig. 26a; Tab. 9), showing significant variation between Autumn vs Spring (p = 0.02), Winter vs Summer (p = 0.002) and Spring vs Summer (p = 0.04) (Tab. 10, Fig. 16).

Table 9; S data trend

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	21	15	18	30
Maximum	24	16	29	43
Mean	22.33	15.67	24	36
Std. Deviation	1.528	0.5774	5.568	6.557
Std. Error of Mean	0.8819	0.3333	3.215	3.786

Table 10; ANOVA calculated on seasonal S data

ANOVA summary	
F	11.23
P value	0.0031
P value summary	**
Significant diff. among means (P < 0.05)?	Yes
R squared	0.8081

Tukey's multiple comparisons test	**Mean Diff,**	**95,00% CI of diff,**	**Summary**	**Adjusted P Value**
Autumn vs. Winter	6.667	-4.780 to 18.11	ns	0.3135
Autumn vs. Spring	-1.667	-13.11 to 9.780	ns	0.9644
Autumn vs. Summer	-13.67	-25.11 to -2.220	*	0.0211
Winter vs. Spring	-8.333	-19.78 to 3.114	ns	0.1697
Winter vs. Summer	-20.33	-31.78 to -8.886	**	0.0021
Spring vs. Summer	-12	-23.45 to -0.5529	*	0.0402

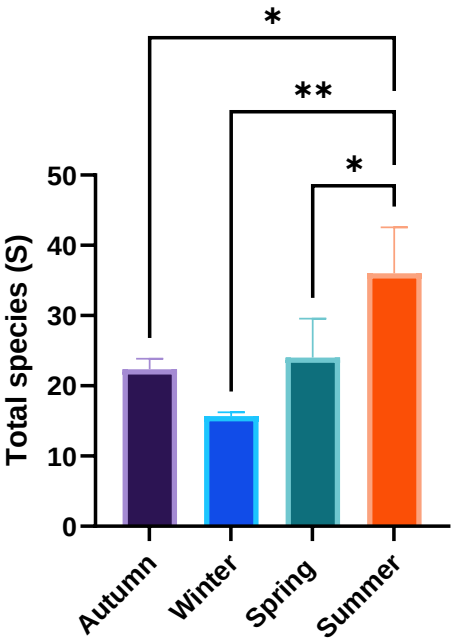

Figure 16; Graphical representation of fish diversity showing the highest number of species (43) in summer, while the lowest (15) during Winter specifying significant variation between autumn and summer, winter and summer and spring and summer.

3.3.2 Species abundance

The abundance ranged from a mean of 116.3 in Spring to 136.6 during Autumn (Tab. 11), without highlighting significant differences in the seasonal trend ($p > 0.05$) (Tab. 12, Fig. 17).

Table 11; N data trend

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	78.91	113.7	88.83	106.9
Maximum	192.1	127.7	132	159.4
Mean	136.6	119.4	116.3	127.1
Std. Deviation	56.61	7.324	23.87	28.22
Std. Error of Mean	32.69	4.228	13.78	16.29

Table 12; ANOVA calculated on seasonal N data

ANOVA summary	
F	0.2132
P value	0.8845
P value summary	Ns
Significant diff. among means (P < 0.05)?	No
R squared	0.07404

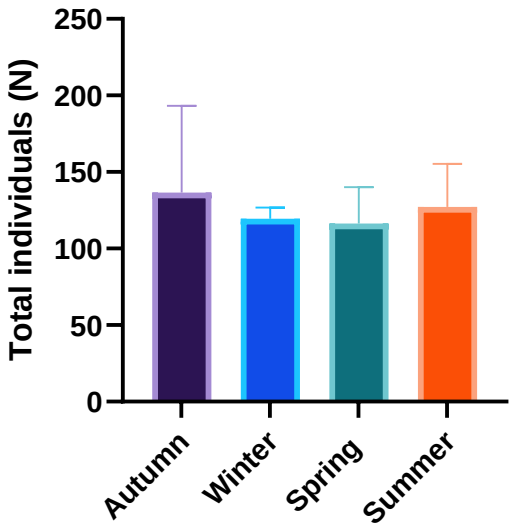

Figure 17; Graphical representation of the abundance without highlighting significant differences in the seasonal trend.

3.3.3 Margalef's index

Margalef's index ranged from 2.95 in Winter to 8.2 in Summer (Tab. 13), highlighting differences between Autumn vs Summer (p = 0.02) and Winter vs Summer (p = 0.003) (Tab. 14, Fig. 18).

Table 13; d data trend

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	3.804	2.958	3.504	6.11
Maximum	5.265	3.151	6.241	8.282
Mean	4.442	3.067	4.886	7.223
Std. Deviation	0.7481	0.09916	1.369	1.087
Std. Error of Mean	0.4319	0.05725	0.7902	0.6275

Table 14; ANOVA calculated on seasonal d data

ANOVA summary	
F	9.897
P value	0.0045
P value summary	**
Significant diff. among means (P < 0.05)?	Yes
R squared	0.7877

Tukey's multiple comparisons test	**Mean Diff,**	**95,00% CI of diff,**	**Summary**	**Adjusted P Value**
Autumn vs. Winter	1.375	-1.114 to 3.864	ns	0.3528
Autumn vs. Spring	-0.4445	-2.933 to 2.044	ns	0.9377
Autumn vs. Summer	-2.781	-5.270 to -0.2926	*	0.0296
Winter vs. Spring	-1.819	-4.308 to 0.6695	ns	0.1675
Winter vs. Summer	-4.156	-6.645 to -1.667	**	0.003
Spring vs. Summer	-2.337	-4.826 to 0.1520	ns	0.0659

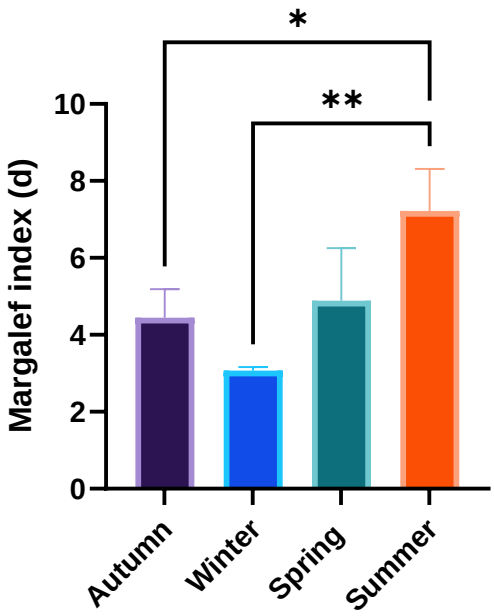

Figure 18; Graphical representation of Margalef index (d) highlighting differences between autumn and summer and winter and summer.

3.3.4 Pielou's index

J' index values ranged from a minimum of 0.23 in March to a maximum of 0.7 in September with a gradual decrease from October to March followed by a rapid increase from April to September, reaching the highest levels in the warmer months and the lowest levels in winter. (Fig. 19). Pielou's evenness index (J′) ranged from 0.695 in Summer and 0.15 during Spring (Tab. 15). No significant seasonal variation was observed ($p > 0.05$) (Tab. 16; Fig. 20).

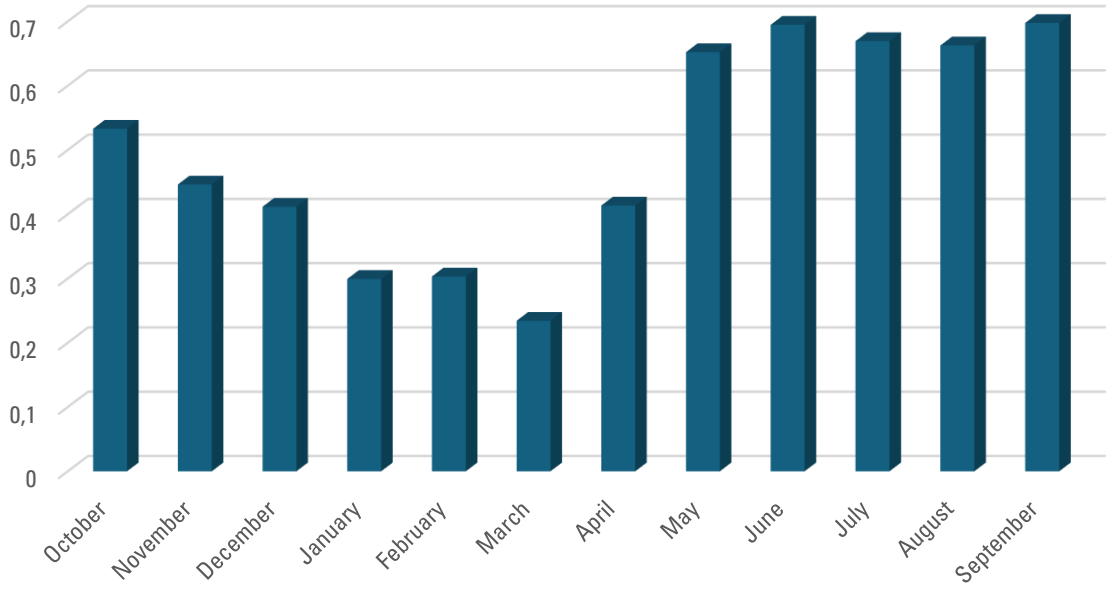

Figure 19; Annual trend of the Pielou's equitability index (J') in Faro Lake from October 2022 to September 2023.

Table 15; J' data trend

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	0.4501	0.282	0.1559	0.656
Maximum	0.6904	0.4337	0.6453	0.6955
Mean	0.5399	0.3436	0.3831	0.6693
Std. Deviation	0.1312	0.07981	0.2465	0.02269
Std. Error of Mean	0.07575	0.04608	0.1423	0.0131

Table 16; ANOVA calculated on seasonal J' data

ANOVA summary	
F	3.174
P value	0.0851
P value summary	ns
Significant diff. among means (P < 0.05)?	No
R squared	0.5434

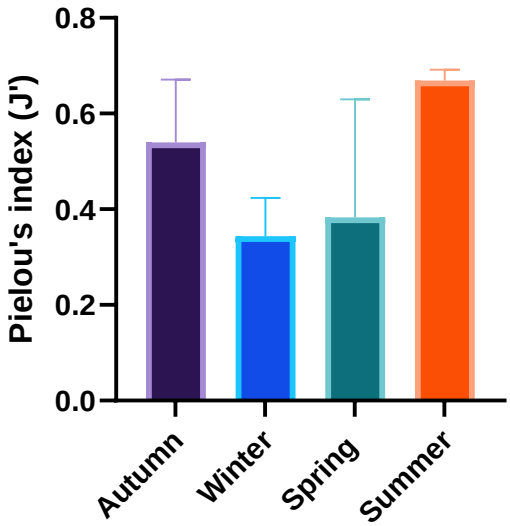

Figure 20; Graphical representation of Pielou's index (J') without significant seasonal variation.

3.3.5 Shannon-Wiener's index

H' values ranged from a minimum of 0.6 in March to a maximum of 2.46 in July. A gradual increase is observed starting in April, with a peak during the summer months (July-September) and lower values recorded in winter and early spring (Fig. 21). Shannon-Wiener's species diversity index ranged from 2.46 during Summer to 0.45 during Spring (Tab. 17), showing significant differences only between Winter and Summer ($p = 0.03$) (Tab. 18, Fig. 22).

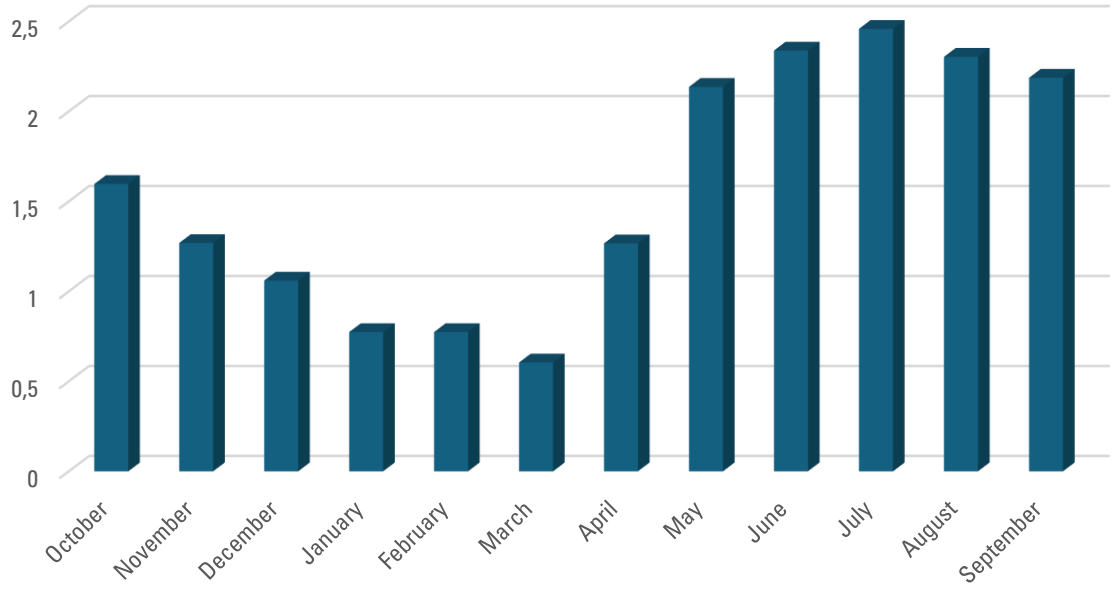

Figure 21; Annual trend of the Shannon-Wiener diversity index (H') in Faro Lake from October 2022 to September 2023.

Table 17; H (loge) data trend

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	1.37	0.7818	0.4506	2.332
Maximum	2.194	1.203	2.173	2.469
Mean	1.682	0.9458	1.248	2.389
Std. Deviation	0.4472	0.2252	0.8681	0.07112
Std. Error of Mean	0.2582	0.13	0.5012	0.04106

Table 18; ANOVA calculated on seasonal H (loge) data

ANOVA summary	
F	4.661
P value	0.0363
P value summary	*
Significant diff. among means (P < 0.05)?	Yes
R squared	0.6361

Tukey's multiple comparisons test	Mean Diff,	95,00% CI of diff,	Summary	Adjusted P Value
Autumn vs. Winter	0.736	-0.5775 to 2.049	ns	0.342
Autumn vs. Spring	0.4337	-0.8798 to 1.747	ns	0.7228
Autumn vs. Summer	-0.7071	-2.021 to 0.6064	ns	0.3724
Winter vs. Spring	-0.3023	-1.616 to 1.011	ns	0.8796
Winter vs. Summer	-1.443	-2.757 to -0.1296	*	0.0321
Spring vs. Summer	-1.141	-2.454 to 0.1727	ns	0.0906

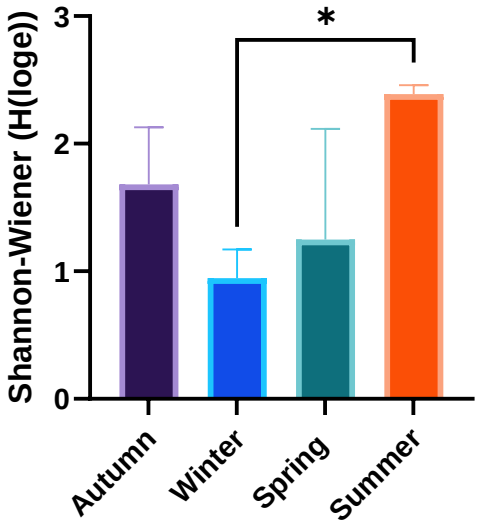

Figure 22; Graphical representation of Shannon-Wiener (H(loge)) showing a significant variation between winter and summer.

3.3.6 Simpson's index

Finally, the Simpson index ranged from 0.86 during Summer to 0.14 during Spring (Tab. 19), without showing significant seasonal variations ($p > 0.05$) (Tab. 20, Fig. 23).

Table 19; 1-Lambda' data trend

	Autumn	Winter	Spring	Summer
Number of values	3	3	3	3
Minimum	0.548	0.3089	0.147	0.854
Maximum	0.8388	0.5247	0.8111	0.8676
Mean	0.6478	0.3843	0.4571	0.8613
Std. Deviation	0.1654	0.1218	0.3342	0.006845
Std. Error of Mean	0.0955	0.0703	0.1929	0.003952

Table 20; ANOVA calculated on seasonal 1-Lambda' data

ANOVA summary	
F	3.559
P value	0.0671
P value summary	ns
Significant diff. among means (P < 0.05)?	No
R squared	0.5716

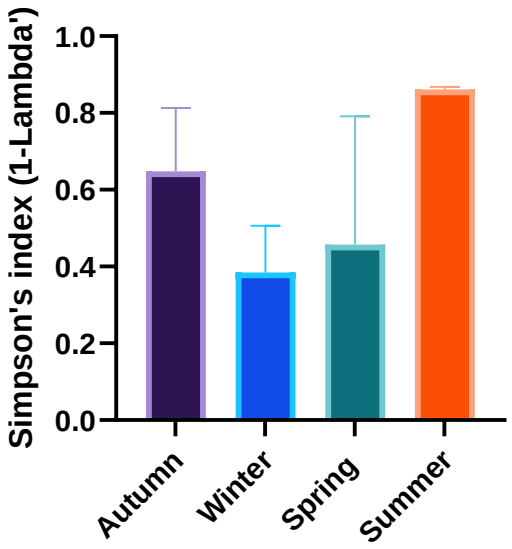

Figure 23; Graphical representation of Simpson's index (1-Lambda') showing no significant seasonal variation.

3.3.7 Multivariate analysis

The BIO-ENV procedure showed a high degree of correlation between environmental and biological data. Water temperature resulted the variable with the largest correlation coefficient, ρ= 0.854, followed by O_2 ml/L, ρ = 0.728.

Cluster analysis and MDS ordination grouped the whole set of data into four main clusters (Fig. 24).

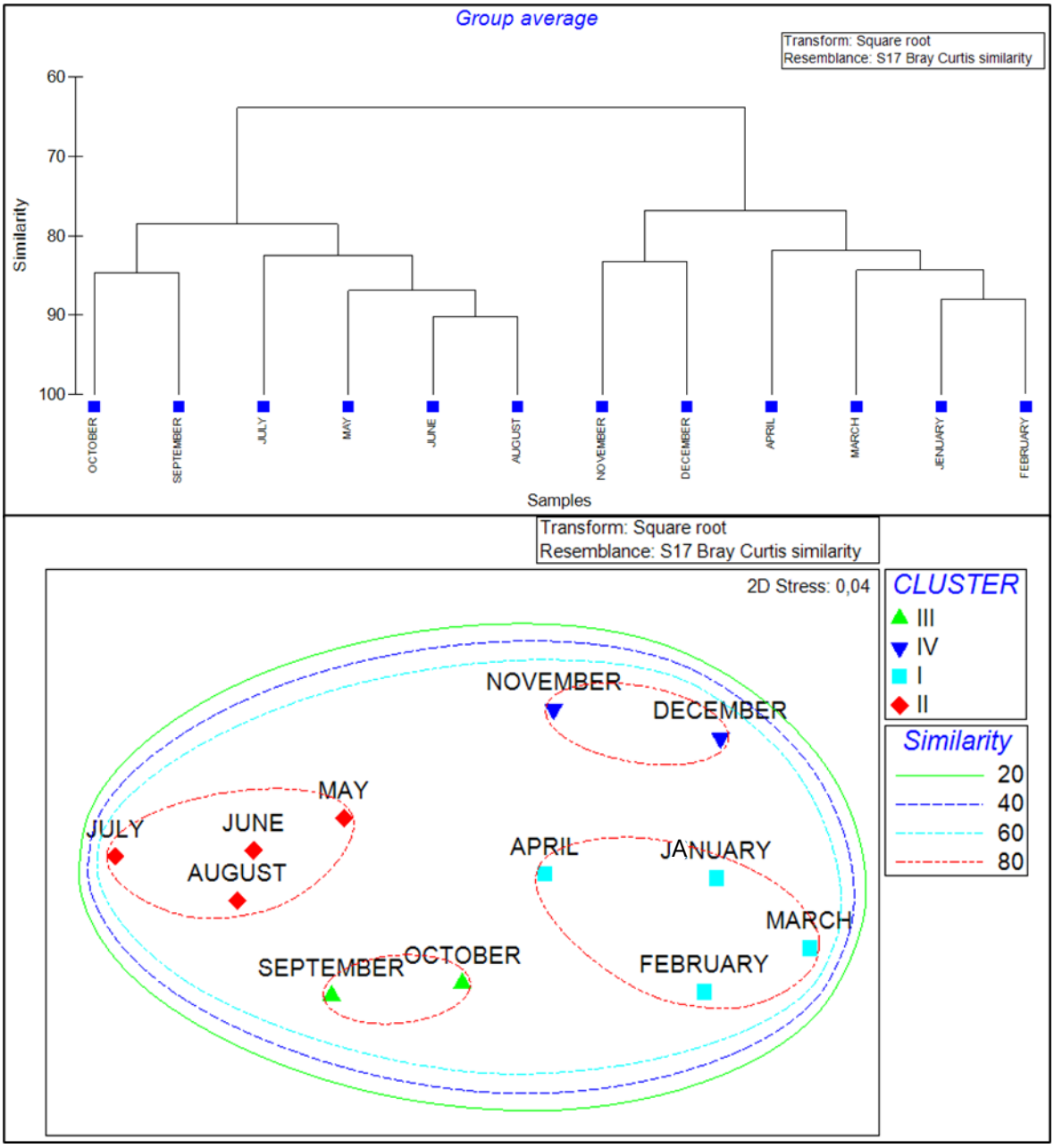

Figure 24; Dendrogram and nMDS ordination of Bray-Curtis similarities from abundance data (square root transformation) for sampling area in the Faro Lake (Messina).

Each cluster is characterized by similar abundance value and/or similar species composition. Cluster I showed the highest percentage of abundance of the species *Atherina hepsetus* Linnaeus, 1758, compared to the other clusters (from 76 to 92.39%), and a very similar % composition in species among the samples included in the cluster (Fig. 24).

Cluster II is characterized by the highest percentage of abundance of the species *Sarpa salpa* Linnaeus, 1758, compared to the other clusters (ranging from 7.92 to 24.75%), a similar percentage of abundance of the species *Diplodus sargus* Linnaeus, 1758 and high % abundance of *Gobius paganellus* Linnaeus, 1758 (4-6%), *Gobius niger* (from 6.60 to 9.67%) and *Epinephelus marginatus* Lowe, 1834 (from 0.08 to 0.13%); as well as the only cluster showing the presence of the species *Pomatoschistus minutus* Pallas, 1770 (from 0.2 to 1.35%). Furthermore, cluster II is the only one in which the presence of *Oedalechilus labeo* Cuvier, 1829, *Serranus scriba* Linnaeus, 1758, *Apogon imberbis* Linnaeus, 1758, *Scorpaena porcus* Linnaeus, 1758, *Murena helena* Linnaeus, 1758 and *Coris julis* Linnaeus, 1758 is reported (Fig. 24).

Cluster III includes samples with a similar richness in species ranged from 22 and 24 (S value) (Fig. 24).

Cluster IV showed similar abundance values for the species *Atherina hepsetus* (65.67% and 66.74% respectively, *Sarpa salpa* (about 3.30% for both samples), the highest abundance values of *Lithognathus mormyrus* Linnaeus, 1758, compared to the other Clusters (from 13.80 to 17.44 %), similar percentages of abundance of the species *Gobius niger* (2.60 and 2.26% respectively) and the same percentage of *Chromis chromis* Linnaeus, 1758 (0.41%). Cluster IV is the only one in which the presence of *Balistes capriscus* Gmelin, 1789 is reported (Fig. 24).

The outcome of SIMPER analysis, reported in Tab. 21, revealed that the most significant similarity between group members was associated with cluster II (average similarity: 85.25), followed by the cluster III (average similarity: 84.73). Tab. 21 also includes the percentage contribution of the main species to the similarity within each group.

Regarding cluster II, the species with the highest contribution was *Atherina hepsetus* with a percentage of 15.46 %. In cluster III, the highest contribution was given again by *Atherina hepsetus* with a percentage of 18.02 %. Cluster I showed an average similarity of 83.72% with *Atherina hepsetus* giving a contribution of 46.52%. Similarly for cluster IV, with an average similarity of 83.27%, the species with the highest contribution was *Atherina hepsetus* with a percentage of 32.72%.

Table 21; The most representative species that contributed to the similarity within each cluster, determined by SIMPER analysis.

CLUSTER I				
Average similarity: 83.72				
Taxa	**Av.Abund**	**Av.Sim**	**Contrib%**	**Cum.%**
Atherina hepsetus	10.11	38.95	46.52	46.52
Chelon labrosus	1.7	5.79	6.91	53.43
Sarpa salpa	1.86	5.39	6.44	59.88
Gobius niger	1.57	5.31	6.34	66.22
Parablennius sanguinolentus	1.43	5.25	6.27	72.49
Chelon auratus	1.6	5.2	6.21	78.69
Gobius paganellus	1.21	3.95	4.72	83.41
Thalassoma pavo	0.83	2.42	2.89	86.3
Gobius incognitus	0.6	2.15	2.57	88.87
Mugil cephalus	0.67	2.08	2.48	91.35
CLUSTER II				
Average similarity: 85.25				
Taxa	**Av.Abund**	**Av.Sim**	**Contrib%**	**Cum.%**
Atherina hepsetus	5.92	13.18	15.46	15.46
Chelon labrosus	3.44	7.44	8.72	24.18
Sarpa salpa	4.19	7.26	8.51	32.69
Gobius niger	2.97	6.43	7.54	40.23

Gobius paganellus	2.48	5.5	6.45	46.68
Chelon auratus	2.62	5.4	6.33	53.01
Mugil cephalus	2.56	5.11	5.99	59
Parablennius sanguinolentus	2.02	4.4	5.16	64.16
Salaria pavo	1.99	4.14	4.86	69.02
Gobius incognitus	1.42	3.01	3.53	72.55
Thalassoma pavo	1.27	2.84	3.33	75.88
Gobius cobitis	1.26	2.39	2.8	78.68
Boops boops	1.34	2.34	2.75	81.43
Lithognathus mormyrus	0.92	1.72	2.01	83.44
Pomatoschistus minutus	1.02	1.69	1.99	85.43
Chromis chromis	0.73	1.54	1.8	87.23
Coryphoblennius galerita	0.67	1.29	1.51	88.74
Trachurus *sp.*	0.56	1.08	1.26	90.01

CLUSTER III

Average similarity: 84.73

Taxa	**Av.Abund**	**Av.Sim**	**Contrib%**	**Cum.%**
Atherina hepsetus	7.39	15.27	18.02	18.02
Chelon labrosus	3.4	9.74	11.5	29.52
Mugil cephalus	2.74	7.74	9.14	38.66
Gobius niger	2.27	6.62	7.81	46.47
Gobius paganellus	1.95	5.64	6.65	53.12

Sarpa salpa	2.05	5.43	6.4	59.53
Chelon auratus	2.04	5.24	6.19	65.71
Salaria pavo	1.71	4.51	5.33	71.04
Parablennius sanguinolentus	1.52	4.26	5.03	76.07
Gobius incognitus	1.25	3.65	4.31	80.38
Gobius cobitis	1.22	3.58	4.23	84.61
Thalassoma pavo	1.33	2.97	3.5	88.12
Belone belone	0.65	1.8	2.13	90.24

CLUSTER IV

Average similarity: 83.27

Species	**Av.Abund**	**Av.Sim**	**Contrib%**	**Cum.%**
Atherina hepsetus	10.23	27.25	32.72	32.72
Lithognathus mormyrus	4.93	13.93	16.73	49.45
Sarpa salpa	2.3	6.11	7.34	56.79
Chelon labrosus	2.27	5.55	6.66	63.45
Thalassoma pavo	1.86	5.04	6.05	69.5
Gobius niger	1.97	5.01	6.02	75.51
Parablennius sanguinolentus	1.55	4.24	5.09	80.6
Gobius paganellus	1.68	4.24	5.09	85.69
Gobius cobitis	1.19	2.77	3.33	89.01
Chromis chromis	0.81	2.15	2.58	91.6

3.4 Teleost assemblages

3.4.1 Visual census

During the monitoring period, 50943 specimens belonging to 52 species were recorded.

In total, six species (*Aphanius fasciatus* Valenciennes, 1821, *Sardinella aurita* Valenciennes, 1847, *Engraulis encrasicolus* Linnaeus, 1758, *Syngnathus* sp., *Seriola dumerili* Risso, 1810, and a species belonging to the Alosidae family) were not considered for various reasons. *Aphanius fasciatus, Sardinella aurita* and *Engraulis encrasicolus* were excluded because of difficulties in estimating their numbers due to their elusive behaviour; *Syngnathus* sp. was not sampled, making identification difficult; Seriola dumerili, represented by a juvenile specimen, was excluded because it was observed following a floating object; Alosidae species was not considered due to the difficulty of the identification. *Atherina hepsetus* was the most abundant species (29929 specimens). A total of 19 family were observed. The most represented family were Sparidae with 9 species (18%), following by Gobiidae, Mugilidae and Serranidae with 5 species (10% for each one), Blenniidae, Carangidae and Labridae with 4 species (8%). The remaining 28% includes families represented by smaller numbers of species: Aphanidae, Apogonidae Atherinidae, Balistidae, Belonidae, Engraulidae, Dorosomatidae, Alosidae, Moronidae, Mullidae, Muraenidae, Pomacentridae, Scorpaenidae, Syngnathidae (Tab. 22).

The species observed in the Faro Lake have been classified into three functional ecological categories based on their behavior and annual presence patterns. These categories are as follows: Sedentary Species (SS), which complete their entire life cycle within these environments and are present throughout the year; Seasonal Migrants (SM), which inhabit these ecosystems during specific seasons for essential activities such as feeding, reproduction, or refuge, before leaving during other seasons; and Occasional Visitors (OV), whose presence is rare and generally attributable to specific ecological events or unique environmental conditions. 18 sedentary species, 25 seasonal migrants and 8 occasional visitors were observed (Tab. 22).

The abundance of individuals recorded during the monitoring period was also reported. Species whose abundances were not recorded were marked by “n.d.” (not determined).

Table 22; List of teleost species observed in the Faro Lake through visual census technique, indicating the family, their functional ecological category (FEC) and relative abundances (N) from October 2022 to September 2023.

TAXA	FAMILY	FEC	N
Aphanius fasciatus (Valenciennes, 1821)	Aphanidae	SS	n.d.
Atherina hepsetus (Linnaeus, 1758)	Atherinidae	SS	29929
Chelon labrosus (Risso, 1827)	Mugilidae	SS	3181
Mugil cephalus (Linnaeus, 1758)	Mugilidae	SS	1598
Chelon auratus (Risso, 1810)	Mugilidae	SS	1660
Sarpa salpa (Linnaeus, 1758)	Sparidae	SS	3816
Thalassoma pavo (Linnaeus, 1758)	Labridae	SS	688
Salaria pavo (Risso, 1810)	Blenniidae	SS	857
Parablennius sanguinolentus (Pallas, 1814)	Blenniidae	SS	1160
Gobius cobitis (Pallas, 1814)	Gobiidae	SS	469
Gobius paganellus (Linnaeus, 1758)	Gobiidae	SS	1510
Gobius niger (Linnaeus, 1758)	Gobiidae	SS	2178
Gobius incognitus (Kovačić & Šanda, 2016)	Gobiidae	SS	468
Epinephelus marginatus (Lowe, 1834)	Serranidae	SS	24
Epinephelus aeneus (Geoffroy Saint-Hilaire, 1817)	Serranidae	SS	5
Mycteroperca rubra (Bloch, 1793)	Serranidae	SS	16
Chromis chromis (Linnaeus, 1758)	Pomacentridae	SS	166
Syngnatus sp.	Syngnathidae	SS	n.d.
Oedalechilus labeo (Cuvier, 1829)	Mugilidae	SM	33
Chelon ramada (Risso, 1827)	Mugilidae	SM	2
Oblada melanurus (Linnaeus, 1758)	Sparidae	SM	98
Diplodus sargus (Linnaeus, 1758)	Sparidae	SM	29
Diplodus vulgaris (Geoffroy Saint-Hilaire, 1817)	Sparidae	SM	18
Diplodus annularis (Linnaeus, 1758)	Sparidae	SM	34
Diplodus puntazzo (Walbaum, 1792)	Sparidae	SM	3
Sparus aurata (Linnaeus, 1758)	Sparidae	SM	63
Boops boops (Linnaeus, 1758)	Sparidae	SM	284
Lithognathus mormyrus (Linnaeus, 1758)	Sparidae	SM	1956
Belone belone (Linnaeus, 1760)	Belonidae	SM	74
Caranx crysos (Mitchill, 1815)	Carangidae	SM	52
Caranx rhonchus (Geoffroy Saint-Hilaire, 1817)	Carangidae	SM	9
Trachurus sp.	Carangidae	SM	61
Trachinotus ovatus (Linnaeus, 1758)	Carangidae	SM	2

Symphodus ocellatus (Linnaeus, 1758)	Labridae	SM	10
Symphodus roissali (Risso, 1810)	Labridae	SM	7
Mullus barbatus (Linnaeus, 1758)	Mullidae	SM	99
Mullus surmuletus (Linnaeus, 1758)	Mullidae	SM	15
Coryphoblennius galerita (Linnaeus, 1758)	Blenniidae	SM	115
Pomatoschistus minutus (Pallas, 1770)	Gobiidae	SM	159
Epinephelus costae (Steindachner, 1878)	Serranidae	SM	52
Sardinella aurita (Valenciennes, 1847)	Dorosomatidaeidae	SM	n.d.
Engraulis encrasicolus (Linnaeus, 1758)	Engraulidae	SM	n.d.
Alosidae	Alosidae	SM	n.d.
Aidablennius sphynx (Valenciennes, 1836)	Blenniidae	OV	1
Balistes capriscus (Gmelin, 1789)	Balistidae	OV	11
Serranus scriba (Linnaeus, 1758)	Serranidae	OV	1
Apogon imberbis (Linnaeus, 1758)	Apogonidae	OV	5
Dicentrarhus labrax (Linnaeus, 1758)	Moronidae	OV	13
Scorpaena porcus (Linnaeus, 1758)	Scorpaenidae	OV	5
Murena helena (Linnaeus, 1758)	Muraenidae	OV	1
Coris julis (Linnaeus, 1758)	Labridae	OV	1

The species richness trend shows a gradual increase towards the summer months. July was the month with the highest species richness, and January the lowest (Fig. 25a). Instead, the abundance of species trend exhibited two peaks in the autumn and summer periods, and in May displayed the lowest values (Fig. 25b).

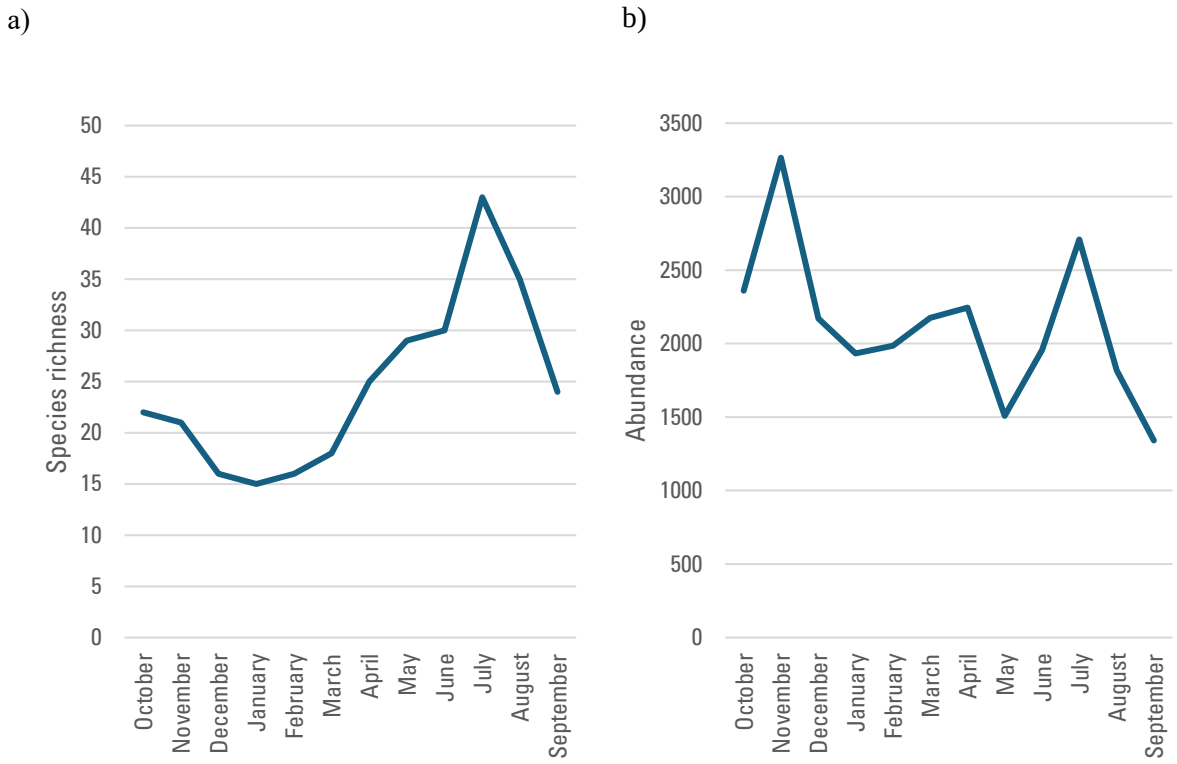

Figure 25; a) Monthly species richness and b) abundance trend from October 2022 to September 2023.

Average of the seasonal species richness and abundance shows opposite trends. The trend in species richness over time, as observed using the visual census technique, shows a decrease from autumn to winter, followed by a progressive increase from winter to summer, with the highest peak recorded during the summer months (Fig. 26a). Instead the abundance exhibited decreasing trend from Autumn to Summer with the lowest values in Spring (Fig. 26b).

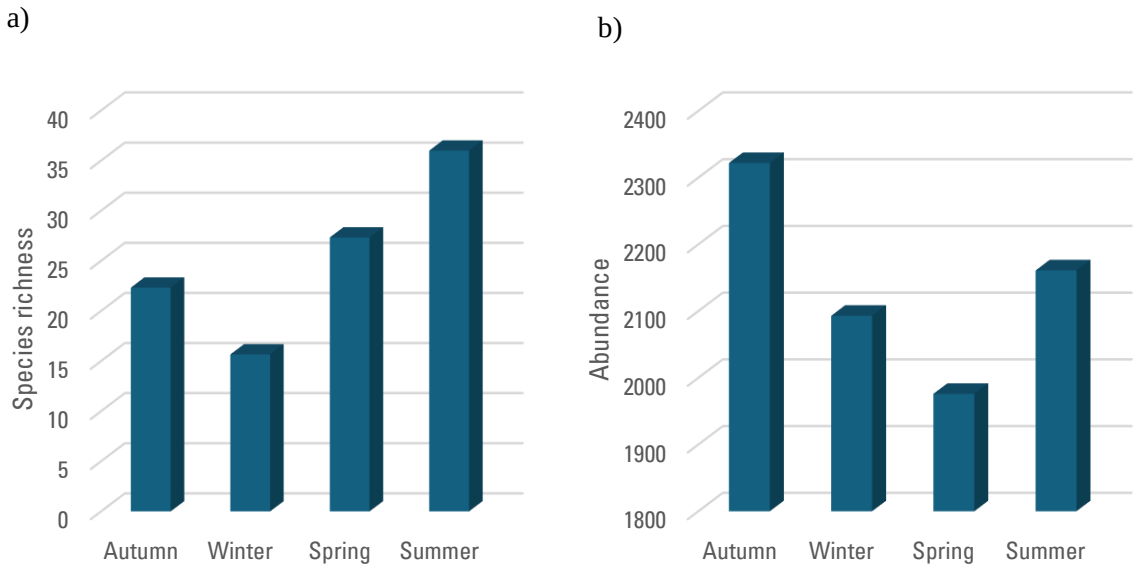

Figure 26; a) Average of the seasonal species richness and b) abundance from October 2022 to September 2023.

3.4.1.1 Distribution of sedentary species

The visual census technique allowed to identify the areas of greatest presence of sedentary species. *A. hepsetus* occupies the entire perimeter. *C. auratus* was almost always observed near the English Canal. *C. labrosus* was observed in the central part of the lake and along the eastern side, whereas *M. cephalus* was found in the slightly deeper central area and near the entrance of the Margi Canal into the lake. Despite the wide distribution of Gobiidae species in the lake, the highlighted points represent areas where these species were recorded for each monitoring execution. *G. incognitus* and *C. chromis* were mostly recorded in either side of the entrance of the English canal leading into the lake, *G. niger* was widely distributed throughout the lake but was most observed near the English Canal and in the areas in the western and eastern sides. *G. cobitis* has been observed in the northern and eastern areas, where pebbles and conglomerates are located. *G. paganellus* has been observed in the western part, most frequently between the north-western end of the lake, characterised by large pebbles and sand at the base. *P. sanguinolentus* has been most frequently recorded in the northern area, in areas with pebbles bottom. *S. pavo* has been observed more frequently near the center of the lake, in proximity to structures used for mussel farming. *A. fasciatus* has been observed at the entrance of the Margi Canal connected to the lake. *T. pavo* has been observed on the left side of the entrance of the English Canal and on the right side of the entrance of the Faro Canal. *S. salpa* has been observed in the eastern and southern part of the lake in correspondence with mussel farming activities and also near the English Canal. *M. rubra* and *E. marginatus* has been recorded in the left side of the entrance of the English Canal, *E. marginatus* was also observed in the southern part of the lake. *E. aeneus* was spotted in the slightly deeper area in front of the English Canal. *Syngnathus* sp. was observed at a bed of *Cymodocea nodosa* at the entrance to the Faro Canal. Sedentary species were almost all recorded mostly along the perimeter of the lake, with the exception of a few species of Mugilidae (*C. labrosus* and *M. cephalus*) and Blenniidae (*S. pavo*) (Fig. 27).

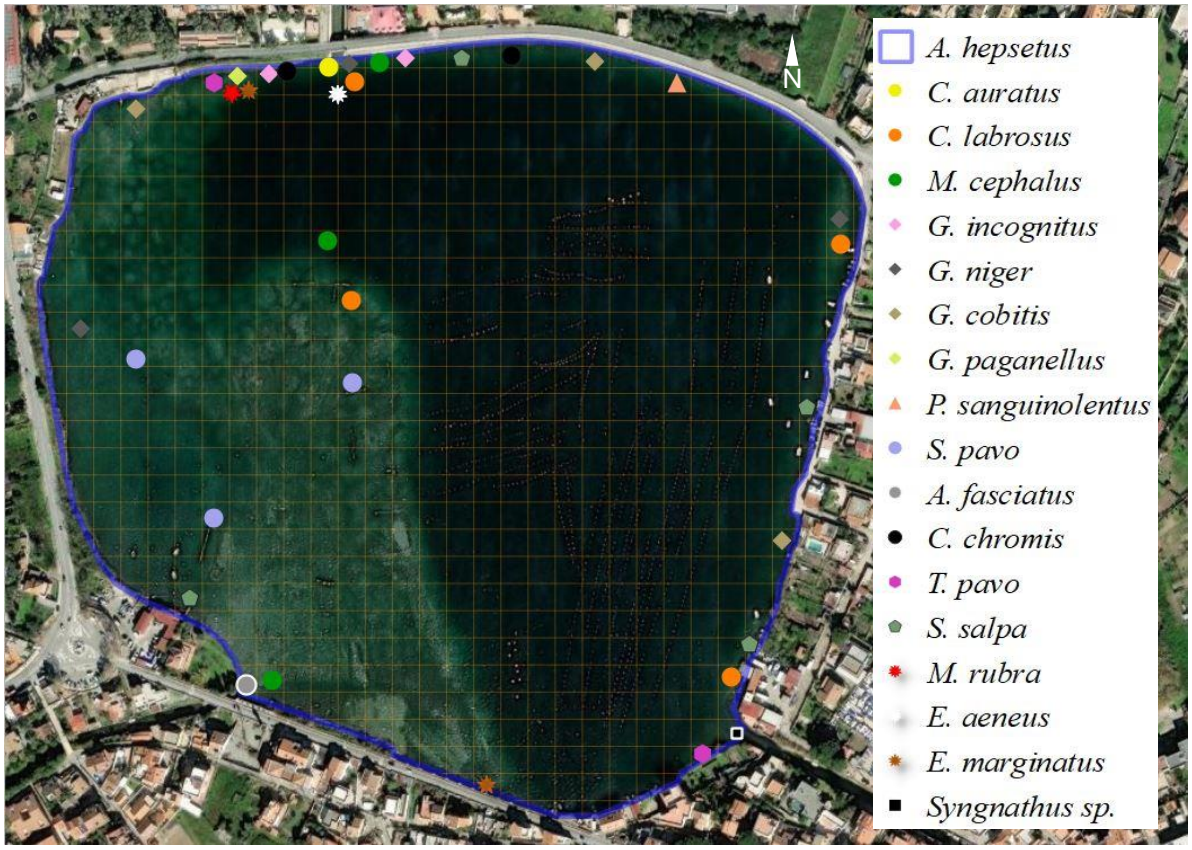

Figure 27; Sites consistently utilized throughout the year, where these species were observed with the greatest abundance and where sightings were repeatedly recorded over time. The blue perimeter line represents A. hepsetus specimens distribution. Map generated with QGIS.

3.4.1.2 Distribution of seasonal migrants

The visual census technique allowed for the identification of the areas most frequently used by seasonal migrant species. *E. encrasicolus* and *S. aurita* were observed in proximity to both the English Canal and the Faro Canal. *C. crysos, C. ramada, L. mormyrus, M. surmuletus, O. labeo, P. minutus* and *T. ovatus* were recorded in proximity to the entrance of the English Canal (*L. mormyrus* was also recorded in the southern part of the lake, near the mussels farm activity). *D. puntazzo, D. vulgaris, S. ocellatus* and *S. roissali* were observed near the entrance of the Faro Canal. *D. sargus* and *O. melanurus* were observed in proximity to both the English Canal and the Faro Canal (*O. melanurus* specifically in the right side). *B. belone* and *B. boops* were recorded in the right side of the entrance of the English Canal. *E. costae* was mainly found in a single visual census, during which several juvenile specimens were observed on the left side of the English Canal. The map shows that several seasonal migrant species have been recorded near the canals. *C. rhonchus* and *Trachurus* sp. were observed in the center of the lake, in correspondence with the beginning of the slope toward the deeper areas. *C. galerita* was observed on the mussel rows in front of the Faro Canal. *D. annularis* was observed in the center of the lake, *M. barbatus* was found in the southern part of the lake, *S. aurata*. were recorded in the center of the lake and among the mussel long line system (Fig. 28).

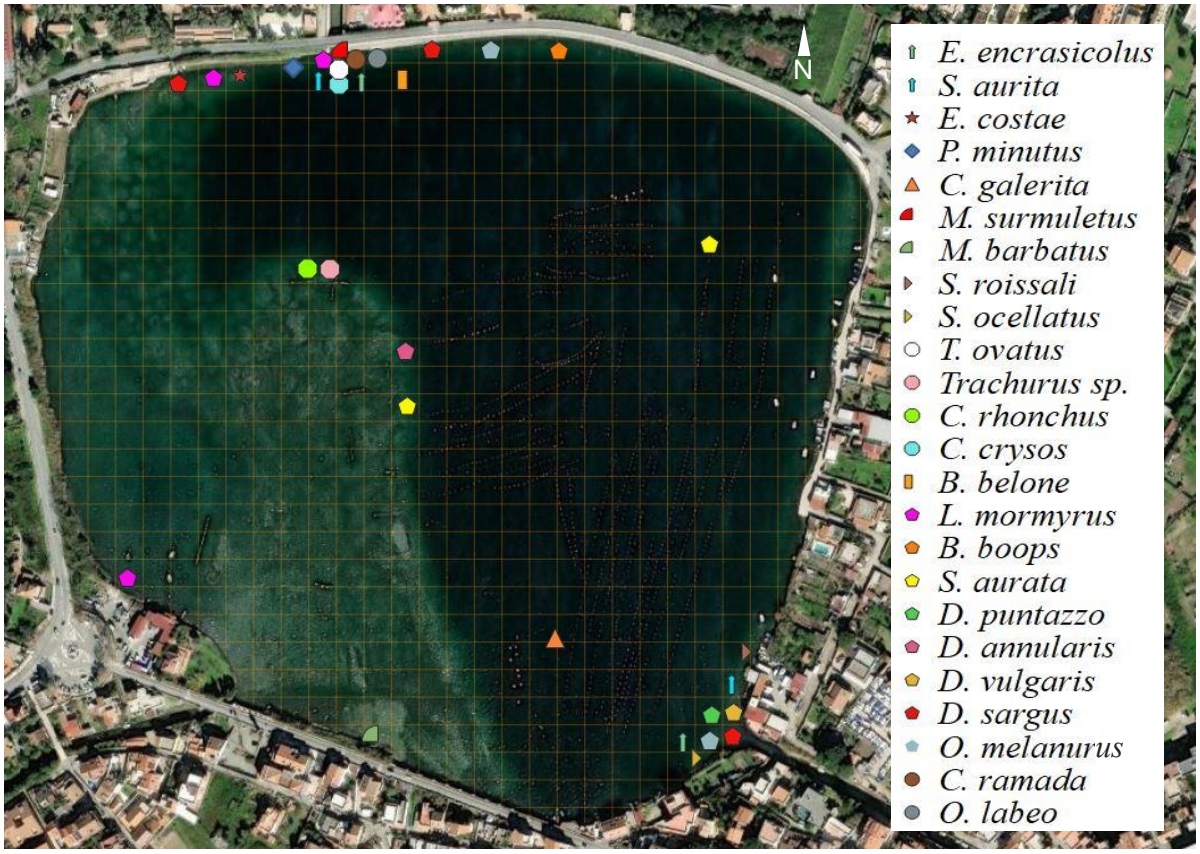

Figure 28; Spatial distribution of species classified as seasonal migrants in Faro Lake. The map highlights the preferential areas used during seasonal migrations, with particular reference to periods of higher abundance linked to their temporary presence driven by reproductive or feeding cycles. Map generated with QGIS.

3.4.1.3 Distribution of occasional visitors

The map shows the locations of occasional visitor species. The positions indicated correspond to the specific points where each species was observed once or only a few times. *C. julis, M. helena, S. porcus* and *S. scriba* have been observed in proximity of the entrance of the Faro Canal. *A. sphynx* was observed in the western part of the lake, at the end of the reed bed. *D. labrax* was observed in the center part of the lake. *A. imberbis* was found in correspondence with the entrance of the English Canal. *B. capriscus* was observed in the right side of the entrance of the English Canal (Fig. 29).

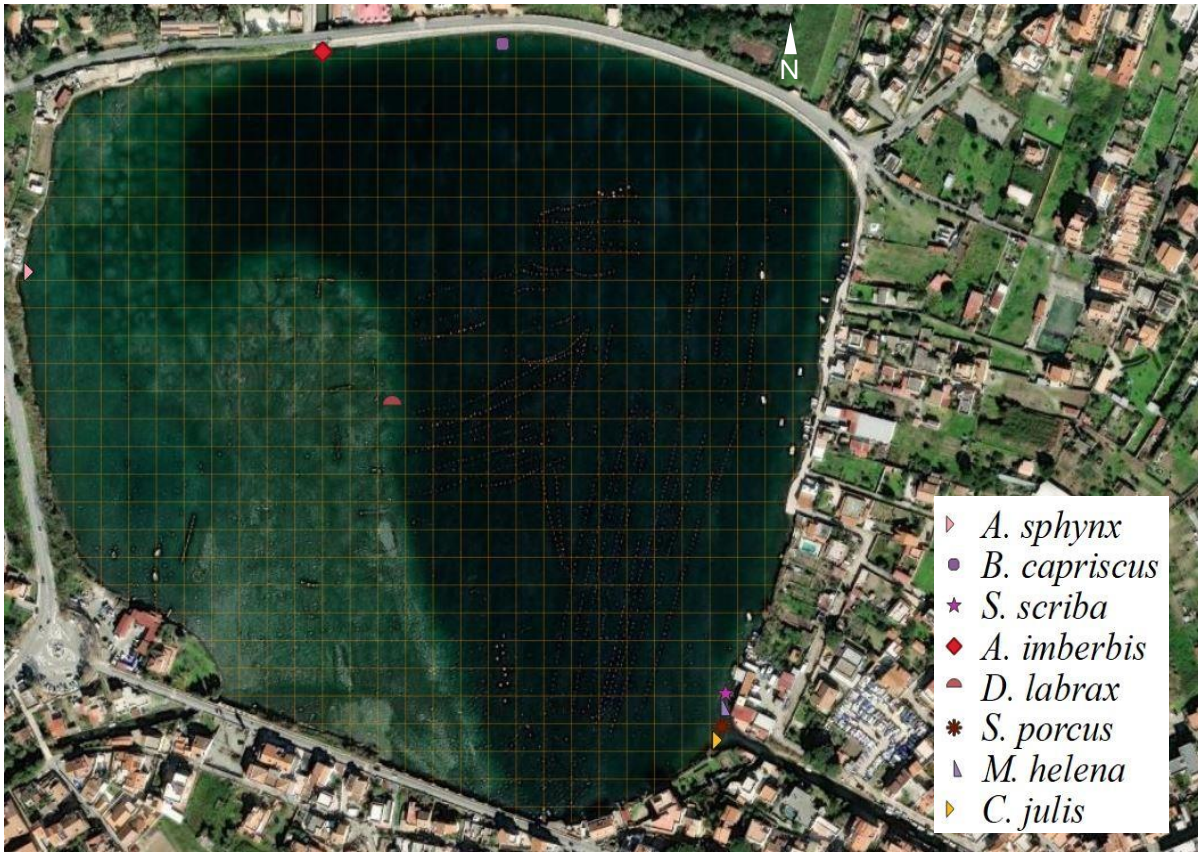

Figure 29; Spatial distribution of occasional visitors in Lake Faro. The highlighted points represent areas where these species were sighted sporadically or only once during the study period. Map generated with QGIS.

3.4.2 BRUVs

Eighteen species belonging to 9 family were recorded through the BRUVs technique. The most represented family was the Sparidae with 5 species, followed by the Gobiidae and Mugilidae with 3 species, Carangidae with 2 species and Atherinidae, Blenniidae, Dorosomatidae, Labridae and Pomacentridae with one species. 11 sedentary species and 7 seasonal migrants were observed. A total of 18 species were observed during the summer, 12 in autumn, 9 in spring, and 6 in winter (Tab. 23).

Table 23; List of teleost species observed in the Faro Lake through BRUVs technique, indicating the family, their functional ecological category (FEC) and seasonal presence.

Taxa	Family	FEC	Autumn	Winter	Spring	Summer
Chelon auratus (Risso, 1810)	Mugilidae	SS	*	*	*	*
Atherina hepsetus (Linnaeus, 1758)	Atherinidae	SS	*	*	*	*
Chelon labrosus (Risso, 1827)	Mugilidae	SS	*	*	*	*
Gobius niger (Linnaeus, 1758)	Gobiidae	SS	*		*	*
Gobius incognitus (Kovačić & Šanda, 2016)	Gobiidae	SS	*		*	*
Sarpa salpa (Risso, 1810)	Sparidae	SS	*	*	*	*
Lithognathus mormyrus (Linnaeus, 1758)	Sparidae	SM	*	*		*
Gobius cobitis (Pallas, 1814)	Gobiidae	SS	*			*
Sardinella aurita (Valenciennes, 1847)	Dorosomatidae	SM			*	*
Boops boops (Linnaeus, 1758)	Sparidae	SM			*	*
Caranx crysos (Mitchill, 1815)	Carangidae	SM				*
Mugil cephalus (Linnaeus, 1758)	Mugilidae	SS				*
Parablennius sanguinolentus (Pallas, 1814)	Blenniidae	SS				*
Trachurus sp.	Carangidae	SM				*
Chromis chromis (Linnaeus, 1758)	Pomacentridae	SS	*			*
Oblada melanurus (Linnaeus, 1758)	Sparidae	SM	*		*	*
Sparus aurata (Linnaeus, 1758)	Sparidae	SM	*			*
Thalassoma pavo (Linnaeus, 1758)	Labridae	SS	*	*		*

3.4.2.1 Total species richness detected by BRUVs

The trend in species richness over time, as observed using the BRUVs technique, is characterized by a decrease from autumn to winter (from 12 species to 6), followed by a progressive increase from winter to summer (from 6 species to 9), with the highest values recorded during the summer months (18 species) (Fig. 30). This trend is consistent with the results observed using the visual census technique (Fig. 26a).

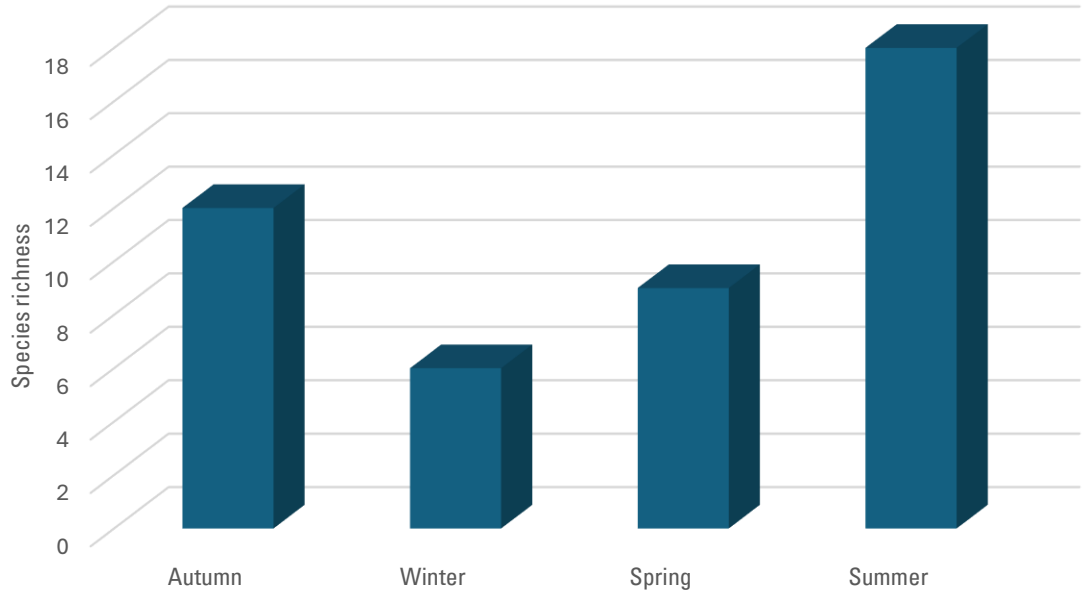

Figure 30; Graphical representation of total of the seasonal species richness e from October 2022 to September 2023.

3.4.2.2 First appearance time: Autumn

During the autumn months, it was observed that *Atherina hepsetus, Gobius niger, Gobius cobitis, Gobius incognitus, Chelon auratus and Chelon labrosus* were the first species interacting with the bait, with an average response time ranging from 1 to 1,8 minutes, followed by *Sarpa salpa, Thalassoma pavo* and *Oblada melanurus*, with response times ranging from 21,8 minutes to 27,4 minutes. The species with longer response interaction times were *Lithognathus mormyrus, Sparus aurata and Chromis chromis* with response times ranging from 38,2 minutes to 57,4 minutes (Fig. 31).

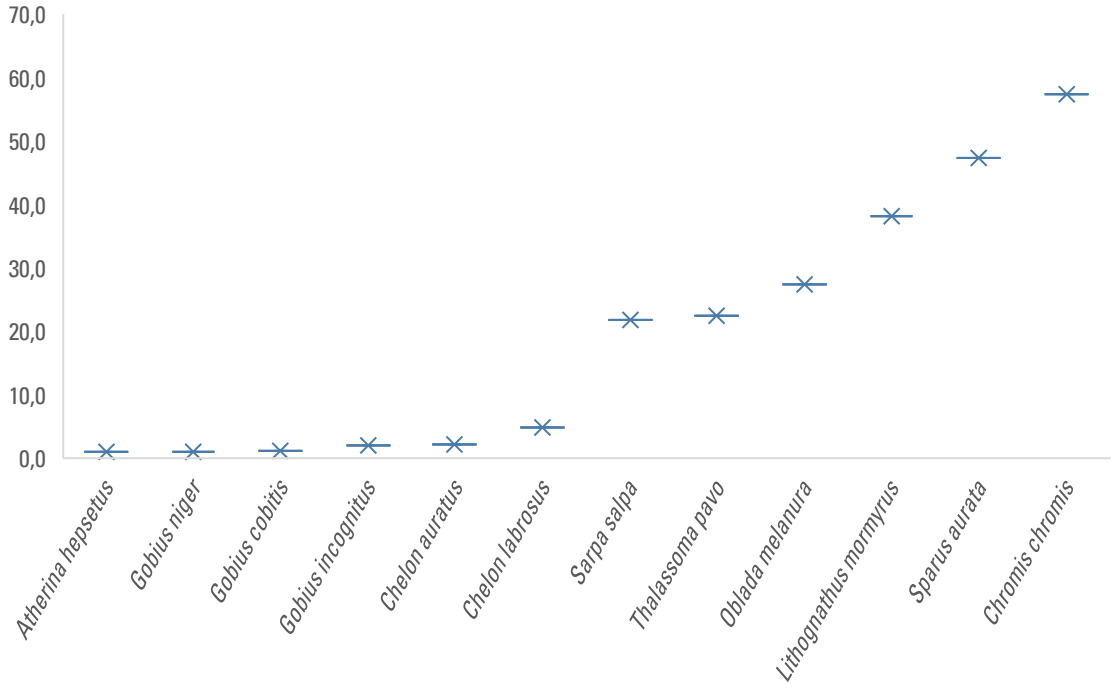

Figure 31; Autumnal first appearance time species.

3.4.2.3 First appearance time: Winter

During the winter months, it was observed that the interaction times after bait placement delayed. *Atherina hepsetus* remained one of the first species to interact with the bait, along with *Chelon auratus*, which were recorded at 4,5 and 5,9 minutes, respectively. *Lithognathus mormyrus* and *Chelon labrosus* followed later, with interaction times of 12,6 and 16,6 minutes, respectively. The species with the longest interaction times were *Sarpa salpa* and *Thalassoma pavo*, with times of 26 and 39,1 minutes respectively (Fig. 32).

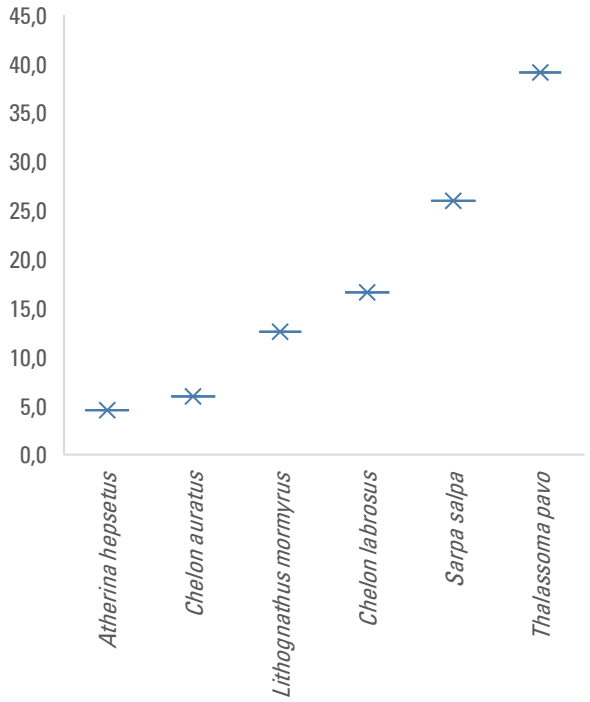

Figure 32; Winter first appearance time species.

3.4.2.4 First appearance time: Spring

During the spring period, the species began to interact with the bait again, showing less long response times. In the first 16,1 minutes, the arrival of the species occurred in increased gradual time order as follows: *Atherina hepsetus* (1,8 minutes), *Chelon auratus* (2,4 minutes), *Gobius niger* (3,9 minutes), *Gobius incognitus* (6,5 minutes), *Boops boops* (9,1 minutes), *Chelon labrosus* (11,9 minutes) *and Sardinella aurita* (16,1 minutes). *Sarpa salpa* and *Oblada melanurus* interacted with the bait with longer times, respectively with 31,6 minutes and 36,5 minutes (Fig. 33), confirming the behaviour also observed in previous seasons (Fig. 31; Fig. 32).

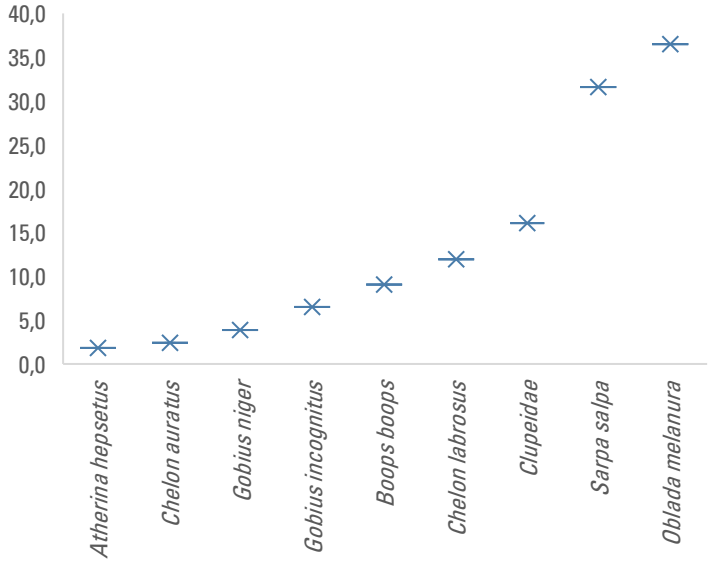

Figure 33; Spring first appearance time species.

3.4.2.5 First appearance time: Summer

The summer season was characterized by the highest number of species interacting with the bait. The first species to interact were *Chelon auratus, Atherina hepsetus* and *Chelon labrosus* with response times ranging from 1 to 1,7 minutes. Subsequently, *Gobius niger*, *Gobius incognitus*, *Sarpa salpa*, *Lithognathus mormyrus*, *Gobius cobitis*, *Sardinella aurita*, *Boops boops*, *and Caranx crysos* arrived, with progressively longer response times ranging from 4,9 to 13,4 minutes. *Mugil cephalus*, *Parablennius sanguinolentus*, *Trachurus* sp., *Chromis chromis* and *Oblada melanurus* followed, with response times ranging from 21,0 to 27,4 minutes. *Sparus aurata* and *Thalassoma pavo* arrived with respectively response times of 38,1 and 51,4 minutes (Fig. 34).

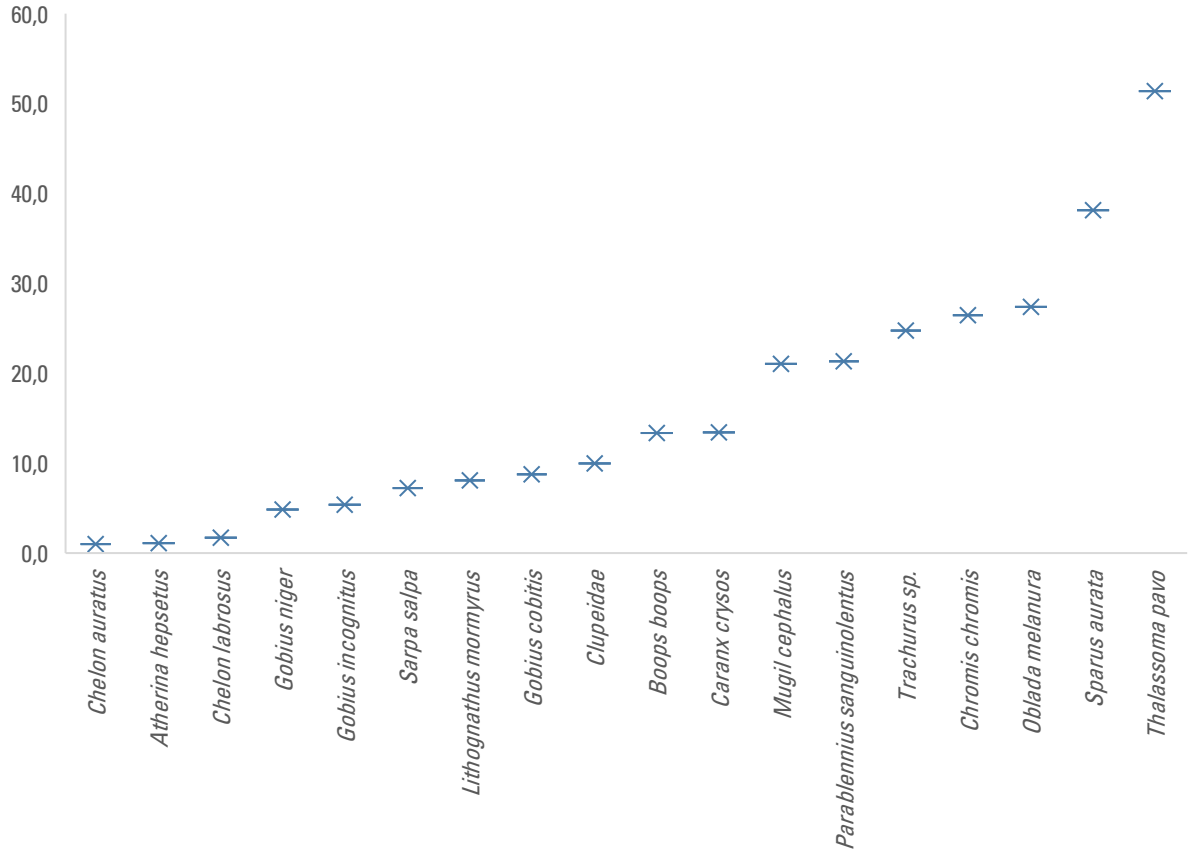

Figure 34; Summer first appearance time species.

3.4.2.6 Species accumulation time

As shown in (Fig. 35), the time required to record the first cumulative sightings of the fish species observed in Faro Lake is represented. The species were detected in distinct groups along the timeline. In the first 2 minutes, the following species were observed: *A. hepsetus*, *G. niger*, *G. cobitis*, *G. incognitus*, *C. auratus*, and *C. labrosus*. Subsequently, between minute 7.2 and minute 13.4, five additional species were recorded: *B. boops*, *S. salpa*, *L. mormyrus*, specimens of the *Clupeidae* family, and *C. crysos*. Between minute 21.12 and minute 27.4, six more species were observed: *M. cephalus*, *P. sanguinolentus*, *T. sp.*, *C. chromis*, *T. pavo*, and *O. melanurus*. Finally, the last species, *S. aurata*, was recorded at minute 38.1. (Fig. 35)

In conclusion, considering all the surveys conducted over the seasons and adding up the number of species for each minor time of first appearance, a total of 38.1 minutes was required to record all 18 species.

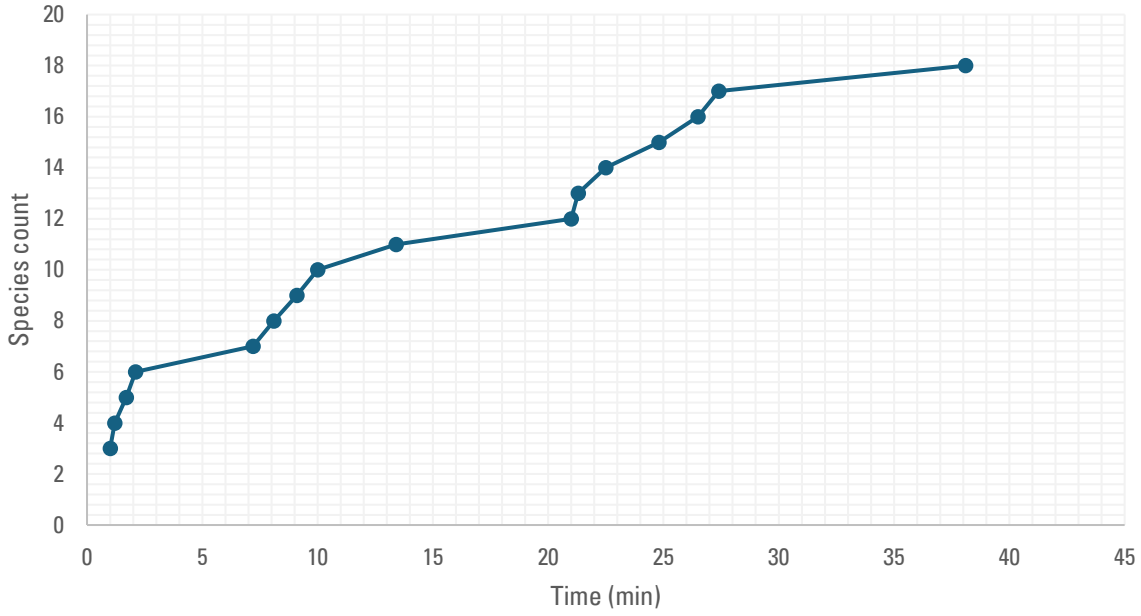

Figure 35; Cumulative first sighting of the fish species.

3.4.3 Visual census and BRUVs: species richness comparison

The comparative evaluation of the two methodologies showed that species richness followed a similar trend in all seasons, with some slight differences for the BRUVs, where three fewer species were recorded in spring than in autumn. The visual census consistently recorded more species in each season.

Summer represented the period with the highest specific richness recorded. Winter represented the period with the lowest specific richness (Fig. 36).

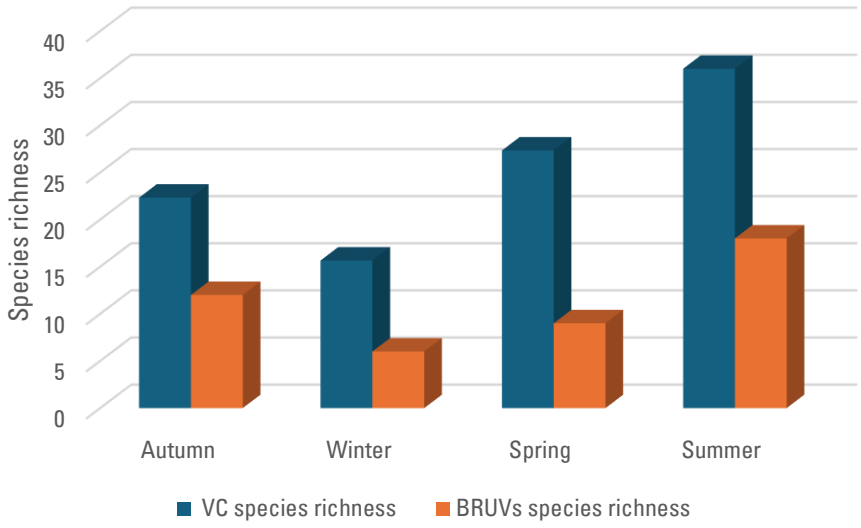

Figure 36; Graphical representation of the comparison of the seasonal species richness detected by the two applied techniques.

3.5 G. niger interaction with seafloor litter

A total of 492 seafloor litters were recorded along transects in the four sites of Faro Lake. Mostly of these were glass bottles (N = n° of ML) N = 303, followed by N = 97 plastic bottles and N = 92 aluminum cans (Tab. 24). The spatial distribution of ML showed the highest densities in site 2, whereas the lower abundance was observed in site 4 (Fig. 37).

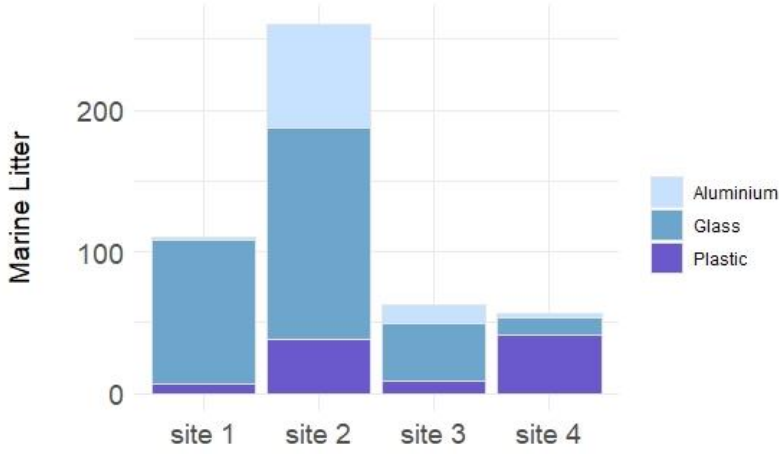

Figure 37; Number of ML material compositions recorded in each site

In Faro Lake, black goby was the predominant species inhabited seafloor litter (*n* = n° of specimens; *n* = 58), only one glass bottle in site 2 was occupied by *S. pavo* (Fig. 38; Tab. 25). Regarding this, all statistical tests were run considering black goby the sole species inhabit MLs in Faro Lake. Moreover, in each site we found seafloor litter with only one black goby inside. When more than one specimen was recorded near the anthropogenic litter the individual with the biggest size swam into it.

*Figure 38; Representative images of species inhabited seafloor litter and anthropogenic items found in the four different sites and in Canale Faro (*T. pavo *in aluminium can).*

Comparing the material composition of seafloor litter, black goby was observed mainly in glass (N = 36) and plastic bottles (N = 21) except for two individuals in site 2 recovered in aluminum can. In site 4 only plastic bottles were observed inhabited (Tab. 25).

Table 24; *Marine litter composition inhabited by black goby. Length and width of seafloor litter (mean ± SD) recorded in the sampling area. N = number of inhabited seafloor litter in the different sites.*

	Length (mm)	Width (mm)	Site 1	Site 2	Site 3	Site 4
			N	N	N	N
Glass bottle	270.1 ± 58.4	113.0 ± 73.5	13	14	9	0
Plastic bottle	300.5 ± 41.7	196.5 ± 160.5	1	3	2	15
Aluminum can	173.0 ± 38.2	118.0 ± 11.3	0	2	0	0

Site significantly influenced the occupancy of seafloor litter across the four sites (Tab. S1; GLM: $P < 0.05$). The %IR was significantly higher in site 4 respect the other three sites (27%) followed by 17% in site 3, 13% in site 1 and 7% in site 2 (Fig. 39). Furthermore, a significantly difference of %IR was detected between site 2 compared to site 3 and 4. These significant

differences suggested us to incorporate site as random effect into the forward statistical analysis.

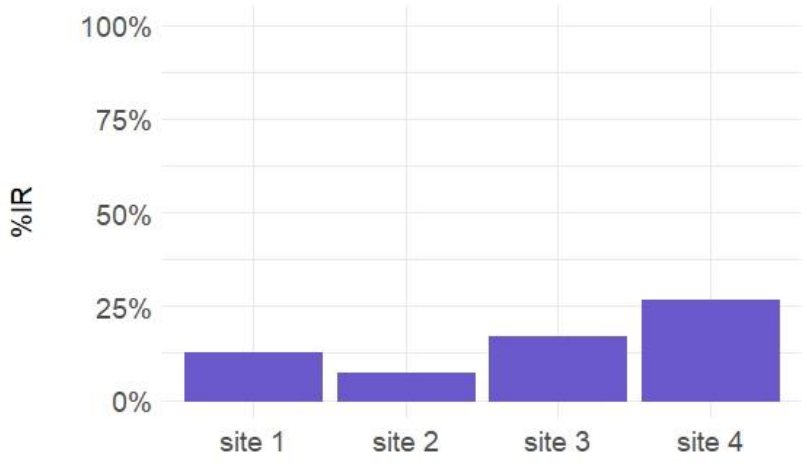

Figure 39; Percentage of ML inhabited by black goby (%IR) in the four different sites

Colour of bottles and cans in Faro Lake was represented by brown (N = 220), green (N = 77), transparent (N = 64), grey (N = 48), white (N = 21) and orange, red, yellow and light blue (N < 5). Nevertheless, black goby was not found in orange, red, yellow and light blue ML and only two individuals in two grey litters. No statistically significant difference of %IR between colours (Fig. 40, Tab.S1; GLMM: $P > 0.05$).

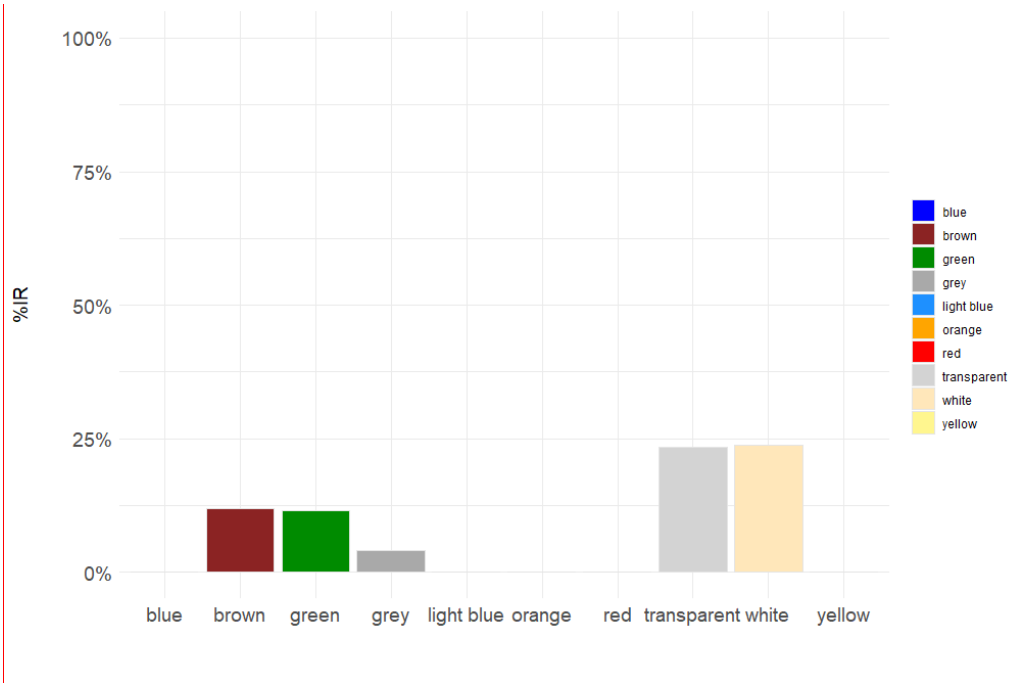

Figure 40; Colour percentage of ML inhabited (%IR) by black goby

Based on the coverage rate, seafloor litter was divided into items with high (3, 4, 5 rank) and low (0, 1, 2, rank) colonization rate of macro-benthic faunal and vegetal taxa. The analysis showed that black goby prefers to inhabit ML with a high colonization rate (Fig. 41, Tab.S1; GLMM: $P < 0.05$).

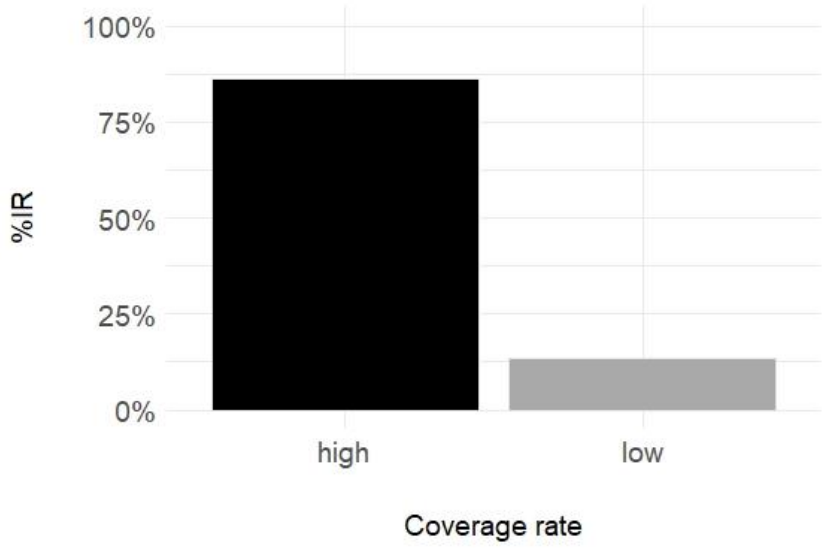

Figure 41; Coverage rate percentage of ML inhabited (%IR) by black goby

Furthermore, during a successive UVCs in the following month we recorded three glass bottles, two aluminum cans, one plastic tank utilized as microhabitat by *P. sanguinolentus*, *T. pavo* and *M. helena* in Faro Canal and Due Torri Canal, two shallow canals linking Faro and Ganzirri Lake with the Strait of Messina respectively (Tab. 25).

Table 25; Morphometric data of the species inhabited ML in all sampling location

Species	Length mm	Sampling area	Marine litter
Gobius niger	92.5	Canale Faro	Glass bottle
Gobius niger	55.0	Canale Faro	Glass bottle
Gobius niger	91.0	Canale Faro	Glass bottle
Salaria pavo	89.0	Faro Lake	Glass bottle
Parablennius sanguinolentus	92.0	Canale Faro	Aluminum can
Thalassoma pavo	88.0	Canale Faro	Aluminum can
Muraena helena	410.0	Canale Due Torri	Plastic tank

4 DISCUSSION

Transitional environments, such as coastal lagoons and brackish lakes, exhibit unique and highly variable characteristics shaped by their status as closed or semi-closed systems. However, the characteristics of these environments can be altered by environmental physico-chemical factors variation (Rosa et al., 2022; Swain et al., 2021), morphological and structural dynamics (Meredith et al., 2022; Pérez-Ruzafa et al., 2024) and anthropogenic impacts, which can lead to biodiversity loss and affect ecosystem functioning, as highlighted by Danovaro & Pusceddu (2007) and Rodrigues-Filho et al. (2023). These environments are crucial for the conservation of biodiversity and providing ecosystem services that support high species diversity communities (Pérez-Ruzafa et al., 2020). Recent studies investigated the influence of environmental factors on fish populations and highlighted their value as areas of nursery (Erzini et al., 2022; Guerreiro et al., 2021), as a refuge against predators (Whitfield, 2020) and feeding areas (Prado et al., 2013). However, to date, few studies have applied an integrated approach to investigate the effect of seasonal and spatial variations in the distribution of species of the teleost community in these environments (Castillo-Rivera et al., 2005; Moreno-Pérez et al., 2024; Turtora & Schotman, 2010; Velázquez-Velázquez et al., 2008) and no similar studies have been conducted in Faro Lake. Faro Lake is a coastal lagoon extensively studied for its bacterial communities (Esposito et al., 2024; Gugliandolo et al., 2011; Raffa et al., 2019; Saccà et al., 2008; Zaccone et al., 2014), benthic organisms (Bruno et al., 2024; Furfaro et al., 2018; Giacobbe et al., 2024, 2024; Lunetta et al., 2024), planktonic dynamics (Furfaro et al., 2018; Giuffre & Pezzani, 2005; Zagami et al., 2018), and water circulation and processes (Ferrarin et al., 2013) representing a coastal lagoon of high scientific interest for its peculiar characteristics. Despite this, research on the fish community inhabiting this brackish ecosystem is fragmentary and dated. Although several studies (Badalamenti et al., 1990; Cavaliere, 1967; Grassi, 1903; Potoschi et al., 2002; Sparta, 1934; Spartà, 1936; Sparta, 1936, 1948; Spartà, 1948; Sparta, 1950) have focused on the fish fauna of the Messina lakes, providing information on the species present in the area of the NOR “Laguna di Capo Peloro”, these are dated and obtained data with invasive and in some cases limited sampling methods; data obtained were often related to single observations or merely descriptive (Giacobbe et al., 2018; Spinelli et al., 2017, 2020). The present study adopted an integrated approach that allowed the collection of detailed data on the fish fauna of the Faro Lake, using two non-invasive monitoring techniques, the visual census and the baited remote underwater video system, allowing the collection of structured

and continuous data on teleost assemblages. In addition, this study analyzed the dynamics of teleost communities throughout the year, highlighting the correlation with environmental variables and species distribution patterns, and in some cases behavioral in situ observations (section 4.2.1).

4.1 Environmental factors effect on teleost presence and distribution

Fish communities in coastal lagoon systems are strongly influenced by environmental parameters and anthropogenic pressures, which determine their qualitative and quantitative variation. Seasonal dynamics, such as fluctuations in temperature and dissolved oxygen, therefore take a decisive role in structuring fish assemblages, as observed in another study on semi-arid subtropical lagoons (Salas-Mejía et al., 2024). Environmental parameters measured in Faro Lake during the study period showed clear seasonal variability. For temperature and dissolved oxygen, fluctuations were recorded throughout the year, showing strong significative differences between winter and summer for temperature and between spring and summer for dissolved oxygen. The periods between autumn and winter for temperature and between autumn and spring and between winter and summer for dissolved oxygen were also significant. This confirms the seasonal trend of Faro Lake, as reported by other authors (Capillo et al., 2018; Rizzo et al., 2024; Sanfilippo et al., 2022). These changes directly influenced the community, contributing to define periods of increased richness and abundance. Salinity and pH showed no significant changes and did not affect the fish community. However, although environmental parameters showed seasonal trends consistent with past studies and in particular temperatures during the study period were similar to those recorded in previous years, (Cavaliere, 1967) reported that during the warm season, when temperatures reach 26–27°C, fish species tend to suffer. This can lead to fish mortality events or to species abandoning the lake to exploit the cooler marine waters.

Most of teleost are ectothermic animals and consequently particularly sensitive to temperature. Optimal thermal conditions can stimulate growth and reproduction (Pankhurst & Munday, 2011); while in the opposite case, extreme or sub-optimal temperatures can reduce survival and compromise the adaptability of the species (Islam et al., 2022). Temperature is directly linked to physiological processes, influencing growth, reproduction and survival of species. It furthermore moderates their ability to adapt to environmental changes by adopting different behavioral strategies, migration and distribution patterns (Alfonso et al., 2021; Little et al., 2020). Understanding the mechanisms at the basis of thermal responses is essential to predict the vulnerability and dynamics of fish species.

In accordance with Guerreiro et al. (2021) and Holm-Hansen et al. (2019) temperature was the most influencing factor in the structure of the community. In fact, an increase in species richness has been recorded, which follows the typical regular seasonal temperature trend (Fig.

25a; Fig. 26a). However, the total abundance of the species did not follow this trend. Data collected showed the highest abundance of *Atherina hespetus* compared to all teleost identified in this research, especially during the autumn, winter and spring periods. Towards the end of May, its abundance began to gradually decrease during summer months. Although *A. hepsetus* was the most abundant species, it was the only species to show a decrease during the summer. This dynamic influenced the variation of total abundance trend. Statistical analysis of the temperature trend showed a strong significance between the spring months, during which *A. hepsetus* reproduces (https://www.fishbase.se), and the summer months (Fig. 9). According to information found on FishBase, *A. hepsetus* exhibits a preferred temperature range between 16.1 and 21.1 °C, with a mean of 19 °C. Therefore, it can be assumed that an increase in temperatures within the area may be unfavorable for this species, causing many individuals to move away from the lake. Simultaneously with the numerical decrease of *A. hepsetus*, seasonal migrants such as *Caranx crysos*, *Caranx rhonchus*, *Trachurus* sp., and *Trachinotus ovatus* were observed in the lake for the first time during the monitoring period. As noted by Psomadakis et al. (2011), these species, which prefer warmer waters, are gradually expanding their range northward in response to the warming of the waters. This phenomenon aligns with the observations of Raya & Sabatés (2015), which revealed that these Carangidae species are particularly abundant in the warmer waters of the southern Mediterranean, showing a correlation with rising temperatures, which explains their increased presence in the lake during the warmer months. In addition, according to the literature (Stergiou & Karpouzi, 2002), the diet of species belonging to the family Carangidae includes members of the genus *Atherina*, suggesting that the numerical reduction of *A. hepsetus* could also be attributed to predation, highlighting that the predation phenomenon should not be underestimated.

As already mentioned, Faro Lake is connected to the Ionian and Tyrrhenian Seas by two canals: the English Canal, which connects the lake with the Ionian Sea, and the Faro Canal, which connects the lake with the Tyrrhenian Sea. The opening of the English Canal is regulated by humans in order to regulate the lagoon water temperature; in fact, to allow the temperature mitigation, the canal is open in summer period until the end of the hot period; in this way the Tyrrhenian Sea water enter the lagoon, avoiding the occurrence of high water temperature (>32-33 °C) that can affect the vitality of mussels and other aquatic species. This opening corresponded to a variation in teleost species number inside Faro Lake. In fact, species as *O. labeo, C. ramada, D. puntazzo, C. rhonchus, T. ovatus, M. surmuletus, E. coste, S. aurita, E. encrasicolus,* Alosidae species, *A. imberbis, D. labrax, S. porcus, M. helena, C. julis,* increased

the species richness (from 37 to 52) in Faro Lake. Despite this, statistical analyses revealed that while more species were observed numerically following the opening of the English Canal, changes in species richness and abundance were not statistically significant. This suggests that although the opening of the canal facilitated the entry of new species, ecological dynamics and species interactions may have limited the overall impact on local biodiversity. Furthermore, it is possible that environmental and biological factors, such as competition and predation, played a crucial role in maintaining stable levels of richness and abundance among resident species, despite the alterations induced by human activity.

4.2 Sedentary species

Sedentary species were found in almost all the visual censuses carried out, indicating a strong affinity with the characteristics of Faro Lake and a greater success of their adaptive strategies, as in the case of *Gobius niger* (section 4.2.1). In particular, several sedentary species, including *Chelon labrosus*, *Chelon auratus*, *Mugil cephalus*, *Sarpa salpa*, *Salaria pavo*, *Parablennius sanguinolentus*, *Gobius niger*, *Gobius cobitis*, *Gobius paganellus*, and *Gobius incognitus*, exhibit an increase in their abundance corresponding to higher temperatures, with peaks observed in June and July (Fig. 18b). However, during the winter period, their abundances were significantly reduced. In similar environments in other geographical areas of the Mediterranean Sea, other authors observed these species, typical of these environments, finding seasonal fluctuations similar to those of the present study, with greater abundance linked to higher temperatures and a lower during the winter months, thus confirming the influence of environmental factors on their distribution and abundance (Cubedo et al., 2008; Pombo et al., 2005). These fluctuations could be linked to the tendency, in the colder months, to exploit deeper areas, as observed by (Kellnreitner et al., 2012) and the greater difficulty of the visual census in detecting such variations. Furthermore, the lower abundance observed could be attributed to the fact that, during the winter period, the metabolic activities of fishes are reduced, leading them to take refuge in hiding places and adopt behavioral patterns which are less evident, as previously observed in several studies (Crawshaw, 1984; Speers-Roeschet al., 2018). Monitoring using BRUVs showed also that temperature influenced the first appearance time of the species. In colder periods, the first appearance time was longer, while in warmer periods it was shorter, due to increased metabolic activity. In addition, species richness followed the same trend as in visual census monitoring, with lower species richness in the winter period. The results of the present study revealed important differences in the presence and classification of several teleost species compared to previous research on Faro Lake. For example, *Chromis chromis*, observed by Potoschi (2002) and classified as an occasional visitor, was reclassified as a sedentary species. This new classification is based on repeated observations of individuals at specific points in the lake over several months. *Epinephelus aeneus*, previously considered an occasional visitor, was also observed repeatedly in the present study, represented by a single large individual that was always the same specimen, suggesting that this species lives permanently in the lake. Another significant difference concerns *Epinephelus marginatus*, a species not reported by Potoschi (2002) but reported as a seasonal migrant by Cavaliere (1967) and as an occasional visitor by Grassi (1903). In the present work,

E. marginatus was consistently observed at specific points in the lake, leading to its reclassification as a resident species. In addition, two species not reported in previous works by Potoschi, Cavaliere, and Grassi were documented in this study: *Gobius incognitus* and *Mycteroperca rubra*. Considering the diversity of sedentary species belonging to the family Serranidae present in the lake, these results highlight the ecological importance of Lake Faro as a habitat for this group. For example, *E. marginatus*, classified as an endangered species on the IUCN Red List (https://www.iucnredlist.org), is a significant case that emphasises the need to continue studies using only non-invasive methods to monitor this area. Finally, some differences may also be attributed to the various observation and sampling methods employed.

4.2.1 Interaction between Gobius niger and marine litter

Seafloor litter as suitable substrate for macrozoobenthos growth is well known but information on ecological aspects regarding the use of ML by benthic fish is scarce (Angiolillo et al., 2015; Angiolillo & Fortibuoni, 2020; Bottari et al., 2024; Han et al., 2023). In this contest, our study evidenced that seafloor litter is selected by benthic fish as alternative shelter in brackish environments, mainly when the presence of natural refuges is a limiting factor. Such anthropogenic waste could have the potential role to mitigate this loss and support a higher abundance of sessile and mobile taxa in marine environment with a low habitat complexity (Pham et al., 2014).

The massive presence of seafloor litter found during our survey in Faro Lake is mostly originated from land-based sources. Our data showed a clear correlation between distribution of ML, anthropogenic pressure and the physical characteristics of the lake, driving by limited water exchanges, high population density and tourism (Bergmann et al., 2015). Indeed, sites located closer to tourist areas and anthropogenic activities such as restaurants and mussel farming exhibit greater ML density. The accumulation of anthropogenic waste is facilitated due to the easy access to the site for humans and the proximity to commercial establishments, which increases the probability of improper waste disposal (Baini et al., 2024; Garcés-Ordóñez et al., 2020; Simeonova & Chuturkova, 2019; Skirtun et al., 2022).

On the other hand, sea currents are an additional dispersal force of seafloor litter (de Carvalho-Souza et al., 2018; Thiel & Gutow, 2005). Faro Lake receives direct and periodic input of Ionian Sea water mainly from Faro Canal. This current influenced by the wind and tides determinate the spread of particulate matter inside the lake and could influence ML accumulation (Bergmann et al., 2015; Ferrarin et al., 2013). In our study Site 4 is characterized by the presence of a reed bed and shows a higher concentration of plastic bottles compared to other materials. The frontal position of this site to the Faro Canal could improve the transport of floating plastic debris directly to this area thanks to the incoming water from Ionian Sea. The action of current may further push this debris into the reed bed accumulating it among the vegetation and eventually sink it into the shallow waters. Instead glass and metal sink rapidly closer to the release area (Pham et al., 2014). Furthermore, in site 4 the massive presence of reed beds makes this spot isolated and not easy to reach by citizens. This characteristic could

explain the lower presence of glass and metal items released, making on the contrary the reed bed an area prone to the deposition of floating plastic litter.

Site 4 is also notable for the high occupancy rate of seafloor litter by gobies attributable to the scarcity of alternative natural habitats in this area as previously observed by Bohnsack et al., 1989 (Bohnsack, 1989). Site 4 is characterized by a fine sediment layer with limited presence of hard substrates and natural cavities formed between pebbles, bioconstruction, rocks, and vegetation typical of the other sites and preferred by gobies (Matern et al., 2021). The utilization of seafloor litter by black goby emphasizes their importance as refugees in an environment lacking natural alternatives. In this site gobies tend also to aggregate around available and more copious artificial structures but only the largest individual inhabits the seafloor litter. This behavior is in line with findings that limited shelter availability often leads to increased competition for space, especially in Gobiidae family. Intraspecific competition is particularly pronounced when multiple individuals are forced to occupy the same artificial shelter. Larger individuals monopolize available refugees to secure reproductive success (Kroon et al., 2000). Dominant gobies typically exhibit aggressive behaviors to maintain exclusive access to these refugees, as observed in other goby species during both reproductive and non-reproductive periods (Immler et al., 2004).

In this study the analysis revealed a significant preference by gobies for bottles with a high rate of colonization by benthic organisms. This behavior could be attributed to various ecological and biological factors. Biological coverage offers a more complex structure and resembles natural habitats more closely, making them more attractive to gobies in an environment lacking adequate options (Perkol-Finkel et al., 2006).

Contrary to expectations, no significant preference was found among gobies for a specific seafloor litter color. This result might indicate that the color of the refuge is not a determining factor in habitat selection for this species. Other factors such as rate of colonization by benthic organisms and unavailability of natural shelter seem to have a greater influence on refuge selection. Further studies may be necessary to confirm this hypothesis and to better understand the habitat selection criteria of gobies in brackish environments.

Brackish environments are particularly threatened by the massive presence of seafloor litter. Monitoring the marine litter is a useful tool to evaluate the risk of habitat degradation, loss of biodiversity and ecosystem services (Kumar et al., 2021; Mancini et al., 2021; Rizzo et al., 2021; Tekman et al., 2022). Furthermore, require data on the use of seafloor litter as refuge is

important to better assess the types and levels of interactions among different taxa and marine litter (de Ramos et al., 2024). This information is important for the management strategy based on removing marine litter without affecting biota that is associated or interact with it (Zielinski et al., 2019).

4.3 Seasonal migrants

Seasonal migrants were observed only during specific periods of the year, suggesting that Faro Lake serves as a key resource for these species, potentially supporting various ecological strategies such as feeding, reproduction, growth, and predation. For instance, Franco et al. (2006) observed several of these species, such as *B. belone*, *C. ramada*, *D. annularis*, *E. encrasicolus*, *L. mormyrus*, *M. surmuletus*, and *P. minutus*, utilizing coastal lagoons as migratory species, demonstrating ecological patterns comparable to those observed in Faro Lake. Similarly to the observation during the present monitoring, Pérez-Ruzafa et al. (2019) identified *S. aurata*, *D. sargus*, and *D. puntazzo* as typical seasonal migrants that likely exploit such lagoons for juvenile development before migrating to the sea in autumn. In particular, the observations conducted on *L. mormyrus* in the present study (section 4.5) confirm this pattern, showing a similar use of coastal lagoons as juvenile development areas before migration to the sea during the coldest months of the year. These species are commonly found in such environments, which further supports the idea that Faro Lake functions as an important resource for their life cycles. *E. costae* was observed exclusively during the warmer months and, in one census in particular, 35 individuals were recorded, a significantly higher number than on other censuses, and all the individuals observed were juveniles. Although data on the presence of this species in coastal lakes are limited, it had already been reported by Chaoui et al. (2006) in a coastal lagoon in Algeria (Mellah). Moreover, although this result seems unusual, it is widely documented that juveniles of *E. costae* prefer shallower waters than adults (Francour and Pollard, 2018; La Mesa, 2006) as was also observed in Faro Lake.

Concerning differences from previous studies in Faro Lake, it was noted that Potoschi et al. (2002) had recorded *C. crysos* and *S. roissali* as occasional visitors, and *O. labeo* as a resident species. Cavaliere (1967), on the other hand, had recorded *S. ocellatus*, a species that had not been observed by Potoschi et al. (2002), and had classified *O. labeo* as an occasional visitor. Grassi (1903) had classified *B. boops* as a resident species, while Cavaliere (1967) reported it as a seasonal migrant. Potoschi et al. (2002) had not detected *B. boops*. Furthermore, *C. rhonchus, C. galerita, E. costae* and *P. minutus* had not been observed in previous censuses.

4.4 Occasional visitors

Occasional visitors were recorded either a single time or over a short time interval during the entire monitoring period, highlighting their rare presence in Faro Lake. These species were not detected during the preliminary surveys conducted prior to the monitoring activities and were not observed during the subsequent months dedicated to collecting additional data.

In October 2022, the species *A. sphynx* was recorded along a small concrete pier located on the western side of Faro Lake. This species has previously been documented in similar environments, such as the Mar Menor, where it was observed in artificial rocky substrates (Pérez-Ruzafa et al., 2006). Additionally Tiralongo et al. (2016) reported its presence in southeastern Sicily, with the highest abundance recorded on sub horizontal hard substrates covered with algae. Despite expectations of further sightings, *A. sphynx* was not observed again during the subsequent surveys.

A. imberbis was observed during July and August 2023, with 1 or 2 individuals. Its presence may be explained by the fact that *A. imberbis* inhabits the beach rock area on the Ionian side of the Natural Oriented Reserve, where it was observed by Capillo et al. (2018). The scarce abundance could explain its occasional entry into the lake through the canals. Furthermore, *A. imberbis* was recorded by Franco et al. (2012) in the Mar Menor, where it was classified as a "marine straggler," a behavior consistent with the observations made in the present study. In the past, the species was described as a resident in Faro Lake by Potoschi et al. (2002). The reduced number of individuals may suggest that the current conditions of the lake are no longer favorable for this species, as they were in the past.

B. capriscus was observed in November, and its presence can be attributed to chance. This pattern is confirmed by Grassi (1903) and Potoschi et al. (2002), while Cavaliere (1967) described it as a seasonal migrant.

S. scriba is a common species of coastal environments, whose presence in Faro Lake, observed in May 2023, can be attributed to chance. This species was not observed in the study by Potoschi et al. (2002), while Grassi (1903) and Cavaliere (1967) confirm its presence as an occasional visitor.

D. labrax, S. porcus, M. helena and *C. julis* are common species of coastal environments. Their presence in the lake was observed in July 2023, coinciding with the opening of the English

Canal. This supports Cavaliere's (1967) observation that biodiversity is positively influenced by the opening of canals.

4.5 Species distribution and taxa representativeness

A relevant aspect emerged from the analysis of the teleost community was the spatial distribution of the species, with a clear predilection for the perimeter areas of the Faro Lake. The perimeter areas of the lake are characterized by shallow water and offer greater structural complexity than the central part. This structural complexity, provided by rocky conglomerate substrates, bioconstructions, submerged vegetation, and accumulations of organic material, creates favorable conditions for teleost to find refuge, feed, and reproduce (Cataudella et al., 2015; Verdiell-Cubedo et al., 2013; Zamora-López et al., 2023). For example, many sedentary species belonging to the Gobiidae and Blenniidae families have shown a strong link to these microhabitats, confirming the observations of other authors (Tiralongo et al., 2016; Tiralongo et al., 2021). Similar patterns have been observed in other coastal lagoons and semi-enclosed ecosystems. For instance, in the Mar Menor lagoon, species such as *Gobius cobitis, Gobius niger* and *Salaria pavo*, belonging to the Gobiidae and Blenniidae families respectively, exhibit strong associations with shallow areas rich in submerged vegetation and bioconstructions (Cubedo et al., 2008). In the case of *S. pavo*, its greater frequency near the rows of the mussel farm is attributable to its affinity for these types of structures, as already documented by Bitetto et al., (2024) Similarly, Guerrero-Gómez et al., (2022) found many similar species distributed along the shallow perimeter areas of the Mar Menor lagoon, including *B. belone*, *S. pavo*, *T. ovatus*, *S. aurita*, *E. encrasicolus*, *G. cobitis*, *G. niger*, *G. paganellus*, *D. labrax*, *C. auratus*, *C. ramada*, *M. cephalus*, *O. labeo*, *M. barbatus*, *M. rubra*, *B. boops*, *D. annularis*, *D. puntazzo*, *D. sargus*, *D. vulgaris*, *L. mormyrus*, *O. melanurus*, *S. salpa*, *S. aurata*, and Syngnatidae species. These findings further emphasize the importance of shallow, structurally complex areas as key habitats for a wide variety of teleost species, consistent with the patterns observed in Faro Lake. In another study (Verdiell-Cubedo et al., 2013) observed that vegetated and shallow muddy habitats supported higher teleost abundance, biomass, and diversity compared to deep sandy areas. Dominant families included Mugilidae, Gobiidae, Sparidae, and Atherinidae. These observations align with observations carried out in Faro Lake during the monitoring period, where shallow and structurally complex habitats shaped the distribution and diversity of the teleost community.

Moreover, *Aphanius fasciatus* was observed exclusively in shallow waters of Faro Lake, consistent with its known preference for such habitats in other Mediterranean lagoons, where

it relies on submerged vegetation and organic-rich substrates for breeding and feeding (Maltagliati, 1999; Pappalardo et al., 2024).

Another factor that could affect the distribution of species in the perimeter areas of Lake Faro is the presence of the chemocline at a depth of about 10 meters, which prevents life below that level (Sanfilippo et al., 2022).

The Sparidae was the most represented family, with 9 species observed (*Sarpa salpa, Oblada melanurus, Diplodus sargus, Diplodus vulgaris, Diplodus annularis, Diplodus puntazzo, Sparus aurata, Boops boops, Lithognathus mormyrus*). Except for the sedentary species *Sarpa salpa*, which was observed in every visual census, the other species were observed only at specific times of the year and in minor abundance. The species *Lithognathus mormyrus* was observed in high abundance (1659 specimens) during the autumn period, with juveniles exclusively. In the summer months, however, adult specimens were sighted, while juvenile individuals appeared again in the following autumn. This seasonal trend was also confirmed the following year, thanks to further surveys conducted in order to collect images of the species surveyed. As observed by Sumer et al. (2014), temperature is one of the most relevant environmental factors for the gonadal development of *Lithognathus mormyrus*. Based on these observations and the dynamics already documented in other geographical areas of the Mediterranean (Reis & Ateş, 2020), it can be assumed that adult individuals use the lake as a breeding area in the summer months, and as a nursery and growth area for juvenile stages in the following months. During the winter period, after growth, the striped seabream leaves the lake and migrate to deeper demersal environments. This behavior is in line with what has been observed by Ayyildiz & Altin (2021).

Regarding the genus *Diplodus*, almost all individuals were observed near the entrance to the canals, probably to exploit the connection between the lake and the marine environment. As reported by Abecasis et al. (2009) and Vinagre et al. (2010) species of the genus *Diplodus* showed a marked seasonality in their movements, with coastal lagoons utilized as nursery areas. Subsequently, adult individuals tended to migrate to adjacent marine waters using canals as ecological corridors to maintain connectivity between coastal and marine habitats.

The species *Sarpa salpa* was the species with the largest number of individuals surveyed. However, the annual fluctuations in observed abundance could be attributed to the species tendency to move to shallower waters during cold seasons (Kellnreitner et al., 2012), making itself less visible and difficult to detect through the visual census method. In conclusion, fish

belonging to the Sparidae family may use the Faro Lake as a transit area for feeding, refuge or reproduction rather than as a stable habitat. This could explain the low number of individuals compared to their specific diversity.

4.6 Effectiveness of Visual Census and BRUVs in Faro Lake

The visual census and BRUVs techniques have been applied over the years in different habitats, showing different results that have allowed the determination of which technique is more effective in relation to the characteristics of the monitored environment. In the comparison made between the two techniques by Lowry et al. (2012), applied on artificial reefs in three coastal estuaries, visual census was found to be more effective than BRUVs in analyzing species richness. The techniques were also applied on nearshore rocky reefs and shallow rocky habitats respectively by Colton & Swearer (2010) e Stobart et al. (2007), also recording more species through the use of the visual census technique. Another study showed that the application of only one technique between visual census and BRUVs can cause an underestimation of species richness (Cheal et al., 2021). On the contrary, several studies (Ghazilou et al., 2019; La Manna et al., 2021) found that BRUVs were the technique that detected the most species.

Although the techniques have been applied separately in environments such as coastal lakes (Franco et al., 2012; Mateos-Molina et al., 2024), the combined application of the two methodologies in these ecosystems is still poorly documented.

Focusing on the combined application of the two techniques in Faro Lake, data showed that the Visual Census consistently detected a greater species richness across all seasons. Notably, all species recorded by the BRUVs were also identified through the Visual Census. This indicates that, in the context of Faro Lake, the integration of the two methods did not reveal additional species beyond those detected by the Visual Census alone, diverging from the findings reported by other authors in similar studies Franco et al. (2012); Mateos-Molina et al. (2024). The greater success of the Visual Census technique could be attributed to the fact that the operator, by covering predetermined transects, is able to monitor a larger area than BRUVs. The BRUVs technique, in fact, is based on attracting fish through bait. Although the locations for positioning the instrument were chosen on the basis of the greater presence of species and individuals, effectiveness could be limited if the dispersion of the bait smell is influenced by factors such as hydrodynamic flows in the area. Faro Lake receives direct and periodic input of Ionian Sea water mainly from Faro Canal (Ferrarin et. al, 2013). This current, influenced by wind and tides, could determine the dispersion of the bait smell. It has also been observed that the ecology and behavior of certain species may affect the detection frequency of the two methods (Lowry et al., 2012). Regarding the results obtained using the BRUVs technique applied in the Faro

Lake, the lower species richness detected could have been influenced by the characteristics of the species. In relation to what was observed, it can be stated that, within Lake Faro, the visual census technique proved to be the most effective, being particularly suited to the specific ecological conditions of this environment. Furthermore, BRUVs appear to be particularly attractive only to certain species that are strongly drawn to odors. Additionally, cryptobenthic species, such as blennies and gobies, while they may be attracted, could remain undetected during video observation due to their cryptobenthic habits and small sizes.

4.6.1 Species accumulation curve using BRUVs

Considering the lower efficiency of the BRUVS method compared to the visual census for assessing teleost species richness in Faro Lake, the species accumulation curve (Fig. 35) was considered a valuable addition, as this approach is similar to those widely applied in other contexts (Colton & Swearer, 2010; Skinner et al., 2020; Unsworth et al., 2014) and, more recently, in coastal areas in southern Italy (Grödl, 2023). It was used to analyze the first appearance time of each species across all surveys, aiming to understand the dynamics of interaction with the bait in this environment and to explore potential behavioral patterns within the various teleost assemblages. From this analysis, it was observed that the first species to interact with the bait were *A. hepsetus*, *G. niger*, *G. cobitis*, *G. incognitus*, *C. auratus*, and *C. labrosus*. Subsequently, another group of species appeared, including *B. boops*, *S. salpa*, *L. mormyrus*, members of the Clupeidae family, and *C. crysos*. A third group, consisting of *M. cephalus*, *P. sanguinolentus*, *Trachurus sp.*, *C. chromis*, *T. pavo*, and *O. melanura*, followed. Finally, *S. aurata* was the last species to interact with the bait, which could imply a more cautious approach or a preference for different foraging conditions. This sequential interaction not only highlights the varying degrees of attraction to the bait but also provides insights into the temporal dynamics of species interactions in the ecosystem. Overall, the application of the species accumulation curve serves as a valuable tool for assessing both the interaction with the bait and the effort required in terms of time to achieve the maximum number of species observed. This method enhances our understanding of species behavior and community structure, offering a more nuanced perspective on the ecological dynamics at play in Faro Lake.

4.7 Changes in teleost assemblages in Faro Lake throughout the time

As mentioned above, several authors have provided information about the fish species that inhabited and still inhabit Lake Faro. One of the first testimonies on the fish species of Lake Faro was provided by Leonardo Grassi (1903), who in 1903 described 36 species present and distinguished them into two categories: those adapted to live permanently in the lake and those found there occasionally. Regarding the first category: *Anguilla vulgaris* (*Anguilla anguilla* Linnaeus, 1758), *Mugil chelo* (*Chelon labrosus* Risso, 1827), *Mugil cephalus* Linnaeus, 1758, *Mugil auratus* (*Chelon auratus* Risso, 1810), *Atherina boyeri* (Risso, 1810), *Belone acus* (*Belone belone* Linnaeus, 1760), *Calyonimus dracunculus* (*Callionymus lyra* Linnaeus, 1758), *Mullus barbatus* Linnaeus, 1758*, Mullus surmuletus* Linnaeus, 1758, *Gobius ater* Bellotti, 1888, *Gobius jozo* (*Gobius niger* Linnaeus, 1758), *Gobius geniporus* Valenciennes, 1837, *Blennius pavo* (*Salaria pavo* Risso, 1810), *Sargus rondeleti* (*Diplodus sargus* Linnaeus, 1758), *Coris gioffredi* (*Coris julis* Linnaeus, 1758), *Clupea pilchardus* (*Sardina pilchardus* Walbaum, 1792), *Sardinella aurita* Valenciennes, 1847, *Trachurus trachurus* Linnaeus, 1758, *Trachurus melanosaurus* (*Trachurus picturatus* Bowdich, 1825), *Box boops* (*Boops boops* Linnaeus, 1758), *Pagellus erithrinus* (*Pagellus erythrinus* Linnaeus, 1758), *Pagellus mormyrus* (*Lithognathus mormyrus* Linnaeus, 1758), *Oblata melanura* (*Oblada melanurus* Linnaeus, 1758), *Maena vulgaris* (*Spicara maena* Linnaeus, 1758) and *Engraulis encrasicholus* (*Engraulis encrasicolus* Linnaeus, 1758); for the second category: *Serranus scriba* Linnaeus, 1758, *Seriola dumerilii* (*Seriola dumerili* Risso, 1810), *Corvina nigra* (*Sciaena umbra* Linnaeus, 1758), *Epinephelus gigas* (*Epinephelus marginatus* Lowe, 1834), *Epinephelus canina* (*Epinephelus caninus* Valenciennes, 1843), *Balistes capriscus* Gmelin, 1789, *Anthias sacer* (*Anthias anthias* Linnaeus, 1758), *Labrax lupus* (*Dicentrarchus labrax* Linnaeus, 1758), *Pagellus acarne* (Risso, 1827), *Cantharus vulgaris* (*Spondyliosoma cantharus* Linnaeus, 1758), *Lichia glauca* (*Trachinotus ovatus* Linnaeus, 1758) and *Amphioxus lanceolatus* (*Branchiostoma lanceolatum* Pallas, 1774). Other works on the fish community in the Messina lakes include contributions by Lo Giudice (1912, 1940), in which species were also observed *Heliastes chromis* (*Chromis chromis* Linnaeus, 1758) and *Muraena helena,* Linnaeus, 1758, D'Ancona (1934) who has taken an interest in several species of the genus *Syngnathus*, Spartà (1934, 1936, 1936, 1948, 1948, 1950) on species of the Gobiidae family, Dulzetto (1947, 1948, 1962) with insights to *Engraulis encrasicolus*. Cavallaro and Ilacqua (1972) following a year-long study conducted in Lake Faro, identified and documented the presence of 17 fish species. Cavaliere (1967), conducted

the first real systematic census of fish species in the Faro Lake and Ganzirri, documenting a total of 98 species. Many of the species documented by Cavaliere in 1967 were not recorded in subsequent censuses, including the present study. According to Potoschi et al. (2002) this could indicate greater species diversity in the past and possible habitat deterioration over time. Cavaliere also enhanced the opening of the English Canal as an event that affected species richness, highlighting a similarity with the dynamics observed in the present study, although statistical analysis did not return significant results.

The research by Potoschi et al. (2002) represents the last detailed census conducted in both Faro Lake and Ganzirri Lake before the present study. This study identified a total of 46 species, 43 of which were found in the Faro Lake and 24 in the Ganzirri Lake, with 21 species common to both lakes. The three species found only in Lake Ganzirri are *Serranus scriba, Caranx rhonchus* and *Alosa fallax.* Considering only the data for Faro Lake, the comparison of the results of the study by Potoschi et al. (2002) and the present monitoring revealed 38 species in common. However, five species documented by Potoschi et al. (2002) were not detected in the present study: *Gobius cruentatus, Pomatoschistus panizzai* (currently *Knipowitschia panizzae*), *Zosterisessor ophiocephalus*, *Anguilla anguilla and Pagellus acarne*. On the other hand, 12 species were found in the present study that did not appear in Potoschi's list. These include: *Aydablennius sphynx*, *Boops boops*, *Caranx rhonchus, Coris julis, Coryphoblennius galerita, Epinephelus marginatus, Epinephelus costae, Mycteroperca rubra, Pomatoschistus minutus, Serranus scriba, Symphodus ocellatus* and *Thalassoma pavo*. Some species, such as *Trachurus trachurus* and *Trachurus mediterraneus, Syngnathus abaster* and *Alosa fallax* were distinguished by Potoschi through direct observation of the sampled fish. However, due to the elusive nature of these species during the visual census, their correct identification was not possible in the present study. Consequently, they were classified in taxon as *Trachurus* sp., *Syngnathus* sp., and Alosidae, respectively. The variation in the composition of the teleost community between the present census and Potoschi et al. (2002) research highlight a different effectiveness of the methods used. In the present study, the use of non-invasive techniques in transitional environments such as Faro Lake produced better results in terms of species richness respect to invasive methods, with 52 species detected compared to the 43 previously observed. Moreover, such differences may suggest the existence of ecological dynamics still to be explored. A future investigation could analyze the shifts in community composition relatively to different functional trophic categories, providing valuable insights for further research.

Table 26; Teleost species recorded in Faro Lake from 1903 to Current Study (CS) and their assigned ecological categories.

Taxa	1903	1967	2002	CS
Aidablennius sphynx (Valenciennes, 1836)				OV
Alosa alosa (Linnaeus, 1758)		SM		
Alosidae				SM
Anguilla anguilla (Linnaeus, 1758)	SS	SM	SM	
Anthias anthias (Linnaeus, 1758)	OV			
Aphanius fasciatus (Valenciennes, 1821)			SS	SS
Apogon imberis (Linnaeus, 1758)			SS	OV
Apterichtus caecus (Linnaeus, 1758)		OV		
Ariosoma balearicum (Delaroche, 1809)		OV		
Arnoglossus grohmanni (Bonaparte, 1837)		OV		
Atherina boyeri (Risso, 1810)	SS	SM		
Atherina hepsetus (Linnaeus, 1758)			SS	SS
Balistes capriscus (Gmelin, 1789)	OV	SM	OV	OV
Belone belone (Linnaeus, 1760)	SS	SM	SM	SM
Blennius ocellaris Linnaeus, 1758		SM		
Boops boops (Linnaeus, 1758)	SS	SM		SM
Bothus podas (Delaroche, 1809)		SM		
Branchiostoma lanceolatum (Pallas, 1774)	OV			
Buglossidium luteum (Risso, 1810)		SM		
Callionymus lyra (Linnaeus, 1758)	SS			
Campogramma glaycos (Lacepède, 1801)		OV		
Caranx crysos (Mitchill, 1815)			OV	SM
Caranx rhonchus (Geoffroy Saint-Hilaire, 1817)				SM
Chelon auratus (Risso, 1810)	SS	SM	SS	SS
Chelon labrosus (Risso, 1827)	SS	SM	SS	SS
Chelon ramada (Risso, 1827)		SM	SS	SM
Chromis chromis (Linnaeus, 1758)			OV	SS
Conger conger (Linnaeus, 1758)		SM		
Coris julis (Linnaeus, 1758)	SS	SM		OV
Coryphoblennius galerita (Linnaeus, 1758)				SM
Dactylopterus OVlitans (Linnaeus, 1758)		OV		
Dalophis imberbis (Delaroche, 1809)		OV		
Dentex dentex (Linnaeus, 1758)		SM		
Dicentrarchus labrax (Linnaeus, 1758)	OV	SM	SM	OV
Dicentrarchus punctatus (Bloch, 1792)		SM		
Diplodus annularis (Linnaeus, 1758)		SM	SM	SM
Diplodus puntazzo (Walbaum, 1792)		OV	SM	SM
Diplodus sargus (Linnaeus, 1758)	SS	SM	SM	SM
Diplodus vulgaris (Geoffroy Saint-Hilaire, 1817)		OV	SM	SM

Echiichthys vipera (Cuvier, 1829)		OV		
Engraulis encrasicolus (Linnaeus, 1758)	SS	SS	SM	SM
Epinephelus aeneus (Geoffroy Saint-Hilaire, 1817)			OV	SS
Epinephelus caninus (Valenciennes, 1843)	OV	SM		
Epinephelus costae (Steindachner, 1878)				SM
Epinephelus marginatus (Lowe, 1834)	OV	SM		SS
Exocoetus volitans (Linnaeus, 1758)		OV		
Gobius ater (Bellotti, 1888)	SS			
Gobius cobitis (Pallas, 1814)		SS	SS	SS
Gobius cruentatus (Gmelin, 1789)			SS	
Gobius geniporus (Valenciennes, 1837)	SS			
Gobius incognitus (Kovačić & Šanda, 2016)				SS
Gobius niger (Linnaeus, 1758)	SS	SS	SS	SS
Gobius paganellus (Linnaeus, 1758)		SS		SS
Hippocampus guttulatus (Cuvier, 1829)		SS		
Hippocampus hippocampus (Linnaeus, 1758)		SS		
Hirundichthys rondeletii (Valenciennes, 1847)		OV		
Knipowitschia panizzae (Verga, 1841)			SS	
Labrus mixtus (Linnaeus, 1758)		OV		
Labrus viridis (Linnaeus, 1758)		SM		
Lepadogaster candolii (Risso, 1810)		SS		
Lepadogaster lepadogaster (Bonnaterre, 1788)		SS		
Lepidotrigla cavillone (Lacepède, 1801)		OV		
Lichia amia (Linnaeus, 1758)		OV		
Lithognathus mormyrus (Linnaeus, 1758)	SS	SM	SM	SM
Microchirus ocellatus (Linnaeus, 1758)		OV		
Mugil cephalus (Linnaeus, 1758)	SS	SM	SS	SS
Mullus barbatus (Linnaeus, 1758)	SS	SM	SM	SM
Mullus surmuletus (Linnaeus, 1758)	SS	SM	SM	SM
Muraena helena Linnaeus, 1758		SM		OV
Mycteroperca rubra (Bloch, 1793)				SS
Nerophis maculatus (Rafinesque, 1810)		SS		
Nerophis ophidion (Linnaeus, 1758)		SS		
Oblada melanurus (Linnaeus, 1758)	SS	SM	SM	SM
Oedalechilus labeo (Cuvier, 1829)		OV	SS	SM
Ophidion barbatum (Linnaeus, 1758)		OV		
Pagellus acarne (Risso, 1827)	OV	OV	OV	
Pagellus erythrinus (Linnaeus, 1758)	SS	OV		
Pagrus pagrus (Linnaeus, 1758)		OV		
Parablennius gattorugine (Linnaeus, 1758)		SM		
Parablennius sanguinolentus (Pallas, 1814)		SS	SS	SS
Pomatoschistus minutus (Pallas, 1770)				SM

Salaria pavo (Risso, 1810)	SS	SS	SS	SS
Sardina pilchardus (Walbaum, 1792)	SS	OV	OV	
Sardinella aurita (Valenciennes, 1847)	SS	SM	SM	SM
Sarpa salpa (Linnaeus, 1758)		SM	SS	SS
Schedophilus medusophagus (Cocco, 1839)			OV	
Sciaena umbra (Linnaeus, 1758)	OV	OV		
Scophthalmus maximus (Linnaeus, 1758)		OV		
Scorpaena notata (Rafinesque, 1810)		OV		
Scorpaena porcus (Linnaeus, 1758)		SM	SM	OV
Scorpaena scrofa (Linnaeus, 1758)		SM		
Seriola dumerili (Risso, 1810)	OV	OV		
Serranus cabrilla (Linnaeus, 1758)		OV		
Serranus scriba (Linnaeus, 1758)	OV	OV		OV
Sparisoma cretense (Linnaeus, 1758)		OV		
Sparus aurata (Linnaeus, 1758)		SM	SS	SM
Sphyraena sphyraena (Linnaeus, 1758)		OV		
Spicara maena (Linnaeus, 1758)	SS	OV		
Spicara smaris (Linnaeus, 1758)		OV		
Spondyliosoma cantharus (Linnaeus, 1758)	OV	SM		
Symphodus mediterraneus (Linnaeus, 1758)		SM		
Symphodus ocellatus (Linnaeus, 1758)		SM		SM
Symphodus roissali (Risso, 1810)			OV	SM
Symphodus tinca (Linnaeus, 1758)		SM		
Syngnathus abaster (Risso, 1827)		SS	SS	
Syngnathus acus (Linnaeus, 1758)		SS		
Syngnathus phlegon (Risso, 1827)		SS		
Syngnathus typhle (Linnaeus, 1758)		SS		
Syngnatus sp.				SS
Synodus saurus (Linnaeus, 1758)		OV		
Thalassoma pavo (Linnaeus, 1758)		SM		SS
Trachinotus ovatus (Linnaeus, 1758)	OV	OV	OV	SM
Trachinus araneus (Cuvier, 1829)		OV		
Trachinus radiatus (Cuvier, 1829)		OV		
Trachurus mediterraneus (Steindachner, 1868)		OV	OV	
Trachurus picturatus (Bowdich, 1825)	SS			
Trachurus sp.				SM
Trachurus trachurus (Linnaeus, 1758)	SS	OV	OV	
Trigla lyra (Linnaeus, 1758)		OV		
Umbrina cirrosa (Linnaeus, 1758)		OV		
Uranoscopus scaber (Linnaeus, 1758)		OV		
Zebrus zebrus (Risso, 1827)		SS		
Zosterisessor ophiocephalus (Pallas, 1814)			SS	

5 CONCLUSIONS

This study represents the first application of an integrated approach based on non-invasive techniques for monitoring the fish community in Faro Lake, revealing significant correlations between environmental factors and the dynamics of the teleost community, an aspect not previously investigated. The techniques employed proved to be highly effective, allowing for the collection of detailed and representative data on the fish community without compromising the ecosystem's integrity. Compared to traditional methods, the visual census detected more species than the last study conducted in 2002. The results provide a solid basis to further investigate these dynamics. In particular, this technique adapted well to the morphological and structural characteristics of the lake, enabling the analysis of trends in species richness and abundance, as well as the determination of their functional ecological categories, thus establishing itself as the most effective methodology in this context. The collection of environmental parameters and their comparison with these trends allowed for the identification of some of the key dynamics characterizing the teleost community. Moreover, the study highlights the effectiveness of integrated approaches based on non-invasive techniques, which have proven to be highly efficient and perfectly aligned with sustainability criteria. This integrated approach not only demonstrates the efficacy of non-invasive methods but also underscores their potential application in conservation contexts such as Natural Oriented Reserves (NOR), Sites of Community Interest (SCI), and Special Protected Areas (SPA). In these areas, the adoption of these techniques is currently a priority choice. These approaches help to preserve the integrity of the monitored habitats, minimizing anthropogenic impacts and thus providing a useful knowledge base for the development of effective strategies for management and conservation purposes. In conclusion, this study not only establishes a foundational understanding of the fish community dynamics in Faro Lake but also advocates for the broader adoption of non-invasive monitoring techniques as essential tools for sustainable management and conservation efforts in sensitive aquatic ecosystems.

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