THE EFFECTS OF MARINE PROTECTED AREAS ON THE ECOSYSTEM IN THE EASTERN MEDITERRANEAN

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ABSTRACT

THE EFFECTS OF MARINE PROTECTED AREAS ON THE ECOSYSTEM IN THE EASTERN MEDITERRANEAN

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Marine Protected Areas (MPAs) are primary tools for restoring degraded marine ecosystems affected by various stressors. Although numerous studies have acknowledged and demonstrated their effectiveness, their effects and efficacy in areas under intense pressure from non-indigenous species (NIS), such as the Eastern Mediterranean, have been debated. Therefore, filling this gap by investigating their effects is important for properly establishing MPAs and implementing effective management plans.

Within this scope, the effects of protection measures on the ecosystem in a region that was degraded at the end of the 1980s due to the intense fishing pressure were investigated. The study primarily focuses on a time series from 1983 to 2023, reflecting the status of commercially exploitable adult fish populations before and after the establishment of the protected area. Data on biomass and species numbers collected from demersal trawl surveys both within and outside the partially protected Fisheries Restricted Area (FRA) were analyzed to assess the impact of conservation measures on fish populations and NIS. The second focus was on juvenile and young-of-the-year fishes within the fully protected No-Take Zone (NTZ), spanning two distinct time periods: (i) the initial years of conservation efforts and (ii) 20 years after their implementation. Observations recorded through underwater visual census in vulnerable nursery areas within the shallow infralittoral zone, inside and outside the NTZ, were analyzed to evaluate changes in biodiversity over time and the effects of conservation measures on NIS.

It was determined that fishing restrictions within the FRA positively affected the exploitable fish stocks, with biomass values showing a recovery over the years. Additionally, although NIS have increased both within and outside the FRA, their biomass has been found to be lower within the FRA, indicating that protection measures have slowed the spread of NIS. These results highlighted the crucial role of conservation strategies in mitigating the NIS and restoring the ecosystem. On the other hand, the NTZ exhibited different results. High indigenous species (IS) diversity was initially observed within the NTZ in the early years of the implemented protection measures. However, observations conducted 20 years later indicated a shift: a decrease in IS diversity was observed, and an increase in the diversity and success (which was estimated based on species richness and evenness) of NIS was noted. In contrast, IS diversity outside the NTZ maintained its status throughout the two periods, and NIS did not show success as they showed within the NTZ. Their low diversity and success are attributed to small-scale fishing activities, which targeted these NIS, thereby controlling their populations. However, without such control within the NTZ, they benefited from the conservation measures and gained an advantage. Additionally, (i) the disappearance of IS within the NTZ over the years, which shared the infralittoral zone and (ii) the reduction in success and diversity of the IS were attributed to the disruption IS' habitat dynamics caused by the introduction of NIS to the ecosystem. These contrasting results between the FRA and NTZ indicate that the impact of conservation efforts on the NIS may vary depending on the habitat structure being protected, the structure of fish groups within the protected area, and the level of protection measures implemented. These findings also may suggest that fisheries may be utilized to mitigate the impacts of NIS in the areas facing high NIS pressure. Additionally, when establishing MPAs in NIS-impacted ecosystems such as the Eastern Mediterranean, with the only aim of enhancing biodiversity or rebuilding fish stocks, it is crucial to consider the effects of NIS in the planning process to ensure the effectiveness of these conservation efforts.

Keywords: No-Take Zone (NTZ), Fisheries Restricted Areas (FRAs), Non-Indigenous Species, Demersal Trawl, Visual Census, Biodiversity

ÖZ

DENİZ KORUMA ALANLARININ DOĞU AKDENİZ'DE EKOSİSTEME OLAN ETKİLERİ

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Deniz Koruma Alanları (DKA), çeşitli stres faktörlerinden etkilenen bozulmuş deniz ekosistemlerini onarmak için en sık kullanılan araçlardır. Birçok çalışma sayesinde DKA'ların etkinliği kabul edilmiş ve kanıtlanmış olmasına rağmen Doğu Akdeniz gibi yabancı türlerin baskısı altındaki alanlarda ne kadar etkili olabildikleri hâlen tartışma konusu olmaktadır. Dolayısıyla, DKA'ların bu tür alanlardaki etkinliğini anlamak ve DKA'ların doğru ve etkin bir şekilde kurulup yönetilmesini sağlamak için var olan bu boşluğun doldurulması büyük önem taşımaktadır.

Bu kapsamda, 1980'lerin sonunda yoğun balıkçılık baskısı nedeniyle bozulan bir bölgedeki koruma önlemlerinin ekosistem üzerindeki etkileri araştırılmıştır. Sunulan çalışmada öncelikle 1983'ten 2023'e kadar uzanan ve ticari olarak avlanan yetişkin balık popülasyonlarının koruma alanının kurulmasından önceki ve sonraki durumunu yansıtan bir zaman serisine odaklanılmıştır. Kısmen korunan Balıkçılık Kısıtlı Alanı'nın (BKA) sınırları içinde ve dışında dip trolü kullanılarak yürütülen araştırmalardan toplanan biyokütle ve tür sayıları verileri, koruma önlemlerinin balık popülasyonları ve yabancı türler üzerindeki etkilerini değerlendirmek için analiz edilmiştir. Çalışmanın ikinci odak noktasında ise koruma önlemlerinin uygulanmaya başlandığı ilk yıllar ve 20 yıl sonrası olmak üzere iki farklı zaman dilimini yansıtan Tam Koruma Alanı (NTZ) içindeki yavru ve genç balıklara odaklanılmıştır. NTZ sınırları içinde ve dışında, sığ infralittoral bölgede bulunan, yavru balıklar için önemli olan hassas alanlarda sualtı görsel sayım yöntemiyle kaydedilen gözlemler, biyoçeşitlilikte zaman içinde meydana gelen değişimleri ve koruma önlemlerinin yabancı türler üzerindeki etkilerini değerlendirmek için analiz edilmiştir.

BKA içindeki balıkçılık kısıtlamalarının, ticari olarak hedeflenen balık stoklarını olumlu bir şekilde etkilediği ve balık popülasyonlarının biyokütle değerlerinin yıllar içinde iyileştiği görülmüştür. Yabancı türlerin biyokütlesinde hem BKA içinde hem de dışında bir artış gözlenmesine rağmen, biyokütleleri BKA içinde daha düşük bulunmuştur, bu da koruma önlemlerinin yabancı türlerin yayılmasını yavaşlattığını göstermektedir. Bu sonuçlar, koruma önlemlerinin yabancı türlerin yayılmasını azaltmada ve ekosistemleri onarmadaki önemli rolünü açıkça göstermiştir. Öte yandan, NTZ içinde yapılan analizler daha farklı sonuçlar sergilemiştir. Koruma önlemlerinin uygulandığı ilk yıllarda NTZ içinde yerli tür çeşitliliğinin korunmayan alanlara kıyasla daha yüksek olduğu gözlemlenmiştir. Ancak, 20 yıl sonra NTZ içinde yapılan gözlemlerde tam tersi bir durum gözlemlenmiştir. Yerli türlerin çeşitliliğinde bir azalma meydana gelmiş, yabancı türlerin çeşitliliği ve başarısı (tür zenginliği ve eşitliğine dayanarak tahmin edilen değer) artmıştır. NTZ dışındaki alanlarda ise yerli türler iki zaman dilimi boyunca değişmeden kalmış ve yabancı türler NTZ içinde gösterdikleri kadar başarı göstermemiştir. NTZ dışında gözlemlenen düşük yabancı tür çeşitliliği ve başarısı, bu alanlarda yabancı türleri hedef alan ve böylece popülasyonlarını kontrol eden küçük ölçekli balıkçılık faaliyetlerine atfedilmiştir. NTZ içinde balıkçılığa bağlı bu tür bir kontrol olmaksızın, yabancı türlerin koruma önlemlerinden faydalanarak avantaj kazandığı sonucuna varılmıştır. Ayrıca, NTZ içindeki infralittoral bölgeyi birbiri ile paylaşan bazı yerli türlerin zamanla kaybolması, başarılarının ve çeşitliliklerinin azalması, yabancı türlerin ekosisteme dahil olduktan sonra yerli türlerin habitat dinamiklerini bozmasına atfedilmiştir. BKA ve NTZ arasındaki bu zıt sonuçlar, koruma çabalarının yabancı türler üzerindeki etkisinin, korunan habitatın yapısına, balık gruplarının yapısına ve uygulanan koruma önlemlerinin seviyesine bağlı olarak değişebileceğini göstermektedir. Bu bulgular ayrıca, yüksek yabancı tür baskısına maruz kalan alanlarda yabancı türlerin etkilerini azaltmak ve yavaşlatmak için balıkçılığın bir araç olarak kullanılabileceğini de düşündürmektedir. Ek olarak, biyolojik çeşitliliği artırma veya balık stoklarını onarma amacıyla, Doğu Akdeniz gibi yabancı türlerden etkilenen ekosistemlerde DKA'lar kurarken, bu koruma çabalarının etkinliğini sağlamak için planlama sürecinde yabancı türlerin etkisinin de göz önüne alınması önemlidir.

Anahtar Kelimeler: Tam Korunan Alan, Balıkçılığa Kısıtlı Alanlar, Yabancı Türler, Dip Trolü, Görsel Sayım, Biyoçeşitlilik

To all the women ancestors whose legacy I carry within me

&

To my parents and grandparents

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GENERAL INTRODUCTION

Fishing activities, pollution, aquaculture, land use changes, water utilization and shipping collectively impact and alter marine ecosystems and their functions globally (Lubchenco et al., 1995). These activities lead to significant alterations in the base of the ecosystem structure, including variations in biodiversity, population sizes, size distribution, behaviors, habitat configurations, biological interactions, trophic dynamics and more (Lubchenco et al., 2003). As attention rises to the importance of marine ecosystems, it becomes essential to develop effective strategies to prevent their deterioration and provide the protection and restoration of marine ecosystems. In this context, one of the most used strategies is the establishment of Marine Protected Areas (MPAs).

From a historical perspective, preserving marine areas has been a longstanding practice for centuries. These measures have commonly served as a means to facilitate the recovery of fish stocks, typically commencing with the closure of specific areas to fishing by a community or political authority (Johannes, 1978). Over time, with the increasing recognition of the value of wildlife, ‘protected areas’ to protect estuaries or other intertidal habitats began to be established in the modern sense. Certain locations were explicitly safeguarded for their intertidal and marine resources. For instance, in Alaska, two areas were designated as protected areas in 1869, primarily for northern fur seal conservation (Scheffer et al., 1984). Similarly, in Jamaica, the Morant and Pedro Cays Act, established in 1907 for fisheries management, also encompassed protection for seabirds and turtles in the Morant and Pedro Cays (Yugorsky & Sutton, 2004). Additionally, Beverton and Holt (1957) confirmed that MPAs began to be established primarily within fisheries management in a broader and more global context. These might be considered the earliest examples of the implementation of the conservation of marine nature and sustainable fisheries.

As the recognition and acceptance of MPAs grew locally, their acknowledgment began to spread globally. An example is the International Conference on Marine Parks and Reserves in Tokyo in 1975, organized by the International Union for Conservation of Nature (IUCN), the first global meeting focusing on MPAs (IUCN, 1976; Wells et al., 2016). At this conference, awareness was drawn to the need to establish and manage MPAs, highlighting the challenges posed by anthropogenic pressures on marine environments.

Nowadays, in a modern sense, the definition of MPAs used by IUCN since 1999, revised and developed by The World Commission on Protected Areas framework in 2008 as follows: “A clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values” (Dudley, 2008). On the other hand, the broad definition that Food and Agriculture Organization (FAO) has made for MPAs is that “any marine geographical area that is afforded greater protection than the surrounding waters for biodiversity conservation or fisheries management purposes will be considered an MPA” (FAO, 2011). This general definition encompasses extensive regions (e.g., exclusive economic zones).

Nevertheless, the term MPA typically refers to designated areas aimed at safeguarding specific ecosystems, components of ecosystems, or other features such as historical sites. MPAs can range from small areas to large national parks. Also, the degree of protection provided in MPAs can vary, ranging from complete protection, where all activities are prohibited, to partial protection, where certain activities are allowed but subject to different restrictions (Lubchenco et al., 2016). The specific regulations associated with an MPA vary depending on the context, and terms are not consistently applied (FAO, 2011). For instance, the ‘reserve’ term can be used in one country to indicate prohibited fishing, while in another country, it is used to prohibit only destructive fishing (FAO, 2011). Various terms are used interchangeably, including fully protected marine areas, ocean sanctuaries, marine sanctuaries, marine parks, no-take zones, and Fisheries Restricted Areas (FRAs) (FAO, 2011). All of them aim to maintain the ecosystem's health. For instance, Fisheries Restricted Areas (FRAs) are geographically defined areas where specific fishing activities (e.g., purse-seines, bottom trawls, and other gear types) are regulated by permanently or temporarily restricting or banning to protect fish stocks, sensitive habitats, and biodiversity (FAO, 2018). Similarly, No-Take Zones are areas where all extractive activities (e.g., fishing, mining) are prohibited (Begg et al., 2005). In addition to these definitions, there are also other areas that encompass marine areas under "other effective area-based conservation measures (OECMs)" applied. These areas are defined as "a geographically defined area other than a Protected Area, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the in-situ conservation of biodiversity, with associated ecosystem functions and services and, where applicable, cultural, spiritual, socioeconomic, and other locally relevant values" (CBD, 2018).

A list of examples of objectives of MPAs are listed below (Roberts et al., 2003; FAO, 2011):

- Providing a recovery of fish stocks by banning fishing on stocks that have collapsed or are near collapse
- Ensuring the sustainability of fisheries and fish stocks (spillover of an exploited species)
- Protecting fragile habitats and marine biodiversity (e.g., by closing areas, permanently or temporally, where discard and bycatch rates are high and conservation of areas essential for vulnerable life stages, such as nursery grounds)
- Increasing ecosystem resilience (the capability and stability of ecosystems to tolerate stress factors) to climate change or other environmental changes

- Supporting traditional and local sustainable marine-based lifestyles and communities
- Education and research

IUCN describes six categories of protected areas according to their management objectives (Dudley, 2008).

Category Ia - Strict Nature Reserve: Strictly protected areas (PAs) to protect biodiversity and geological/geomorphological features. These areas can serve as reference areas for scientific research and monitoring.

Category Ib - Wilderness Area: PAs typically encompass large regions that remain unaltered or slightly modified, maintaining their natural characteristics. These areas do not have permanent or significant human residency and are safeguarded and managed to ensure the preservation of their natural state.

Category II - National Park: PAs are extensive, natural, or nearly natural areas reserved to safeguard large-scale ecological processes and provide a basis for environmental and cultural values. Within these areas, human activities are permitted to a certain extent.

Category III - Natural Monument: PAs designated for conserving a particular natural monument. These areas are typically small-sized and attract a significant number of visitors.

Category IV - Habitat/Species Management Area: PAs primarily aim to conserve specific species or habitats, and their management is based on this priority.

Category V - Protected Landscape/Seascape: PAs shaped by the interplay between humans and nature over time, resulting in a distinct character with significant ecological, biological, cultural, and scenic value.

Category VI – Managed Resources Protected Areas: PAs maintain a predominantly natural state, with a portion dedicated to sustainable natural resource management.

The primary objective is to allow low-level non-industrial use of natural resources that aligns with nature conservation goals.

To complement the IUCN classifications, a guide was prepared by Grorud-Colvert et al. (2021). This guide is one of the most comprehensive research on the classification, planning, and interpretation of MPAs in recent years. This guide reviews MPAs from their initial phases and categorizes them based on the activities allowed within the area. According to this, the areas are assessed at four levels of protection, ranging from minimal to complete protection. The guide also highlights that if an MPA is to be accepted as effective, it must have efficient planning, management, and design, with its success depending on the stages, conditions, and levels of protection (Grorud-Colvert et al., 2021).

1.1 The Effectiveness of MPAs and Their Positive Outcomes in a Fisheries Context

The effectiveness of MPAs in achieving their objectives and providing positive outcomes depends on their proper establishment and management practices. Effective management should begin even before the establishment stage of the area. Certain criteria must be satisfied, such as clearly defined boundaries, clearly stated objectives, processes to describe permitted uses, related regulations or laws to control effects, and traditional recognition (Grorud-Colvert et al., 2021).

After these criteria are met, other factors also play a role. Effectiveness relies on various factors such as the location and size of MPAs and the type of protection implemented (e.g., complete fishing bans or selective restrictions). For instance, establishing an MPA where there are scarce species to protect provides proportionally less protective benefit (FAO, 2011). In terms of protection level, MPAs that restrict all human activities within their borders will provide greater protection outcomes than MPAs that are partially protected (FAO, 2011; Sala & Giakoumi, 2018). In addition, activities occurring outside the MPAs might also influence their effectiveness (FAO, 2011). Following these factors, another critical factor in influencing the effectiveness of the MPAs is the implementation of a monitoring system that tracks ecological changes within the MPAs, allowing managers to adapt and improve their current and future management for better protection outcomes.

The effectiveness of MPAs can become noticeable in many positive ways. From a fisheries perspective, well-managed MPAs restore habitats by increasing individual sizes and biomass of fish species, and these features exhibit higher values within MPA than outside (FAO, 2011; Sala & Giakoumi, 2018). This effect is particularly noticeable in species at higher trophic levels (FAO, 2011). Furthermore, effective conservation measures can also positively affect the area's species richness (Halpern, 2003; Lester et al., 2009). The number of species inside MPAs can increase by protection measures and have higher biodiversity compared to unprotected areas (Bell, 1983; Harmelin et al., 1995; Edgar & Barrett, 1999; Halpern, 2003). Due to these positive effects, ecosystem resilience increases, and therefore, MPAs can also successfully prevent the settlement and invasion of non-indigenous species (Ardura et al., 2016).

On the other hand, the impact of protection can be achieved not only within the boundaries of the MPAs but also outside it. Increased reproduction within an MPA can lead to increased recruitment in the population and spilling over species to adjacent outside areas of the MPA (Di Lorenzo et al., 2016). Therefore, the available fish biomass to the fishery can increase.

While these effects are widespread and expected, they can vary regionally, by level of protection, or between MPAs, revealing varied trends across different studies. These examples are discussed in more detail in Chapter 2 and Chapter 3.

1.2 Marine Protected Areas Worldwide and in the Mediterranean

MPA implementations are showing an increase globally. The Global Ocean is divided into areas within national jurisdiction (National Waters) and those in international waters, such as Areas Beyond National Jurisdiction (ABNJ). Since 2000, the coverage of MPAs has grown by more than 14% in National Waters (Maestro et al., 2019). This trend reflects the increasing use of MPAs by governments worldwide as a tool.

Additionally, their increasing use is also emerging in the initiatives applied by governments. Governments have acknowledged the crucial role of MPAs in preserving biodiversity and sustaining and supporting livelihoods by taking some initiatives to target better conservation strategies, and within this scope, the Convention on Biological Diversity's Aichi Target 11 aimed to protect at least 10% of the world's oceans and seas through MPAs by 2020 (CBD, 2010). However, Aichi Target 11 has only been partially achieved. While progress has been made, gaps and challenges remain in fully achieving the target globally (CBD, 2020). Currently, there are 18,868 MPAs covering almost 30 million km² globally, which is indicating only 8.35% of the ocean is protected (UNEP-WCMC & IUCN, 2024).

On the other hand, when looking at the MPAs in the Mediterranean region, three types of areas are considered MPAs according to the MAPAMED (MedPAN & SPA/RAC, 2021):

- MPAs with a national statute - Legally designated by a State
- Marine Natura 2000 sites - Restricted to EU Member States, also have varying levels of protection
- The Pelagos Sanctuary - The only MPA with an international statute covering areas in France, Italy, and Monaco through an international agreement

Additionally, other effective area-based conservation areas exist, including Fisheries Restricted Areas and Particularly Sensitive Sea Areas. By the end of 2019, MPAs in the Mediterranean included 1,087 sites, covering approximately 209,303 km², which accounts for about 8.3% of the Mediterranean Sea (MedPAN & UNEP/MAP-SPA/RAC, 2023). According to information provided by World Wide Fund for Nature (WWF) report (Gomei et al., 2019), 9.68% of the Mediterranean has been designated as MPAs. However, only 2.48% of these areas have a management plan, and 1.3% have not effectively implemented their management plans.

Despite this, there are MPAs in the Mediterranean with success stories (MedPAN, 2024). Some of the most successful MPAs in the Mediterranean include Côte Agathoise MPA (France) and Gökova Bay MPA (Türkiye) (MedPAN, 2024). These areas have been well-managed, sustaining resources and the environment while increasing economic benefits to local stakeholders, and conducting collaborative work with small-scale fishers. Additionally, other MPAs exist with success stories, such as the Larvotto MPA (Monaco), Gyaros MPA (Greece), and the six Marine Reserves of Mallorca Island (Spain) (MedPAN, 2024).

All areas in the Mediterranean established with the aim of protecting marine biodiversity under any protection status, according to the database of Marine Protected Areas in the Mediterranean (MAPAMED), are shown in Figure 1. 1.

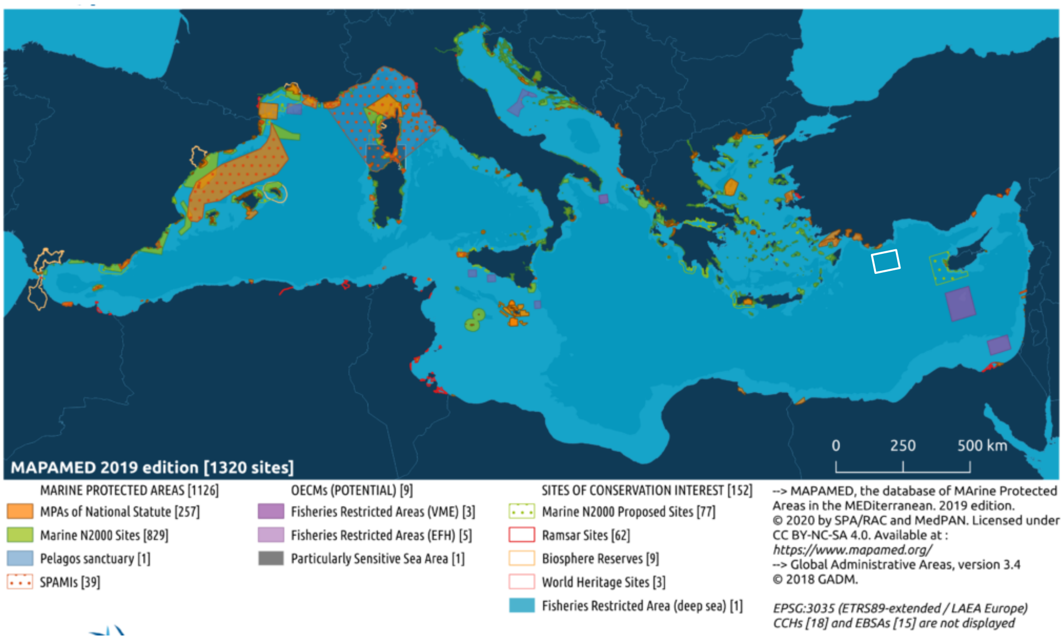

Figure 1. 1 All marine areas designated with any protection status for conserving marine biodiversity in the Mediterranean (MAPAMED 2019 edition, WestMED, 2021). The Finike Submarine Mountains, not marked on the original MAPAMED map but recognized under SEPA (Special Environmental Protected Area) status in Türkiye, have also been included in this map and indicated with a white square (the Republic of Türkiye Official Gazette, 2013).

Focusing on conservation efforts in Türkiye, which hosts one of the successful MPA examples in the Mediterranean (e.g., Gökova MPA), the legal protection categories applied in protected areas differ from IUCN definitions. Nonetheless, there are similar hierarchical levels of restrictions. The spatial protection measures enforced in Türkiye with respect to the level of protection are as follows (the Republic of Türkiye Ministry of Culture and Tourism, 2007):

- Special Environmental Protected Areas (SEPAs)
- Natural Sites
- 1st Degree Natural Sites: These are the areas that are preserved due to their interesting characteristics, scientific values, rare features and are to be maintained for the public interest. Human activities that may damage these

areas' geological features, vegetation, and landscape are completely prohibited, except for scientific research.

- 2nd Degree Natural Sites: These are areas that may open for public interest use only by limited means, providing such usage protects and enhances their natural structure. Except for those built to support the development of tourism, buildings on such sites are prohibited.
- 3rd Degree Natural Sites: These are areas where residential development may be permitted, taking into consideration the use of the potentials and patterns of the region, providing it together protects and enhances the natural structure.

Looking at the history of protected areas in Türkiye, the efforts to protect the marine environment over the years are noticeable. Between 1988 and 1990, 9 MPAs were established as SEPAs, including Köyceğiz-Dalyan, Fethiye-Göcek, and Gökova in 1988, and Göksu Delta, Kekova, Patara, Foça, Datça-Bozburun, and Belek in 1990 (the Republic of Türkiye Ministry of Environment, Urbanization and Climate Change, 2024). In 2013, the Finike Submarine Mountains were established under the SEPAs status, which is the first protected area in Türkiye that includes exclusively marine areas (the Republic of Türkiye Official Gazette, 2013). Currently, there are 19 established SEPAs and 13 out of them include marine areas (the Republic of Türkiye Ministry of Environment, Urbanization and Climate Change, 2024).

There are also protected areas subject to restrictions for fishery regulations but not under the SEPAs directive in Türkiye. An example of this is the Kızılliman MPA, established in 1999 along the coast west of Mersin, which is the focus of this thesis (Figure 1. 2).

The Kızılliman area is an important area since it is the main feeding and breeding grounds of the Mediterranean monk seals. However, the construction of three ports in 1988, 1992, and 1997 (Department of Fisheries and Aquaculture

Services, 2004) caused a transition in the region from small-scale fishing to industrial fishing, causing damage to the area and monk seal population. Due to the continental shelf in this area being narrow, fishing activities, mainly targeting demersal and small pelagic species, were linked to this limited continental shelf. Following this, the region's fish stocks were depleted due to intense fishing pressure (Gücü & Erkan, 1999). Additionally, although only small-scale fishing is permitted to operate in the coastal strip where seagrasses and fish nursery areas are located in accordance with fishing regulations in Türkiye, the large vessels violated this regulation and the 3-mile limit, causing a further decline in fish stocks (Erkan & Gücü, 2005 TUBITAK Report). As a result of depleted food resources for the monk seals, they were forced to feed on trapped fish from gill nets (Salman et al., 2001; Gücü & Erkan, 2005). Therefore, the risk of getting entangled in fish nets and seeing them as a threat by the fishers increased (Gücü & Erkan, 2005). These consequences caused by fisheries have put monk seals in danger as they created difficulties in their breeding success and finding food sources, and also raise the risk of their intentional killings by local fishermen who rely on the same food source (Gücü & Erkan, 2005).

Accordingly, a Marine Protected Area encompassing small core zones was established in the Kızılliman area. The primary goal of protecting this area was to ensure the successful reproduction of monk seals by protecting their breeding caves in the area from human disturbances (sport fishing activities) and also to allow the exploited fish stocks to recover. To protect their reproduction, the position and the number of breeding caves used by the monk seals directed the selection of the location and the number of core zones (Gücü & Erkan, 2005). After that, these core zones were established as No-Take Zone (NTZ) where all uses and human activities, especially gill net fishery, were prohibited (Gücü & Erkan, 2005). Additionally, to reduce the fishing pressure on the monk seals' feeding grounds and allow the fish stocks to recover, 16 nautical miles long, 12 miles No-Trawling Area (Gücü & Erkan, 1999) was established, and the use of all fishing types, excluding bottom longline and gill net, has been restricted (Gücü & Erkan, 2005).

The extensive fishing restrictions and regulations applied in this context indicate that the region has also been designated as a Fisheries Restricted Area (FRA). As mentioned at the beginning of Chapter 1 and stated by FAO (2011), various terms are used interchangeably; therefore, the term FRA will be used throughout this thesis to refer to the Kızılliman area.

All the regulations published in the fisheries circulars and communiques covering this area are listed below.

Fishing regulations applied in the Kızılliman FRA, and its surrounding regions are as follows:

- Fishing with any gear other than gillnets and longlines is prohibited in Mersin Province, Aydıncık District, in the territorial waters between Sancak Cape (36° 7' 28.80'' N - 33° 23' 36.48'' E) and Kızılliman Cape in Bozyazı District (36° 4' 14.58'' N - 33° 4' 42.48'' E).
- In Mersin Province, Anamur District, fishing is prohibited within 200 meters from the shore between Arap Cape in the Kızılliman area (36° 5' 12.00'' N - 33° 5' 27.96'' E) and Gökçevlik area (36° 6' 30.00'' N - 33° 6' 37.98'' E).
- In Mersin Province, Anamur District, within a 1-mile coastal strip between the coordinates (36° 2' 16.92'' N - 32° 41' 54.42'' E) and (36° 2' 7.74'' N - 32° 43' 9.00'' E), with Karaağan Cape at the center, fishing is prohibited within 200 meters from the shore.
- In the territorial waters between İncekumburnu and Kızılliman Cape (36° 4' 14.58'' N - 33° 4' 42.48'' E), fishing with bottom trawls is prohibited within 2 miles.
- In the territorial waters between Kızılliman Cape (36° 4' 14.58'' N - 33° 4' 42.48'' E) and Kesik Cape in Gazipaşa District, Antalya Province (36°

9’ 57.84’’ N - 32° 23’ 25.08’’E), fishing with bottom trawls is prohibited within 1.5 miles.

- In Mersin Province, Silifke District, fishing with bottom trawls is prohibited within 2 miles from the coast of Dana Island.
- In Mersin Province, in Taşucu Bay, fishing with surrounding nets is prohibited in the area north of the line between Güvercinada (36° 14’ 42.36’’ N - 33° 48’ 25.20’’ E) and İncekumburnu (36° 14’ 1.50’’ N - 33° 57’ 45.66’’ E).
- In the Kızılliman area, at the coordinates (36° 06’ 26’’ N - 33° 06’ 40’’ E), (36° 05’ 17’’ N - 33° 05’ 31’’ E), (36° 06’ 09’’ N - 33° 05’ 46’’ E), and (36° 06’ 00’’ N - 33° 07’ 10’’ E) recreational and sports fishing is prohibited (the Republic of Türkiye Ministry of Agriculture and Forestry, 2007 and 2024).

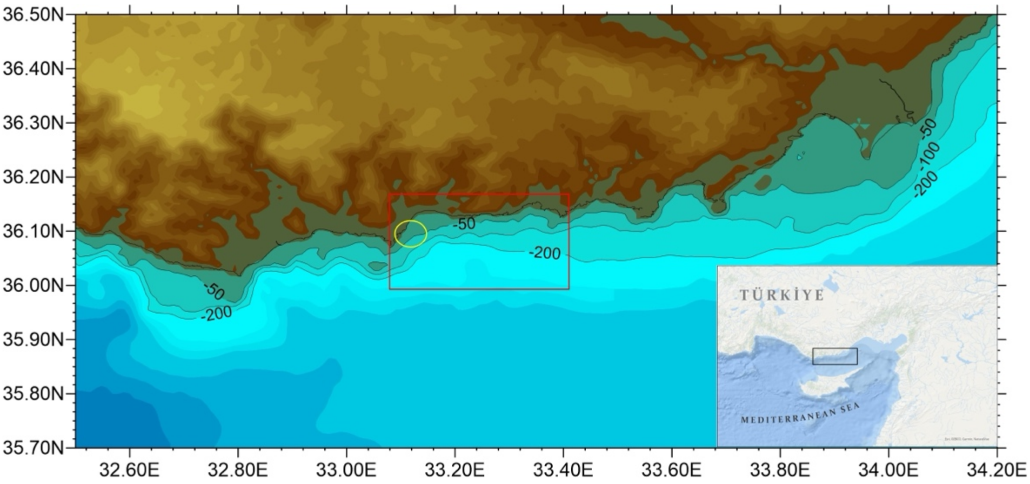

Figure 1. 2 General location and bathymetric map of the Kızılliman Area in the Eastern Mediterranean. The red marked area indicates the region closed to trawling, and the yellow circled area represents the fully protected zone where all activities are prohibited.

Considering all the information given in this chapter about the protection measures, MPAs, and their growth so far, the rise in conservation efforts and the extensive use of MPAs worldwide can be considered positive developments for the protection of marine ecosystems. However, there is a critical factor that must be considered since it could impact the MPAs and implemented protection measures on the ecosystem in the Mediterranean. This factor is the presence of non-indigenous species (NIS) (Iacarella et al., 2019).

1.3 Non-Indigenous Species

Non-indigenous species (NIS) are plants or animals that are purposely or accidentally introduced into a new area, where they have established inhabitants and spread into the new region (IUCN, 2000). The Mediterranean ecosystem and fisheries are significantly affected by these NIS. The non-indigenous species in the Mediterranean are of both the fauna of the Red Sea of tropical Indo-Pacific origin and Atlantic origin species (Golani, 1998; Guidetti et al., 2010). However, Red Sea species are more dominant as the Indo-Pacific origin species can be more adaptable to diverse ecological niches than Atlantic origin ones (Ben-Tuvia, 1973). Also, there is another factor that contributes to their dominance, this factor is the Suez Canal (Turan et al., 2024).

The opening of the Suez Canal in 1869 connected the Mediterranean and the Red Sea. This connection has led to various species originating from the Red Sea or Indo-Pacific immigrating into the Mediterranean through the Canal (Ben-Tuvia, 1966). Although the immigration of NIS may depend on various reasons (e.g., the vessels' ballast waters, aquaculture, and shipping activities), the main pathway of NIS introduction into the Mediterranean Sea is accepted to be the Suez Canal (Katsanevakis et al., 2013; Gücü et al., 2021; Galil, 2023). Their immigration became notable promptly after the Canal opened, and 11 species were recorded in the Mediterranean before 1900 (Galil et al., 2018). After that, in the first decade of the 20th century, 13 NIS were recorded, and their introduction was attributed to the Suez Canal (Galil, 2012). Following that, the total number of NIS recorded in the Mediterranean has continued to increase, more than doubling after 1970, and the greatest numbers were reported, especially in the 1990s and the 2000s (Galil et al., 2018).

In the past, it was believed that the canal-introduced NIS populations would stay limited to the southeast Levantine Basin (Galil et al., 2018). However, they have expanded their range westwards and northwards in the past two decades (Galil et al., 2018). Although more NIS have been recorded in the Eastern Mediterranean (particularly the Levantine Basin) than in the Western Mediterranean so far, NIS have also reached Western regions (Galil, 2012). Recently, their immigration and spreading rate have accelerated (Mavruk et al., 2017), and some anthropogenic factors affected these processes, such as the enlargement of the Canal in 2015 (Zenetos, 2017; Galil et al., 2018) and the increased seawater temperature of the Mediterranean caused by climate change (Raitsos et al., 2010).

The increasing temperature trend of the Mediterranean is known to be enhancing the expansion ability of NIS, which have already been established in the region, to spread within their new environments (Rahel & Olden, 2008) and making the species composition in the Eastern Mediterranean similar to the Red Sea (Raitsos et al., 2010). This process is named as “tropicalization process” (Bianchi & Morri, 2003). This increasing temperature of the Mediterranean is more evident, especially in its northeastern corner, which is known to be heating 20% faster than the average global warming rate (MedECC, 2020). Moreover, besides the increased temperatures, the enlargement of the Suez Canal has also contributed to the introduction of NIS and their spreading ability (Zenetos, 2017; Galil et al., 2018; Galil, 2023). Enlargements of the Canal have increased the entrance of epipelagic larvae/juveniles of deeper living NIS (Galil, 2023). Consequently, although NIS were limited to habitats shallower than 50-meter depths in the 1970s, recent findings documented them on the deeper shelf and even 200-meter depth and beyond (Galil, 2023). Nowadays, NIS numbers have reached high values. More than 1000 NIS have been recorded in the Mediterranean, and 751 of them have been successfully established (Zenetos et al., 2022).

The effects of NIS pressure, which began with the opening of the Suez Canal and continued with other factors, made the Mediterranean Sea one of the most invaded marine regions worldwide (Edelist et al., 2013). In the habitats NIS originated from, they live in balance, and various ecosystem interactions control their populations (Otero et al., 2013). However, they can display invasive behavior in the new marine environment and pose a risk to indigenous species (IS) and their habitats (Giakoumi & Pey, 2017). They can cause biodiversity loss in the Mediterranean by changing food webs and productivity and leading to the loss of indigenous genotypes (Pauly et al., 1998; Galil, 2007). Moreover, they tend to fill available niches in the ecosystem and replace indigenous species in similar ecological position (Edelist et al., 2013). NIS also both positively and negatively affect fisheries in the Mediterranean. Examples of these were seen in the Turkish and Tunisian waters. In Tunisia, fisheries are affected negatively by problems caused by the invasive blue swimming crab (*Portunus segnis*) damaging fishing nets, changing the range of species caught, and reducing catches of many commercial indigenous species (Souissi et al., 2024). Turkish artisanal fishermen have also complained about the NIS, as *Lagocephalus sceleratus* has left damage in fishing gear (Çinar et al., 2021). Conversely, some NIS contribute positively to the economy in certain areas of the Levantine region, and even fishermen have begun to change their target depths and target shallower waters to catch more NIS (van Rijn et al., 2020).

When focusing on the situation in Türkiye, one of the Mediterranean countries heavily impacted by NIS, recent reports showed a total of 100 NIS fish species in Turkish waters (Turan et al., 2024). Of these, 50 have a common distribution, and 32 exhibit invasive behavior (Turan et al., 2024). On the Mediterranean coasts of Türkiye, a total of 89 NIS fish species were recorded, and 41 of these have expanded their distribution and spread to the Aegean Sea to the west (Turan et al., 2024). Currently, the Mediterranean and Aegean coasts of the Türkiye are under intense NIS pressure, with almost no grid area along these coasts remaining free from these species (Çinar et al., 2021). However, in some areas, NIS are more intense, such as in the İskenderun, Mersin, and Antalya Bay (Çinar et al., 2021), where NIS consisting almost half of the total catch (Gücü et al., 2010; Gücü et al., 2012; Gürlek et al., 2018; Mutlu et al., 2023).

1.4 Non-Indigenous Species Pressure in Mersin Province

Mersin Bay, located in the northeastern Mediterranean, has a wide continental shelf, and due to the region's river input and bottom topography, it is a highly productive environment in terms of demersal fish species (Gücü & Bingel, 1994; Gökçe, 2012). Hence, Mersin Bay has been the main trawling ground in Türkiye since the 1950s (Bingel et al., 1993; Gücü & Bingel, 2022). Although the target species in the region generally encompass the Mediterranean's indigenous species (IS), such as *Mullus barbatus*, *Mullus surmuletus*, *Merluccius merluccius*, *Sardina pilchardus*, *Solea solea*, non-indigenous species (NIS), such as *Saurida lessepsianus*, *Siganus rivulatus*, *Siganus luridus*, *Nemipterus randalli* and *Upeneus moluccensis* are also exploited on commercial scale (Bingel et al., 1993; Gücü et al., 2010; TurkStat, 2023; Mersin Governorship, 2023). The presence of these NIS in the Northeastern Mediterranean extends back to the 1950s, indicating a long-standing history of pressure from NIS in this region (Kosswig, 1956; Geldiay & Mater, 1968), and their abundance during these years led to their appearance in local fish markets (Kosswig, 1956). By the early 1980s, NIS had continued to increase in the region, comprising almost half of the total catch by weight, with the *S. lessepsianus* and *Equulites kluzingeri* being the main dominant NIS in the catch (Gücü et al., 2010).

In the following period, the growth of the fishing fleet intensified fishing activities in the region, causing significant pressure on demersal fish stocks (Gücü & Bingel, 2022). As a result, stocks had overfished, and indigenous species' catch per unit effort (CPUE), a measure of relative abundance, declined (Gücü & Bingel, 2022). Meanwhile, NIS opportunistically increased, leading to irreversible alterations in the local ecosystem (Gücü & Bingel, 2022).

By the 2000s, in an evaluation conducted between 2007 and 2010 in the same area, although the percentage of IS in the total catch had increased, NIS still constituted a significant proportion, and *E. kluzingeri* was dominantly contributing to this percentage (Gücü et al., 2010). Same year in the Kızılliman FRA, located west of

Mersin Bay, where the continental shelf is narrow, NIS accounted for a quarter of the total biomass in the catch (Gücü et al., 2010).

Until 2015, NIS constituted a significant portion of the catch in Mersin Bay despite fluctuations in their relative abundance over time (Gökçe et al., 2016). Furthermore, the abundance of NIS was also high in western parts, close to where the Kızılliman FRA is located (Mutlu et al., 2023).

Based on the information presented, it is clear that NIS have been present in and around Mersin Bay for a long time and that their increasing abundance has significant impacts on the local marine ecosystem. In this context, it has become essential to investigate the relationship between the factors and NIS that may influence their presence in these regions.

1.5 The Purpose of the Study

The main objective of this thesis is to investigate the effects of MPAs on regions under intense pressure from NIS in the Mersin Province located in the Eastern Mediterranean.

MPAs are known to promote ecosystems to a healthy state by increasing fish biomass and species richness within their boundaries (Halpern, 2003; Lester et al., 2009; Sala & Giakoumi, 2018). It is also widely accepted that the ecological resilience of a healthy ecosystem can create a barrier against NIS and prevent their establishment in the region (Ardura et al., 2016). In this thesis, it was hypothesized that the Kızılliman FRA provided these positive protection outcomes and negatively affected the NIS. Based on this hypothesis, it was aimed to test the positive effects of protection measures and whether the same preventive effect against NIS can be observed in the Mersin coast of Türkiye, which is one of the regions experiencing the highest pressure from NIS.

Within this scope, the effects of protection measures on the ecosystem and the success of NIS were analyzed by focusing on two different cases: the Kızılliman

FRA, where the trawl ban is enforced, and the fully protected No-Take Zone (NTZ) within the FRA. Two common methods were used for this analysis: an underwater visual census, which is often used due to its minimal impact on the ecosystem (Harmelin-Vivien et al., 1985), and the method of demersal trawl survey (Sciberras et al., 2013). The changes in the biomass and numbers of IS and NIS teleost fish species in the Kızılliman FRA and the effects of protection on the NIS over time are presented in Chapter 2 under the manuscript titled "Recovery of the Overexploited Fishes and the Conservation Impact on Non-Indigenous Species in the Kızılliman Fisheries Restricted Area (FRA), Eastern Mediterranean." Furthermore, the effects of a fully protected No-Take Zone over the recruitment areas in promoting IS biodiversity and preventing the NIS are presented in Chapter 3 under the manuscript titled "The Effectiveness of Marine Protected Areas (MPAs) in Controlling Non-Indigenous Species and Promoting Native Biodiversity in the Eastern Mediterranean."

CHAPTER 2

RECOVERY OF THE OVEREXPLOITED FISHES AND THE CONSERVATION IMPACT ON NON-INDIGENOUS SPECIES IN THE KIZILLIMAN FISHERIES RESTRICTED AREA (FRA), EASTERN MEDITERRANEAN

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Abstract

In this study, we explored the evolution of teleost fish populations within a Mediterranean area, transitioning from pristine to overfished, and ultimately, to a protected region over the last decade. Utilizing data collected since the 1980s, we monitored the shifting states of the ecosystem through 363 valid bottom trawl samples in both the protected Fisheries Restricted Area (FRA) and adjacent fishing grounds, paying close attention to variations between indigenous and non-indigenous teleost species. Notably, the ecosystem's response to conservation efforts was observed five years after the initiation of protection measures. This response period is crucial for understanding the recovery dynamics, with biomass exceeding its original, pre-disturbance levels after ten years. Species were categorized based on their reactions—disappearance, emergence, decline, increase, non-responsive, and competitor. Despite these efforts, species identified with high vulnerability to fishing activities did not show significant recovery. Additionally, we recorded an increase in biomass in the fishing grounds adjacent to the FRA, primarily attributed to Non-Indigenous Species (NIS), though the biomass of NIS within the FRA remained markedly lower than external areas. Our findings underscore the positive impact of fishing closures on the ecosystem in the northeast Mediterranean, a region heavily impacted by NIS pressure. Although protection measures are not a complete solution for NIS, they have been proven to have a role in slowing the spread of NIS, highlighting the crucial role of conservation strategies in mitigating the impacts of NIS.

Keywords: Marine Protected Areas, Non-indigenous species, Trawl surveys, Fishing pressure, Degraded ecosystems

2.1 Introduction

Anthropogenic impacts such as pollution, ocean acidification, overfishing, and climate change have drastically degraded the world's marine environments and coastal ecosystems (Allemand & Osborn, 2019). In response to these disturbances, Marine Protected Areas (MPAs) are widely proposed as management tools utilized to restore environmental damage, safeguard marine biodiversity, enhance ecosystem resilience, and sustain the services provided by marine ecosystems (MedPAN & UNEP-MAP-SPA/RAC, 2016). They play notable roles in restoring fish biomass and promoting biodiversity (Sala & Giakoumi, 2018). Although their role in conserving biodiversity is indisputable (Gaston et al., 2008; Green et al., 2011), the question of whether they offer absolute solutions for restoring degraded ecosystems appears to be case-dependent. The impact of implementing protection measures in heavily degraded areas on biological parameters (e.g., density, biomass, average size) is complex and has revealed varied trends across different studies (Fraschetti et al., 2021). Some instances have demonstrated a gradual increase in these metrics within the protected areas over time (Russ & Alcala, 1996), while others have shown minor, insignificant changes (Sala et al., 1998). Certain cases indicated an initial increase but then regressed to their original levels (Ferreira & Russ, 1995). In this context, it is noteworthy that no-take marine reserves are known as the most effective type of MPA. While partially protected MPAs have some advantages in restricting specific activities to protect habitat destruction (e.g., trawling-free areas), their overall effectiveness is generally lower (Sala & Giakoumi, 2018). After the establishment of marine reserves, fish biomass could be restored over time, but not in partially protected MPAs (Sala & Giakoumi, 2018).

Despite the positive effects of the MPAs, their effectiveness in controlling biological invasions, particularly by NIS, remains subject to debate (Burfeind et al., 2013; Ardura et al., 2016). As MPAs foster diverse ecosystems and species richness (Lester et al., 2009), they may hinder the establishment of NIS (Stachowicz et al., 2002). High species diversity can pose challenges for NIS, making it harder for them to establish themselves within robust, diverse communities. Consequently, MPAs may play a role in deterring the entry and establishment of NIS. Studies examining the effects of protection on NIS biomass and density indicate a significantly negative impact within MPAs compared to unprotected areas (Giakoumi & Pey, 2017; Malherbe & Samways, 2014; Ardura et al., 2015). For instance, in the Caribbean Sea, invasive lionfish species showed lower biomass in MPAs than in unprotected areas (Mumby et al., 2011; Hackerott et al., 2013). The effectiveness of MPAs in deterring NIS proliferation is, however, not universal. Some other studies have shown that MPAs do not always prevent the establishment and thriving of NIS. On the contrary, certain conditions within MPAs may facilitate NIS spread, leading them to benefit from the fishing prohibitions enforced and exhibit higher abundance within protected areas (Kellner & Hasting, 2009). Additionally, the presence of effective dispersal mechanisms enables many NIS to traverse protected area boundaries. If predation and competition show no variation between protected and fished areas, then there is no chance of observed a protected area effect on NIS (Burfeind et al., 2013).

An advantageous behavior from fishing restrictions was exemplified by the higher biomass of *Siganus luridus*, an invasive of Indo-Pacific origin, observed thriving within a well-protected MPA than in the adjacent areas, along with an indigenous

Sparisoma cretense (Giakoumi et al., 2019). Critics argue that NIS typically threaten native populations and habitats through habitat degradation, loss of native genotypes, and disruptions to food webs and ecosystem productivity, ultimately leading to the displacement of indigenous species (Molnar et al., 2008; Albins, 2013; Giakoumi, 2014; Vergés et al., 2014). Despite claims that NIS can enhance regional diversity, studies such as Worm et al. (2006) indicate that these introductions often fail to compensate for the loss of native species.

The Mediterranean ecosystem faces a significant threat from NIS. The construction of the Suez Canal in 1869, connecting the Mediterranean and the Indian Ocean, played a pivotal role in facilitating NIS invasion, as various species originally from the Red Sea or Indo-Pacific region entered the Mediterranean through the canal. Their spreading in the Mediterranean has been strongly influenced by rising temperatures (Raitsos et al., 2010), and recently, it has accelerated (Mavruk et al., 2017), and this process is called as "tropicalization process" (Bianchi & Morri, 2003). The Mediterranean is projected to experience intensified warming, and an increase in the annual average basin sea surface temperature exceeding the global average (Cos et al., 2022). This trend is particularly evident in the eastern corner of the Mediterranean, where climate change has led to the significant decline of multiple species (Rilov, 2016). As temperatures continue to rise, the entry and spread of NIS are expected to intensify.

Within this framework, the present study examines alterations in fish fauna and biomass, focusing on the impact of fishing restrictions on NIS proliferation in the Eastern Mediterranean. Although NIS' responses to protection measures may vary by location (Giakoumi et al., 2016), this study hypothesizes that MPAs can mitigate NIS spread. The research aims to provide insights into (i) the effects of fishing restrictions on fish biomass recovery in the degraded ecosystem, (ii) the impact of restrictions on NIS proliferation, and (iii) the species-specific responses to implemented restrictions.

2.2 Materials and Methods

2.2.1 Study Area

The study was conducted in the coastal area in the northeastern corner of the Mediterranean (Figure 2. 1) which is known to warm 20% faster than the average global warming rate (MedECC, 2020). Prior to the 1980s, large-scale fishing was not achievable due to the lack of suitable ports in the region, and only small-scale fishing was conducted. This period of the region was considered as a "pristine phase" in the present study. However, this scenario began to change towards the end of the 1980s. The construction of a port in 1988 marked a turning point, followed by the addition of two more ports in 1992 and 1997 (Department of Fisheries and Aquaculture Services, 2004). These developments caused a replacement from small-scale fishing to industrial fishing with large trawls and purse seines. Due to the continental shelf in this area being exceptionally narrow, fishing activities, mainly targeting demersal and small pelagic species, were attached to this limited continental shelf. In the subsequent period, the fish stocks in the region were depleted rapidly due to the changing fishing fleet and narrow fishing areas. As a result, a series of measures were implemented, and all forms of fishing, except for a bottom long line and gill net used by regional small-scale fishers, were prohibited in 1999. In the same year, a fully protected No-Take Zone was also established in the region. Thus, 16 nautical miles long, 12 miles No-Trawling-Area and a small No-Take Zone were established for protection (Gücü & Erkan, 1999) delineated between the coordinates 36.1247° N - 33.3935° E and 36.0707° N - 33.0785° E.

As the focused region in the scope of the present study has been closed to only large-scale fishing due to the implementation of fishing restrictions, it exhibits the characteristics of a Fisheries Restricted Areas (FRAs). Consequently, in the continuation of this chapter, FRA terminology will be used to describe the region.

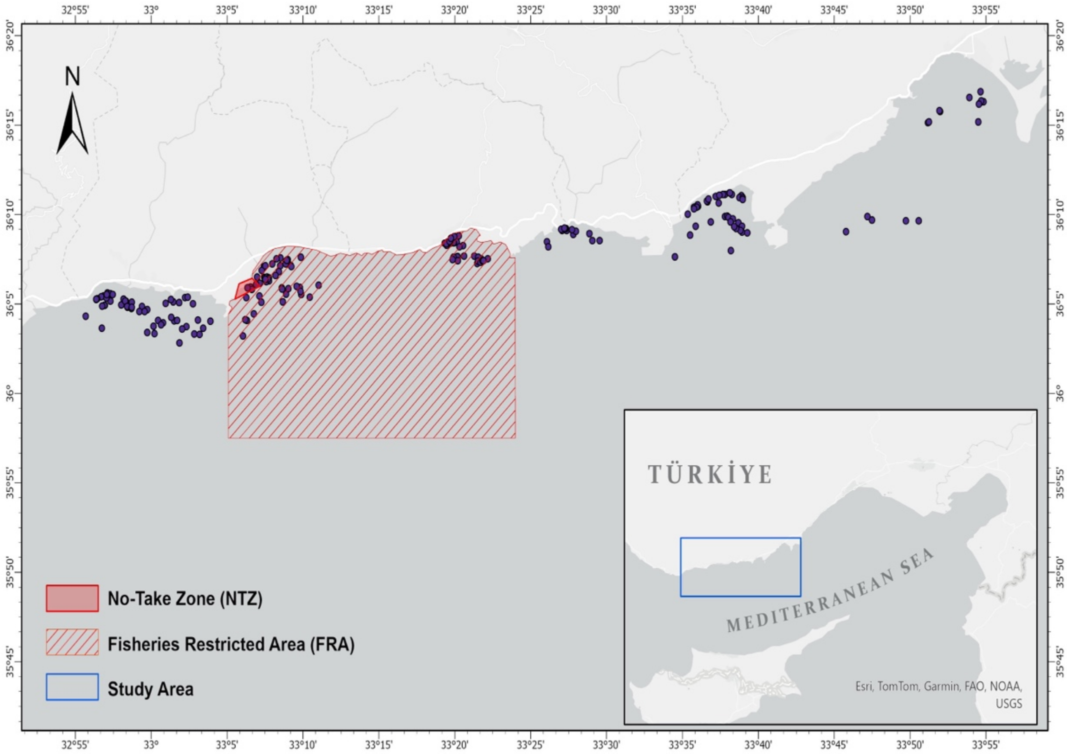

Figure 2. 1 The map of the study area in the Eastern Mediterranean. The red hatched area represents the region banned to large-scale fisheries (Kızılliman FRA). The red marked area within the FRA represents the No-Take Zone, and the small points represent the sampling stations.

2.2.2 Sampling Design and Analysis

The field surveys were carried out twice a year, covering spring (May) and autumn (October), in 1983-2023 as much as the weather conditions, sea state, and the availability of research vessel (R/V Lamas of the Middle East Technical University) permitted. Between 1983 and 2023, in the Eastern Mediterranean, a total of 34 surveys were carried out covering 363 valid bottom trawl hauls. The same local trawl net design, locally called the Ottoman with a stretched cod end mesh size of 22 mm (Gücü, 2012), was used in all surveys, and the same sample protocol was followed throughout the study. In each survey, the lower boundary of the infralittoral region (0-25 m), the beginning of the circalittoral region (25-50 m), and its deep parts (50-100 m) were sampled. The sampled depth strata were selected due to represent different faunal features of the northeastern Mediterranean Sea (Gücü & Bingel, 1994). Additionally, all sampling stations inside and outside the FRA were selected to represent the same depth range and environmental conditions to ensure consistency and accuracy in comparing FRA and adjacent outside areas.

After the catches were brought aboard the vessel, all the samples were sorted out by species level. Each species was weighed with 1 gram sensitivity, and the number of individuals was counted and recorded. To estimate the biomass, CPUA was calculated based on Sparre & Venema (1998) by dividing the catch by the area swept during the haul (A, in square kilometers).

The formulation of it is as follows:

$$A = D * hr * X2$$

Which "*D*" is the distance covered, "*hr*" is the head rope length, and "*X2*" is a fraction of the head rope length. The estimation considered Pauly's (1980) recommended fraction of head rope length (*X2*) = 0.5 as the optimum compromise. Also, it was assumed that the trawl net's catchability coefficient is (q=1). The length of the head rope of the trawl net used was 23.7 m.

During the biomass estimation, only teleost fish species were included, and total teleost fish biomass, NIS biomass, IS biomass, NIS ratio in total biomass, and number of species were calculated.

On a species basis, the biomass for each species was assessed and subsequently categorized into seven distinct groups based on their biomass change responses to the establishment of FRA (Table 2. 1). For fish species included in the categories, biological information such as trophic level (TL), resilience (the ability of a fish population to tolerate impacts and persist over time in the face of environmental variation and change), minimum population doubling time (the minimum number of years required for a population to double in size at the current rate of population increase), generation time (the average age of parents at the time their young are born) and fishing vulnerability (vulnerabilities of marine fishes to fishing) were used from www.fishbase.org (Froese & Pauly, 2024).

Table 2. 1 Classification of teleost fish species and definitions based on their biomass response following the establishment of the FRA.

Group	Definition
Group 1 Vanished Species	Species that were initially present in the region but have since disappeared in terms of biomass after the depletion of fish stocks and have not appeared after the establishment of the FRA.
Group 2 Reappeared Species	Species that were initially present in the area but disappeared with the depletion of fish stocks and reappeared in terms of biomass after the establishment of FRA.
Group 3 Non-Indigenous Competitors	Non-indigenous species that compete with other non-indigenous species within the FRA.
Group 4 Guardian Species	Indigenous species that, despite experiencing a decrease in biomass following the introduction and spread of non-indigenous species in the region, subsequently display an increase with the establishment of FRA.

Group 5 Resident Non-Indigenous Species	Non-indigenous species that were consistently present within and outside the FRA.
Group 6 Newly Abundant Non-Indigenous Species	Non-indigenous species that have increased in recent years and reached substantial biomass within and outside the FRA.
Group 7 Unresponsive Indigenous species	Indigenous species that did not show a response in terms of biomass after the establishment of FRA.

All statistical analyses were conducted in R software v.4.3.0 (R Development Core Team, 2023) using the RStudio (Posit team, 2023). The dataset used in this analysis was tested for normality by plotting a QQ-plot. Given the large sample size of the dataset (363 samples) and considering that the study focuses on the estimation of the average values in the trawl hauls for the comparison and evaluation of the areas, the data was assumed to be normally distributed according to the Central Limit Theorem (CLT) (Lumley et al., 2002; Zuur et al., 2009). The CLT states that, regardless of the distribution of the underlying data, the sample averages approach normality as the sample size increases. Therefore, despite slight deviations from normality, the sufficiently large sample size (more than 30) allows the use of parametric methods in the analysis (Lumley et al., 2002).

For the estimation of sample averages, the average values of biomass and number of species in the trawl hauls both inside and outside the FRA were calculated for each survey, with 95% confidence intervals. Following this, to determine changes in the NIS ratio in both areas, the biomass of NIS within the total biomass and the number of NIS species within the total species count were calculated. To estimate and compare the rate of increase in NIS% within the total species count since the beginning of conservation measures over the years between areas and to theoretically project the year in which the NIS% within and outside the FRA is expected to reach 100%, the “lm” function was used. Seasonality was not considered in this analysis for the increase in the NIS%. The difference between observed values inside and outside the FRA was tested using the “lm” and “ANOVA” functions in base R.

2.3 Results and Discussion

In this study, the changes in the number of teleost fish species and their biomass over a 40-year period were analyzed using 135 trawl hauls within the protected area and 228 in the neighboring area adjacent to the FRA. Based on the results, the evolution of fish populations within the FRA was divided into phases: the pristine phase in the early 1980s (before the establishment of the FRA), the overfished phase from the late 1980s to 1999, the response period from 1999 to 2005, and the recovery phase from 2005 to 2010.

The comparative annual average biomass of teleosts inside and outside the FRA, as derived from yearly aggregated data, is depicted in Figure 2.2. At the beginning of the 1980s, the biomass of teleost fish in the area, which was later designated for protection and was largely pristine at that time, showed no statistically significant difference ($t= 1.34$, $p> 0.05$) from that in the surrounding areas (Figure 2.2). However, upon comparison, the percentage of NIS in the total biomass of teleosts is found to be lower within the FRA compared to outside it (Figure 2.3). The observed difference may be attributed to the westward expansion pattern of species originating from the Suez Canal along the Anatolian coast (Gücü & Bingel, 1994). This is compounded by the fact that many stations outside the future FRA are situated in regions that are further east.

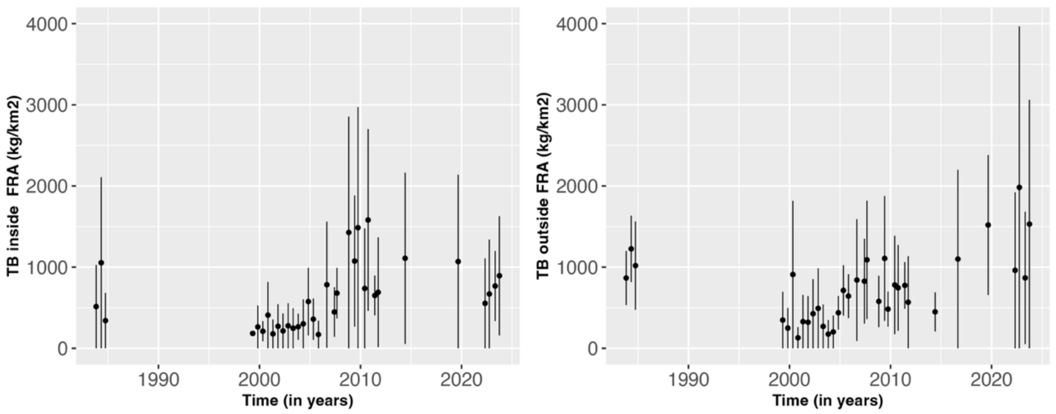

Figure 2.2 Comparison of average total teleost biomass (TB) changes inside (Left) and outside (Right) the FRA. Black dots represent the average value of the biomass of species in the trawl hauls with 95% confidence intervals (CI)

In 1999, the year the FRA was established, the biomass in both areas experienced a drastic and similar decline, indicating that the cause of this reduction is a large-scale factor that affected both areas equally. The construction of a series of fishing ports (Department of Fisheries and Aquaculture Services, 2004) and the unmanaged expansion of industrial fishing fleets, coupled with the region's steep bathymetry and narrow continental shelf that confines fishing grounds to limited areas, have significantly increased fishing pressure on fish stocks (Gücü & Erkan, 1999). This has inevitably led to their overexploitation within two decades, making overexploitation a primary factor for the decline, as indicated in Figure 2.2.

The decline in the percentage of NIS within the total teleost biomass (NIS%) during the same period in the outer area, where the biomass was higher in its pristine state, indicates that the fishery targeted not only indigenous species but also significantly depleted NIS populations (Figure 2.3), including species such as *Saurida lessepsianus* and *Upeneus moluccensis* (Table A1 and A2, see Appendix).

Figure 2.3 Changes in the percentage of NIS (NIS%) within the total teleost biomass (TB) over the years. The purple triangle shape represents the outer area, the red dot shape represents the FRA, and the lines represent the LOESS smoother lines.

After the implementation of conservation measures, no immediate significant changes in biomass were observed. However, a gradual increase became apparent after five years of the protection measures and was observed in 2005, displaying the ecosystem's lag in responding to conservation. This delay reflects the unique recovery dynamics inherent to each ecosystem, with the length of the response period varying significantly. Past experiences in various locations indicate that this period can range from a few years to several decades. The time needed to re-establish natural trophic balances within protected areas often contributes to this variation (Halpern & Warner, 2002; Shears & Babcock, 2002; Shears & Babcock, 2003). The response time noted in this study points to a relatively prompt reaction of the ecosystem to the protective measures put in place. Considering that the species observed in this study are generally within a 2-5 year age range at capture (personal observation), and given the heavy fishing pressure these stocks have faced—

resulting in a demographic composition dominated by younger or juvenile fish—it is understandable that these populations require a considerable amount of time to recover and regenerate.

Over this five-year response period, changes within the teleost composition were also noted. The presence of NIS within the FRA increased, albeit at a slower rate than in areas outside of the protection (Figure 2.3 and Figure 2.4). In respect of the total number of species, there was an increase outside the FRA at the end of the 1990s and the beginning of 2004 (Figure 2.5). Within the FRA, a parallel uptick in species diversity was also recorded in 2004; however, the increase observed in 2004 occurred concurrently with an increase in biomass, presenting a contrast to the changes observed during the end of the 1990s (Figure 2.2). Accordingly, the surge in species numbers, concurrently with increased biomass within the FRA was primarily due to transient species that appeared during the period and then vanished. These transient species were indigenous species, including *Uranoscopus scaber* (UUC), *Pagrus auriga* (REA), *Citharus linguatula* (CIL), *Dentex dentex* (DEC), *Epinephelus costae* (EPK) as well as *Arnoglossus* species, and non-indigenous species, such as *Lagocephalus sceleratus* (LFZ), *Fistularia commersonii* (FIO) (Table A1, see Appendix).

Figure 2.4 Changes in the percentage of NIS (NIS%) within the total number of teleost fish species (TN) over the years. The green triangle shape represents the outer area, and the red dot shape represents the FRA, and the lines represent the LOESS smoother lines.

After the first response observed in 2005, fish populations entered the recovery phase, and although observed at different scales, the trends in total teleost fish biomass both within the FRA and in adjacent areas aligned up until 2010 (Figure 2.2). At this point, a notable divergence occurred: the biomass within the FRA surged to a peak nearly seven times greater than its initial, untouched, pristine state. The recorded peak biomass value in the FRA was achieved after a span of 10 years. Despite the notable increase in teleost biomass within the FRA, suggesting a stronger resilience against NIS, an analysis of new species introductions in the region shows no significant differences (t= -1.68, p>0.05) between the FRA and the surrounding areas, as illustrated in Figure 2.5. This indicates that although the total teleost biomass in the FRA exhibits robust resistance to NIS, species diversity does not differ, whether inside or outside the FRA. Despite studies conducted in New

Caledonia (Wantiez et al., 1997) and a comprehensive literature review covering multiple areas (Lester et al., 2009) have shown that protected areas typically exhibit greater species richness, no such difference was observed in the Eastern Mediterranean, consisting with other studies (Roberts & Polunin, 1992).

Throughout the recovery phase, from 2005 to 2010, a consistent upward trend in the NIS% was observed outside the FRA (Figure 2.3). Within the FRA, in contrast, the trend in NIS% did not display steadiness, suggesting that the primary contributors to biomass increase were indigenous. Moreover, this increase in biomass within the FRA was not uniformly distributed, as indicated by the wide confidence intervals (CIs) (Figure 2.2). This variability implies that the ecological responses to recovery measures, and possibly the effectiveness of these measures, varied across different sections of the FRA. Additionally, the observed increase in NIS biomass in unprotected areas to the extent that NIS have surpassed those of indigenous species in the samples (Figure 2.6). The marked statistical difference ($p<0.01$) in the rates of increase between the two areas can be directly linked to the conservation measures in place within the FRA.

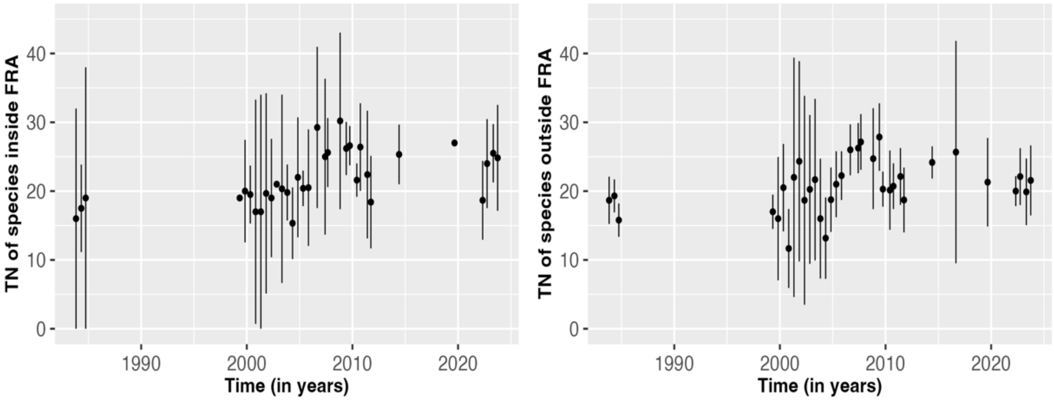

Figure 2.5 Comparison of total number teleost species (TN) changes inside (Left) and outside (Right) the FRA. Black dots represent the average value of the number of species in the trawl hauls with 95% confidence intervals (CI).

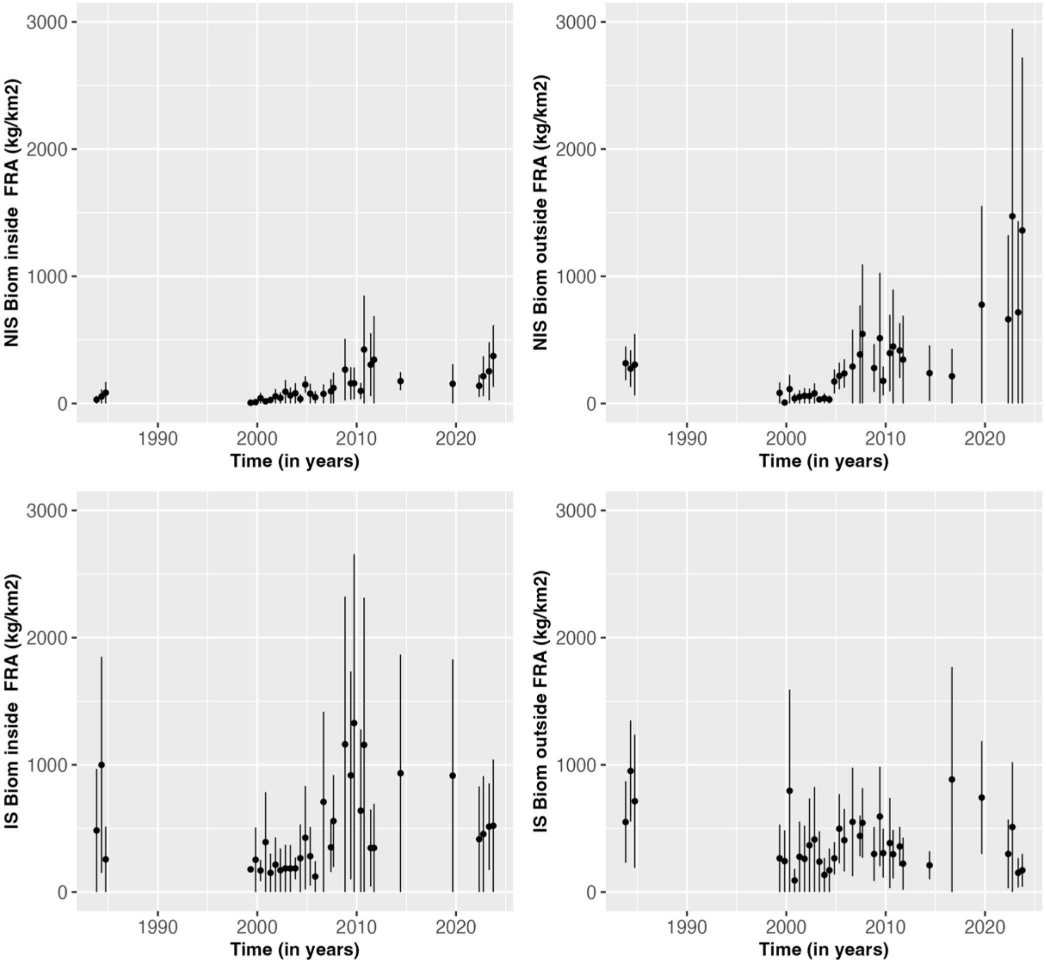

Figure 2.6 Comparison of average IS (Lower panel) and NIS (Upper panel) biomass changes inside (Left) and outside (Right) the FRA. Black dots represent the average value of the biomass of species in the trawl hauls with 95% confidence intervals (CI)

After that recovery phase onwards (after 2010), it is observed that the situation has begun to change completely. While the biomass within the FRA displayed a decreasing trend, the opposite was observed outside this area (Figure 2.2). Although there are similar cases that indicated an initial increase but then decreased to pre-protection levels (Ferreira & Russ, 1995), in the present study, biomass levels that were observed in recent years were above the pristine phase levels within the FRA. On the other hand, the observed steady increase outside the FRA was mainly due to the contribution of the NIS (Figure 2.6). This increasing pattern of NIS has also been observed within the FRA, although not as much as seen in the area outside the FRA. Besides the positive influence of protection on teleost biomass within the FRA, its resistance against NIS also appears to be a marked reversal (Figure 2.3 and Figure 2.6). This decline in previously noted positive effects is thought to be associated with the heat waves and their long-term effects that have swept through the Mediterranean during the period after 2010 (Martínez et al., 2023; Marullo et al., 2023). As for the total teleost biomass, there is a potential explanation that can account for the reversal patterns within the FRA after 2010: the duration of the conservation period and its influence on efficacy. While the observation period in the present study is extensive, previous research suggests that achieving "complete" recovery necessitates a significant considerable time, as well as instances where the impact of conservation efforts amplifies progressively over time (Watson et al.,1996; Russ & Alcala,1998). However, in some unfortunate instances, ecosystems may never fully recover and return to their previous states despite conservation efforts (Halpern., 2003). Hence, although initial positive developments were observed in the conservation response regarding biomass shortly after the beginning of conservation efforts, the subsequent decline after 2010 suggests that the effectiveness of protection may increase over time, potentially leading to a subsequent rise in biomass. However, it should also be considered that even if the initial response is constructive, irrespective of the duration of the protection, the ecosystem within the FRA may not fully recover as observed in some fully protected marine reserves (Sala & Giakoumi, 2018). Since small-scale fishing continues in the region, the effectiveness of protection observed in marine reserves may not be observed in FRA. Despite this, observations in recent times indicates a pronounced increase in biomass within the FRA. This increase, however, was not uniform across all monitoring stations but rather irregular, as suggested by the widened confidence intervals (CIs), indicating a variation in the recovery rates within the FRA (Figure 2.2).

Following the observed reversal changes after the recovery phase, a linear regression analysis was conducted to shed light on the future trajectory of NIS. Through a linear regression analysis based on the rate of increase in the NIS% within the total species count since the beginning of the conservation, it has been observed that the rate of increase over the years was significantly high in both areas ($p<0.001$, $t= 8.90$). However, although this rate of increase outside the FRA was found to be higher (slope = 0.015 NIS ratio/per year, CIs = 0.013 - 0.020) than within the FRA (slope = 0.013 NIS ratio/per year, CIs = 0.010 - 0.016), the statistical difference between the areas was not significant ($p>0.05$, $t= -1.04$). Additionally, in the theoretical projection of the year in which the NIS% is expected to reach 100% within and outside the FRA based on this rate of increase, it is projected that by the year 2064, NIS might have replaced all indigenous species within the FRA, and by 2053 outside the FRA. These results emphasize that although the FRA's conservation strategies may not completely block the entry of NIS, they play a critical role in limiting their rate of entry and expansion within the protected area. On the other hand, in terms of new indigenous species introductions and the counts, the statistical analysis of data presented in Figure 2.7 conclusively demonstrates that there has been no statistically significant change in the number of indigenous species at FRA stations throughout the study period ($p>0.05$), indicating that the diversity of indigenous species have remained relatively unchanged within the FRA. In contrast, a significant decline in indigenous species counts has been observed in samples from areas outside the protection zone over time (Figure 2.7), highlighting the positive impact of conservation efforts.

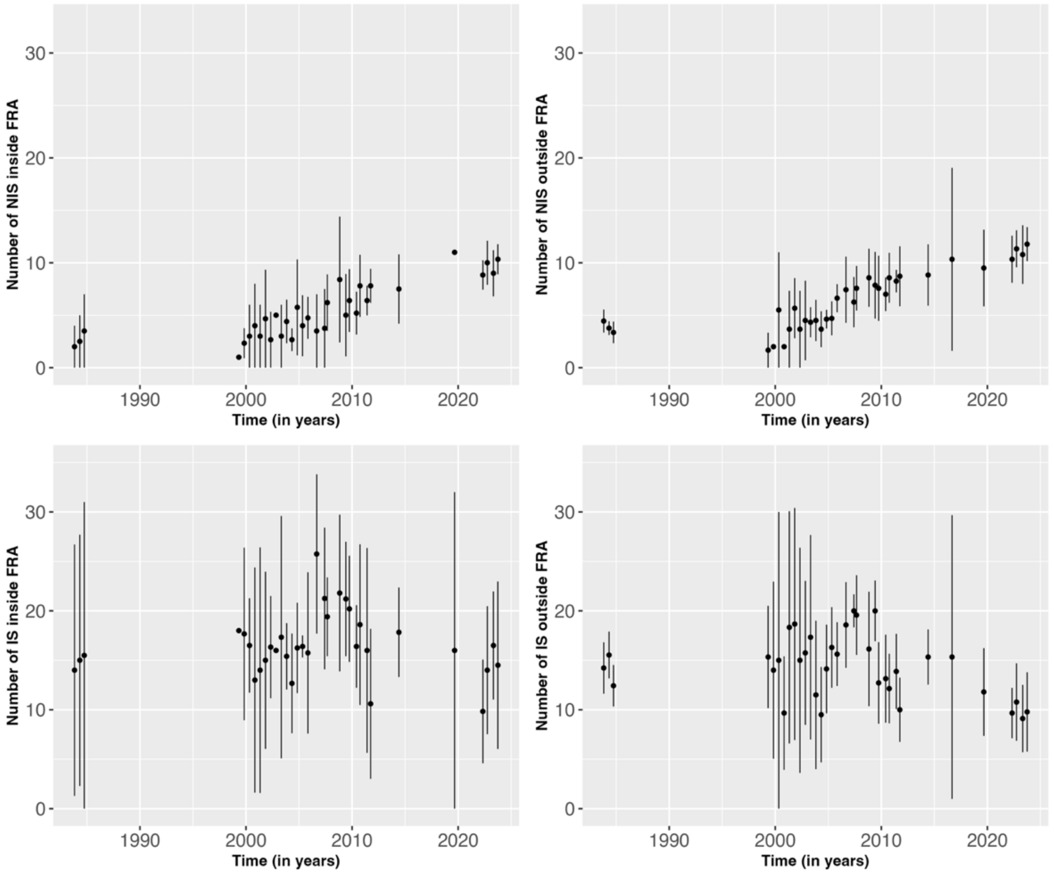

Figure 2.7 Comparison of number of IS (Lower panel) and NIS (Upper panel) changes inside (Left) and outside (Right) the FRA. Black dots represent the average value of the number of species in the trawl hauls with 95% confidence intervals (CI)

During the study period, a species-specific assessment was also undertaken covering the FRA and outside it to achieve deeper insights into the biomass and biodiversity changes and specifically identify the NIS that caused the notable changes. To provide a more systematic evaluation of teleost species' reactions to conservation efforts, the species were grouped based on their respective responses. Within this scope, Group 1 vanished species was first determined. Before the protection, indigenous species *Merluccius merluccius* (HKE)*, Trachinus draco* (WEG)*, Trachurus trachurus* (HOM)**,** *Arnoglossus laterna* (MSF) and *Dentex macrophthalmus* (DEL) were present and had substantial biomass within the FRA (Table A1, see Appendix).

However, probably as a result of overfishing in the late 1980s, these species experienced a decline. Despite subsequent conservation efforts, they have unfortunately failed to reestablish their presence within the FRA. The trophic levels of these specified species are changing 3.5 - 4.4 (Table 2.2), showing carnivorous character and predominantly feeding on fishes and decapods as their primary food source (Morte et al.,1999; Froese & Pauly, 2024; Kilongo et al., 2007; Bayhan et al., 2008; Fehri-Bedoui & Gharbi, 2008; Stagioni et al., 2011; Segadilha et al., 2011). From here, it can be inferred that the disrupted ecosystem balance due to overfishing affected the common food sources of these species. Besides these shared feeding habits, their moderate to high vulnerability to fishing (Table 2.2) can also be associated as a contributing factor to responses observed in conservation efforts. The noticeable indigenous species exhibiting positive responses to conservation efforts were identified as *Bothus podas* (OUB), *Chelidonichthys lastoviza* (CTZ), *Citharus linguatula* (CIL) and *Lepidotrigla cavillone* (LDV) (Group 2 reappeared species). After experiencing a decline in their biomass, these species initially struggled to maintain their presence in the area and even vanished for a while. However, they have successfully reestablished their existence over time through ongoing conservation efforts within FRA. In contrast to Group 1, Group 2 species have a low level of vulnerability to fishing, as shown in Table 2.2, and their ability to succeed in reappearing within the FRA depends on being less affected by the effects of fishing activities.

Table 2.2 Biological information on teleost fish species classified into groups (Froese & Pauly, 2024; Cheung et al., 2005; Froese et al., 2017).

Groups and Species	**Resilience (minimum population doubling time)**	**Trophic Level**	**Generation Time**	**Fishing Vulnerability**
Group 1 vanished species		Median value: 3.7		Median value: 54
Merluccius merluccius	Medium (1.4 - 4.4 years)	4.4	9	65 (High)
Trachinus draco	Medium (1.4 - 4.4 years)	4.2	-	52 (Moderate to high)
Trachurus trachurus	Medium (1.4 - 4.4 years)	3.7	6.5	56 (High)
Arnoglossus laterna	Medium (1.4 - 4.4 years)	3.6	-	36 (Moderate)
Dentex macrophthalmus	Medium (1.4 - 4.4 years)	3.5	-	54 (Moderate to high)
Group 2 reappeared species		Median value: 3.4		Median value: 33.5
Bothus podas	Medium (1.4 - 4.4 years)	3.4	-	35 (Low to moderate)
Chelidonichthys lastoviza	Medium (1.4 - 4.4 years)	3.5	3	32 (Low to moderate)
Citharus linguatula	Medium (1.4 - 4.4 years)	4.0	-	37 (Moderate)

Lepidotrigla cavillone	Medium (1.4 - 4.4 years)	3.3	-	25 (Low)
Group 3 non-indigenous competitors		Median value: 3.5		Median value: 10
Upeneus moluccensis	Medium (1.4 - 4.4 years)	3.6	1.4	22 (Low)
Upeneus pori	Medium (1.4 - 4.4 years)	3.5	-	10 (Low)
Equulites klunzingeri	Medium (1.4 - 4.4 years)	3.5	-	10 (Low)
Equulites elongatus	High (< 15 months)	3.2	-	10 (Low)
Group 4 guardian species		Median value: 3.3		Median value: 37
Mullus barbatus	Medium (1.4 - 4.4 years)	3.1	3.2	34 (Low to moderate)
Pagellus erythrinus	Medium (1.4 - 4.4 years)	3.5	6.1	40 (Moderate)
Group 5 resident non-indigenous species		Median value: 3.5		Median value: 10
Upeneus moluccensis	Medium (1.4 - 4.4 years)	3.6	1.4	22 (Low)
Upeneus pori	Medium (1.4 - 4.4 years)	3.5	-	10 (Low)
Saurida lessepsianus	High (< 15 months)	4.0	2.7	28 (Low to moderate)

Group 6 newly abundant non-indigenous species		Median value: 3.65		Median value: 25
Nemipterus randalli	Medium (1.4 - 4.4 years)	3.8	-	37 (Moderate)
Pomadasys stridens	Medium (1.4 - 4.4 years)	4.0	-	35 (Moderate)
Parupeneus forsskali	High (< 15 months)	3.5	2.9	15 (Low)
Torquigener hypselogeneion	High (< 15 months)	3.3	-	10 (Low)
Group 7 unresponsive indigenous species		Median value: 3.55		Median value: 46
Diplodus annularis	Medium (1.4 - 4.4 years)	3.6	4.5	36 (Moderate)
Diplodus vulgaris	Medium (1.4 - 4.4 years)	3.5	4.3	32 (Low to moderate)
Diplodus sargus	Medium (1.4 - 4.4 years)	3.4	-	57 (High)
Pagrus pagrus	Medium (1.4 - 4.4 years)	3.9	6.6	48 (Moderate to high)
Serranus cabrilla	Medium (1.4 - 4.4 years)	3.4	5.7	55 (Moderate to high)
Serranus scriba	Medium (1.4 - 4.4 years)	3.8	-	44 (Moderate)

During the species-specific assessment, a potential competition among non-indigenous species was suggested. This state may exist between *Upeneus molluccensis* (UPM) and *Upeneus pori* (UPH), as well as between *Equulites klunzingeri* (PON) and *Equulites elongatus* (KZK) (Group 3 non-indigenous competitors). Within the FRA, *U. molluccensis* and *U. pori* presented disparate biomass patterns: where one species experienced an increase, the other either declined or showed no presence (Table A1, see Appendix). Given their notably similar diet preferences (Golani, 1993), competitive interactions between these species within the FRA could be inferred. A similar pattern may also be present between *E. klunzingeri* and *E. elongatus* (Table A1 and A2, see Appendix). The presence of *E. elongatus* has only been found within the boundaries of the FRA, while *E. klunzingeri* showed higher biomass outside the protected zone. Although direct competition between these two species has not been reported yet, the potential for a competitive interaction could explain *E. elongatus'* inability to increase its biomass outside the FRA.

Some indigenous species, thought to have been exposed to various stressors such as fishing pressure, heat waves in the Mediterranean, and new NIS introductions to the area, have experienced declines in their biomass. Nevertheless, they have recovered from these impacts and demonstrated resilience against these stress factors. The species economically crucial *Mullus barbatus* (MUT) and *Pagellus erythrinus* (PAC), demonstrating such resilient responses, were categorized as Group 4 guardian species. These species have consistently been observed within and outside the FRA. However, it is noteworthy that their biomasses were higher within the FRA compared to outside the area (Table A1 and A2, see Appendix). The Mullidae family, to which the *M. barbatus* belongs, is known for its relative abundance, wide distribution, tolerance of a wide range of environmental conditions, and easy to sample. Consequently, they are frequently utilized as indicators in fisheries studies to assess response to ongoing intense fishing activities or recovery of fish populations within MPAs after reduced fishing pressures (Claudet et al., 2006; Uiblein, 2007). In the present study, the significantly high biomasses and recovery of *M. barbatus* and *P. erythrinus* after encountering various stressors highlight their role as resilient natural guardians within the FRA. This underscores how conservation efforts have demonstrably yielded restorative effects, particularly concerning commercially important species. Prior to species-specific evaluation, it was anticipated that *M. surmuletus* might also be classified as a Group 4, given its status as an indicator species within non-fished areas, with its high abundance in the northwestern Mediterranean (Claudet et al., 2006). However, despite showing high biomass within the FRA compared to outside areas overall, since it has not been observed in recent years in either area, it has not been included in this group. Moreover, the resilient impact of these two guardian species became notably apparent in their interaction with *U. pori* and *U. molluccensis*. Despite the settlement and consistent presence of *Saurida lessepsianus* (LIB), *U. molluccensis*, and *U. pori* (Group 5 resident non-indigenous species) in both areas, they failed to demonstrate a concurrent increase in biomass within the FRA relative to outside it (Table A1 and A2, see Appendix). Previous studies have documented the replacement of *M. barbatus* and *M. surmuletus* by *U. pori* and *U. molluccensis* in the southwestern Mediterranean (Golani, 1994). However, it appears that guardian species in the northeastern Mediterranean, situated within the FRA, demonstrated resilience against this shift.

Contrary to the previously mentioned positive effects and resistance, certain non-indigenous species undergoing recent increases in biomass have been identified and classified as Group 6 newly abundant non-indigenous species. *Nemipterus randalli* (NNZ), *Pomadasys stridens* (PKS), *Parupeneus forsskali* (RPF) and *Torquigener hypselogeneion* (QFF) which have recently expanded their abundance in the Mediterranean (Lelli et al., 2008; Bariche et al., 2015; Mavruk et al., 2017; Eşkinat et al., 2023) and attracted the attention of Mediterranean researchers by showing an increasingly successful distribution, were initially detected outside the protected area (Table A1 and A2, see Appendix). Subsequently, although they have not preceded the attainment of substantial biomass levels as in the outside, their unsteady existence and increasing biomass were noted within the protected area boundaries. In addition, compared to other non-indigenous species, *P. forsskali* showed a more rapid introduction and increase in biomass in the borders of the FRA. The rapid increase observed in the biomass of *P. forsskali* can be attributed to a minimum population doubling time of less than 15 months and a relatively low generation time (2.9) (Froese & Pauly, 2024). In comparison with *S. lessepsianus*, which is a resident non-indigenous species (Group 5), it is apparent that their minimum population doubling time and generation time values are similar (Table 2.2). This may suggest the possibility of *P. forsskali* being the resident non-indigenous species within the FRA in the upcoming years, similar to *S. lessepsianus*.

In the species-specific assessment, the final group was identified as Group 7 unresponsive indigenous species. Despite conservation efforts, the reactions of certain indigenous species remained unclear. These species, including *Diplodus annularis* (ANN), *Diplodus vulgaris* (CTB), *Diplodus sargus* (SWA), *Pagrus pagrus* (RPG), *Serranus cabrilla* (CBR), and *Serranus scriba* (SRK) generally showed parallel biomass patterns, with minor discrepancies, in both areas. Consequently, it can be suggested that these non-commercial species may not have fully benefited from protection measures. The responses of fish species to conservation measures can vary across taxa, and non-commercial species may not exhibit as pronounced a positive response to protection measures as commercially targeted species (e.g., *M. barbatus*) (Mosqueira et al., 2000).

2.4 Conclusion

The findings of the present study revealed that the response of teleost fish populations over a 40-year period within the FRA established in an area subjected to both overfishing and the pressure of NIS. Following the implementation of fishing restrictions, fish populations within the FRA exhibited a prompt initial response to these protective measures, and the restorative effects of conservation efforts were started to be observed five years after the establishment of the FRA. Accordingly, the biomass of fish populations within the FRA returned to pre-disturbance levels after a total of ten years from the start of the protection measures. However, the decline in biomass observed after 2010 suggested that a longer period may be needed for full recovery. On the other hand, the difference between FRA and fished areas regarding NIS indicated that the increased biomass of indigenous species within the FRA has contributed to the establishment of resilience against NIS. The implemented fishing restrictions have helped the observed resilience and significantly slowed the entry and spread of NIS into the FRA. This underscores the positive impact of fishing closures on the ecosystem in the northeast Mediterranean, a region heavily impacted by NIS pressure. The findings of this study can provide further insights into future management strategies for mitigating NIS in the coming decades.

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CHAPTER 3

THE EFFECTIVENESS OF MARINE PROTECTED AREAS (MPAs) IN CONTROLLING NON-INDIGENOUS SPECIES AND PROMOTING THE NATIVE BIODIVERSITY IN THE EASTERN MEDITERRANEAN

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Abstract

In this study, we investigated changes in species richness within the No-Take Zone (NTZ) in the Eastern Mediterranean. Utilizing data collected from 556 transects with using the underwater visual census technique in the shallow infralittoral zone, we compared observations conducted inside and outside the NTZ in the early 2000s with those performed in the same region during the 2020s. This comparison focused closely on the changes in the diversity of indigenous species (IS) and non-indigenous species (NIS) over these two distinct time periods. Within the NTZ, we observed a high IS diversity at the onset of the protection measures. However, observations in the 2020s indicated a shift: while the diversity and success of NIS increased, IS diversity decreased. In contrast, outside the NTZ, IS maintained their status. Reduced success and diversity of IS in the NTZ attributed to the disruption caused by the introduction of NIS into the ecosystem, which altered the habitat dynamics shared by IS in the infralittoral zone. Outside the NTZ, intensive small-scale commercial or sports fishing, predominantly targeting NIS, played a role in controlling their populations. The findings revealed that when the fishing pressure on NIS in the coastal infralittoral zone was eliminated with the implemented protection measures, it inadvertently created favorable conditions for NIS. These observed results indicate that when an ecosystem already under NIS pressure is protected, NIS can become advantageous in areas such as the Eastern Mediterranean.

Keywords: Coastal areas, Visual census, Species richness, No-Take Zone, Invasive species, Ecosystem resilience

3.1 Introduction

Climate change and human activities, particularly unsustainable fishing, have profoundly impacted the ecology of coastal areas (Jackson et al., 2001; Lotze et al., 2006; Allemand & Osborn, 2019), leading to significant declines in species abundances and alterations in food web structures (Pauly et al., 1998), and have become a threat to marine biodiversity. Consequently, protecting and restoring marine biodiversity has become a significant priority (Balmford et al., 2005). Marine Protected Areas (MPAs) have been implemented worldwide as a strategy to address biodiversity loss and restore marine ecosystems (He et al., 2022). These areas refer to sections of the coastline or sea where human activities, particularly fishing, are restricted or prohibited (Agardy et al., 2003). The most significant effects of MPAs are promoting and preserving biodiversity (Côté et al., 2001; Halpern, 2003; Lester et al., 2009; Guidetti et al., 2014) and enhancing ecosystem resilience. Besides these, they can also play a role in the recovery of density, biomass, and individual size of the populations usually targeted by fishing (Mosquera et al., 2000; Micheli et al., 2004; Prato et al., 2017). Despite these general positive effects, the effectiveness of MPAs can show a variation, with some instances showing no significant difference between protected areas and adjacent non-protected areas (Roberts & Polunin, 1992; Sala et al., 1998). Furthermore, their effectiveness in managing biological invasions of non-indigenous species (NIS) remains uncertain (Burfeind et al., 2013; Ardura et al., 2016). As MPAs enhance species richness (Lester et al., 2009), they may exhibit resilience against the establishment of NIS within diverse, healthy-state communities (Stachowicz et al., 2002). High indigenous species (IS) diversity can challenge NIS, making it harder for them to establish themselves within resilient, rich communities. Consequently, MPAs may play a role in preventing the settlement of NIS by boosting the IS richness. Studies investigating the impacts of MPAs on NIS indicate a notable negative effect on NIS within MPAs compared to unprotected areas (Mumby et al., 2011; Hackerott et al., 2013; Malherbe & Samways, 2014; Ardura et al., 2015; Giakoumi & Pey, 2017). However, this negative effect of MPAs on the NIS is not always constant. Some other studies have indicated that MPAs do not consistently prevent the establishment and success of NIS. Certain conditions (e.g., reduced fishing pressure) within MPAs may facilitate NIS spread, leading them to benefit from conservation measures, and become more abundant within these protected zones (Kellner & Hasting, 2009; Kleitou et al., 2024).

The Mediterranean Sea is rich in biodiversity and has a high species density (Bianchi & Morri, 2000; Coll et al., 2010), and shows ecological significance with its high rate of endemism and interesting biodiversity by hosting over 17,000 species (Coll et al., 2010; Abd Rabou et al., 2023). The rich biodiversity of the Mediterranean is mainly found in the Western Mediterranean, and it decreases from west to east (Boudouresque, 2004; Coll et al., 2010). Additionally, high species diversity is found in its coastal ecosystems, and these areas are home to over 750 fish species (Froese & Pauly, 2024). These coastal areas are also significantly important as they represent nursery areas for various fishes, including commercial species (Cheminée et al., 2021). However, Mediterranean coastal ecosystems have been increasingly impacted by stressors (e.g., fishing, pollution, climate change, and habitat destruction) and exposed to significant changes to their biodiversity (Coll et al., 2010). Furthermore, the Mediterranean Sea is recognized as a primary hotspot for marine bioinvasions (Rilov & Galil, 2009). Especially its coastal areas are significant recipients of NIS through immigration via the Suez Canal (Galil, 2006; Rilov & Galil, 2009). The influx of these Indo-Pacific and Red Sea origin species has been significantly influenced by increasing temperatures (Raitsos et al., 2010), and more recently, has accelerated in the eastern part of the Mediterranean (Mavruk et al., 2017). NIS immigrations pose a significant risk to indigenous fish diversity, and although there are arguments suggesting that they can increase regional diversity, these introductions often do not compensate for the loss of indigenous species (IS) (Worm et al., 2006). Moreover, some NIS can show invasive behavior in the Mediterranean and cause significant changes in the ecosystem (Molnar et al., 2008; Sala et al., 2011; Giakoumi & Pey, 2017). For instance, in the Eastern Mediterranean, the two NIS *Siganus luridus* and *Siganus rivulatus* have already invaded the coastal areas and surpassed total herbivorous IS biomass (Bariche et al., 2004; Sala et al., 2011). In these coastal areas, their excessive grazing on macroalgae has led to a significant change in habitat by causing seagrass beds to transform into barren rocky substrates (Sala et al., 2011). These transformations lead to species richness decreasing in both fish and benthic invertebrates (Vergés et al., 2014).

Given this information, there is emerging curiosity about the MPAs located in the Eastern Mediterranean. Although MPAs have been acknowledged to increase biodiversity and control NIS, as the region is subjected to multiple stressors (e.g., climate change, NIS influx), it raises concerns about the efficacy of Eastern Mediterranean MPAs in promoting and conserving species diversity and preventing the NIS (Giakoumi et al., 2019; Frid et al., 2023; Kleitou et al., 2024). The present study hypothesizes that MPAs can control NIS by enhancing IS diversity and protected areas have high biodiversity compared to fished areas. To shed light on this, the present study focused on the shallow infralittoral zone, which is under full protection status (No-Take Zone) and is used by juveniles of various Mediterranean species as a nursery area. Within this scope, the diversity of indigenous and non-indigenous fish species in the shallow infralittoral zone was assessed using the visual census technique. This method is one of the primary techniques used for studies in MPAs (Tunesi et al., 2006) since it has minimal impact on the ecosystem (Harmelin-Vivien et al., 1985). The research aims to provide insights into (i) the effects of protection measures on fish species diversity within the protected area and (ii) the impact of protection on NIS in the protected area.

3.2 Materials and Methods

3.2.1 Study Area

The data used for this study were derived from field surveys conducted in the coastal area of the Eastern Mediterranean (Figure 3. 1). This coastal region is an infralittoral zone, generally too shallow for recreational fishing and commercial activities; therefore, only bait trap fishing and sports fishing are practiced. On the other hand, the rocky areas and seagrass beds within this infralittoral zone serve as crucial habitats for juvenile and young-of-the-year fishes, providing shelter and nursery grounds.

In 1999, this area was designated as a No-Take Zone (NTZ) to safeguard Mediterranean monk seals by minimizing human disturbance around their crucial habitats, including breeding and shelter caves (Gücü & Erkan, 2005). In the NTZ, all activities, including recreational and sports fishing, are prohibited within the coordinates (36.1072° N - 33.1111° E), (36.0881° N - 33.0919° E), (36.1025° N - 33.0961° E), and (36.1000° N - 33.1194° E). Additionally, this NTZ is also located within an area designated as a Fisheries Restricted Area (FRA), where trawl fishing is prohibited (Figure 3. 1).

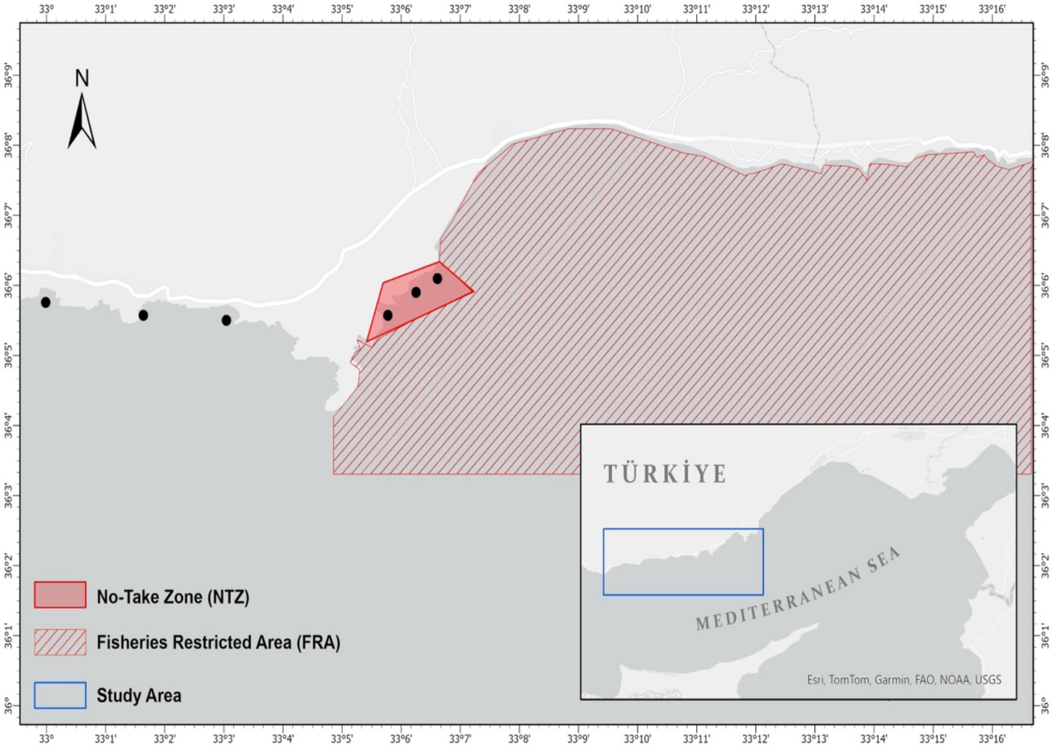

Figure 3. 1 The map of the study area in the Eastern Mediterranean. The red hatched area represents the FRA, the red marked area between the coordinates (36.1072° N - 33.1111° E), (36.0881° N - 33.0919° E), (36.1025° N - 33.0961° E), and (36.1000° N - 33.1194° E) represents the fully protected No-Take Zone, and the small points represent the sampling stations.

3.2.2 Data Collection and Analysis

Visual census surveys were conducted twice a year, and a total of 556 observations were carried out by five observers. 298 of these observations were recorded between the years 2001 and 2003. The remaining observations were recorded from the surveys from 2022 and 2023. The stations were selected both within (3 stations) and outside (3 stations) the NTZ, ensuring they all had the same habitat characteristics. The same sampling method was used for all surveys. The samplings were carried out using freediving visual census techniques at depths ranging from 0 to 5 meters. For the sampling, a 20-meter-long rope was used to mark the census transect. The rope was fitted with lead weights at one-meter intervals to ensure it remained on the seabed. At each station, three transect lines covered rocky hard substrate areas, and one covered the *Posidonia oceanica* habitat.

In the analysis, the observations recorded by each observer were evaluated separately. Observer 1 was a single experienced individual who observed during both periods (2001-2003 and 2022-2023). Observer 2 consisted of two experienced individuals with identical backgrounds in fish identification and this methodology. One individual recorded observations in 2001-2003, while the other observed during 2022-2023. Observers 3 and 4 were two individuals who were relatively less experienced compared to the others. During sampling, Observers 1 and 2 sampled together, as did Observers 3 and 4. The paired observers swam parallel to each other and simultaneously recorded the fish species observed within the transect area on PVC plates.

To provide a comprehensive view of the number of fish species recorded within and outside the NTZ, data from 2001-2003 and 2022-2023 were combined. These combined data sets were then assigned as Phase 1 and Phase 2, respectively, to facilitate comparisons and analyze the changes in species diversity over time. To assess the impact of the protection measures on different habitat structures, the transects on the rocky substrate were categorized as “Rocky infralittoral” and those along the seagrass line as “*Posidonia oceanica*” separately.

To compare the areas and phases, the percentage of occurrence of each species in total sampled transects and the cumulative increase in the number of species observed were analyzed. Observers 1 and 2 provided data for both Phase 1 and Phase 2, whereas Observers 3 and 4 contributed exclusively to Phase 2. Therefore, to provide a general assessment of the changes in species diversity inside and outside the NTZ across different phases, the data from Observers 1 and 2 were prioritized in the analysis.

In the cumulative increase analysis, the species accumulation curve (Thompson et al., 2003), commonly used in ecology to assess biodiversity, was adapted to examine fish species diversity. The cumulative number of indigenous and non-indigenous species was computed for each observer based on transect counts. An increase in biodiversity within the ecosystem was indicated by the rising number of species recorded by observers on each transect. The richness of fish diversity was reflected in the steepness of the accumulation curve; specifically, if there are, say, X number of fish species in the area, the sections where this number is reached more quickly in observations were considered richer in biodiversity (i.e., the steeper the curve, the greater the diversity). In order to quantify this approach, the following assumptions and the corresponding method were used. It was considered that the number of species in a region is not infinite; however, with the method applied in this study (conducting counts over a short distance of just 20 meters), it is not feasible to reach this number quickly. Therefore, it was assumed that the total number of species that can be encountered in an observation could be explained by an asymptotic relationship.

In this relationship, the cumulative number of species observed (S) after "t" observations is given by:

Equation 1: $S_t = S_\infty(1 - \exp(-k\ t))$

Where, S_∞ represents the total number of species that would be observed after an infinite number of observations, and k is the curve's steepness, a rate constant indicating how quickly S_∞ is approached. In other words, k represents the species evenness, while S_∞ is an indicator of the species richness.

To estimate the parameters $\mathbf{S}_\infty$ and $\boldsymbol{k}$, the non-linear least squares regression (NLS) method was applied using the “nls()” function in R software (R Development Core Team, 2023). *Equation 1* was fitted to the cumulative counts of indigenous species (IS) and non-indigenous species (NIS) for this calculation. As seen in the equation, the increase in species count was controlled by two parameters: the asymptote, representing the total number of species that would be observed after an infinite number of observations, and the rate constant, indicating the speed at which this asymptotic species count is approached.

Both the asymptote and the rate of increase were important for the analysis. The asymptote provided an estimate of species richness, while the rate constant reflected how evenly the species were distributed (species evenness).

However, in some cases, the asymptote was high, but the rate of increase was slow, and in other cases, the opposite was true. Since both parameters are important indicators of biodiversity, making a direct comparison was difficult in such cases. Therefore, the following method was applied to evaluate conservation performance. The asymptote and rate constant obtained from all observations were double log-transformed and plotted (Figure 3. 2 and Figure 3. 3), revealing a linear relationship between them.

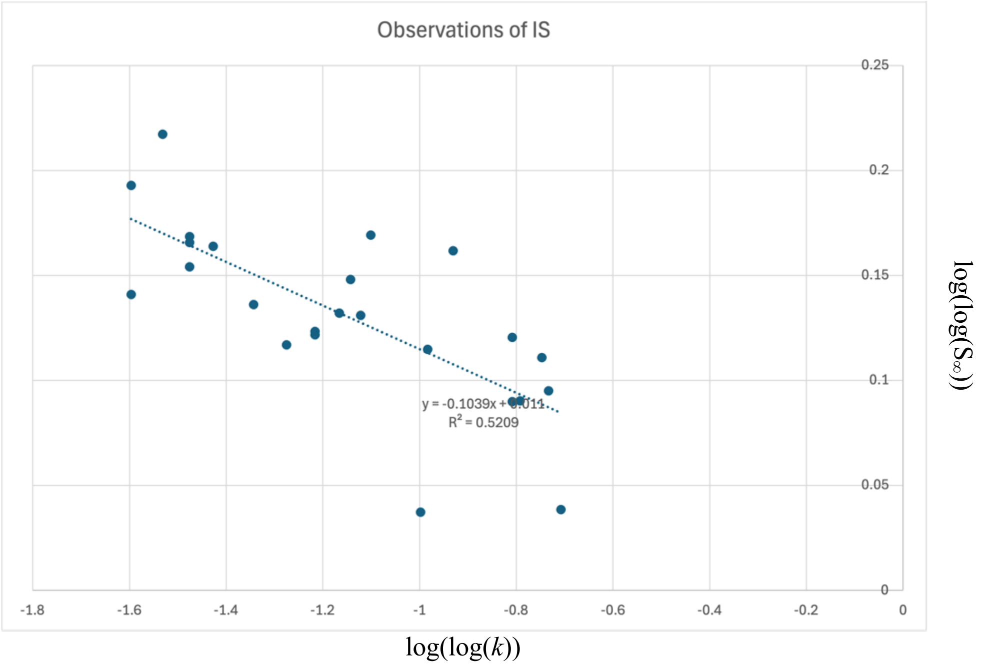

Figure 3. 2 The linear relationship between double log-transformed asymptote (S_∞) and rate constant (*k*) obtained from IS observations.

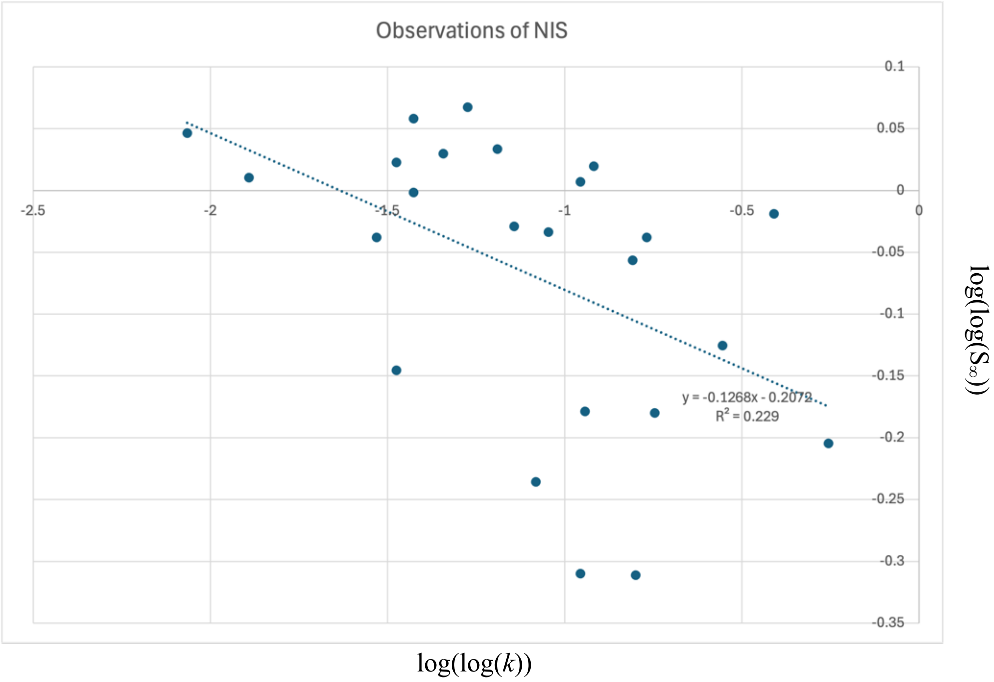

Figure 3. 3 The linear relationship between double log-transformed asymptote (S_∞) and rate constant (*k*) obtained from NIS observations.

To compare conservation performance, the parameter pairs were analyzed based on their distances from this linear relationship. Specifically, pairs that were further above the line indicated higher conservation performance, as they suggested both high species richness and rapid approach rates.

To calculate conservation performance within and outside the NTZ the following equation was used (*Eq 2*).

$$\textit{Equation 2:}\ \ Bioperf = \log_{10}(\log_{10} curv) + b * \log_{10}(\log_{10} asym) + \epsilon$$

Where *Bioperf* is the biodiversity performance (conservation performance) and *b* is the constant slope which obtained from linear relationship between double log-transformed asymptote and rate constant (Figure 3. 2 and Figure 3. 3). Also, *curv*, and *asym* are the coefficients of *Eq 1*, represent the rate constant (*k*) and the asymptote value (S_∞), respectively.

Finally, all analyses were conducted in R software v.4.3.0 (R Development Core Team, 2023) using the RStudio (Posit team, 2023), and all graphs were prepared in the ggplot package of RStudio.

3.3 Results

In Phase 1, a total of 61 species were observed inside the NTZ, of which 10 were NIS and 51 were IS. Outside the NTZ, there were a total of 47 species, of which 5 were NIS and 42 were IS.

In Phase 2, a total of 43 species were recorded inside the NTZ, with 13 being NIS and 30 being IS. Outside the NTZ, a total of 46 species were recorded, with 6 being NIS and 40 being IS.

Evaluation of the NTZ showed that in the early years following the establishment of the NTZ, the number of species observed in at least 50% of the samples was 3, and all of them were IS. In Phase 2, this number reached 10, and 7 of them were IS. Additionally, in Phase 1, species *Atherina hepsetus*, *Serranus cabrilla*, and *Sarpa salpa*, which were observed in at least 10% of the transects, disappeared in Phase 2 (Figure 3. 4). Conversely, species *Fistularia commersonii*, *Pterois miles*, *Scarus ghobban*, and *Mycteroperca rubra*, absent in the first period, appeared in over 5% of the transects, and *Torquigener hypselogeneion* and *Parupeneus forsskali* emerged with a high percentage of occurrence in Phase 2 (Figure 3. 4).

On the other hand, the evaluation for the outside of the NTZ showed that in Phase 1, the number of species observed in at least 50% of the samples was 6, of which 4 were IS. In Phase 2, this number reached 11, and 8 of them were IS. Additionally, in Phase 1, *Serranus cabrilla*, which was observed in at least 10% of the transects, vanished in Phase 2 (Figure 3. 5). Conversely, species *Atherinomorus forskalii*, *Pteragogus pelycus*, *Scarus ghobban*, *Pomadasys incisus*, *Epinephelus costae*, *Caranx crysos*, and *Tripterygion tripteronotum* appeared in over 5% of the transects. Furthermore, identical to the NTZ, *Torquigener hypselogeneion*, and *Parupeneus forsskali* emerged with a high percentage of occurrence in Phase 2 (Figure 3. 5).

The scientific names of all species observed by Observer 1 and 2, both inside and outside the NTZ, are presented in Table 3. 1, and the percentages of occurrence in transects are shown in Figure 3. 4 and Figure 3. 5.

Table 3. 1 Families and scientific names (according to the FishBase, Froese & Pauly, 2024) of all species observed inside and outside the NTZ by Observer 1 and 2 and their corresponding code names. Codes starting with ‘N’ represent indigenous species, while codes starting with ‘L’ represent non-indigenous species.

Family	**Species**	**Code**	**Family**	**Species**	**Code**
Atherinidae	*Atherinomorus forskalii*	L_FCW	Mullidae	*Mullus barbatus*	N_MUT
	Atherina sp.	N_ATH		*Mullus surmuletus*	N_MUR
	Atherina hepsetus	N_AHH		*Parupeneus forsskali*	L_RPF
Apogonidae	*Apogon imberbis*	N_OGT		*Upeneus pori*	L_UPH
	Cheilodipterus novemstriatus	L_HWZ	Muraenidae	*Muraena helena*	N_MMH
Belonidae	*Belone belone*	N_GAR	Pempheridae	*Pempheris rhomboidea*	L_PVL
Blenniidae	*Blennius ocellaris*	N_NUO	Pomacentridae	*Chromis chromis*	N_CMK
Bothidae	*Bothus podas*	N_OUB	Scaridae	*Sparisoma cretense*	N_PRR
Carangidae	*Caranx crysos*	N_RUB		*Scarus ghobban*	L_USY
	Seriola dumerili	N_AMB	Sciaenidae	*Sciaena umbra*	N_CBM
	Trachinotus ovatus	N_POP	Scorpaenidae	*Pterois miles*	L_PZO
Epinephelidae	*Epinephelus costae*	N_EPK		*Scomber colias*	L_MAS
	Epinephelus marginatus	N_GPD		*Scorpaena scrofa*	N_RSE
	Epinephelus aeneus	N_GPW	Serranidae	*Serranus scriba*	N_SRK
	Mycteroperca rubra	N_MKU		*Serranus cabrilla*	N_CBR
				Serranus hepatus	N_SRJ
Fistulariidae	*Fistularia commersonii*	L_FIO	Siganidae	*Siganus rivulatus*	L_SRI
				Siganus luridus	L_IGU
	Fistularia petimba	L_FIP		*Boops boops*	N_BOG

Gobiidae	*Deltentosteus quadrimaculatus*	N_DEQ	Sparidae	*Diplodus annularis*	N_ANN
	Gobius sp.	N_GOB		*Diplodus cervinus*	N_SBZ
	Gobius niger	N_GBN		*Diplodus sargus*	N_SWA
Haemulidae	*Pomadasys incisus*	N_BGR		*Diplodus puntazzo*	N_SHR
Holocentridae	*Sargocentron rubrum*	L_HWH		*Diplodus vulgaris*	N_CTB
Labridae	*Coris julis*	N_COU		*Lithognathus mormyrus*	N_SSB
	Labrus mixtus	N_USI		*Oblada melanurus*	N_SBS
	Labrus viridis	N_VIR		*Pagellus erythrinus*	N_PAC
	Pteragogus pelycus	L_PEL		*Pagrus auriga*	N_REA
	Symphodus tinca	N_TIN		*Spicara flexuosum*	N_FLE
	Symphodus roissali	N_ROI		*Spicara smaris*	N_SPC
	Symphodus mediterraneus	N_MED		*Sparus aurata*	N_SBG
	Symphodus melops	N_YFM		*Sarpa salpa*	N_SLM
	Symphodus rostratus	N_ROS	Sphyraenidae	*Sphyraena sphyraena*	N_YRS
	Symphodus cinereus	N_YFC	Synodontidae	*Synodus saurus*	N_SDR
	Thalassoma pavo	N_TMP	Tetraodontidae	*Torquigener hypselogeneion*	L_QFF
Moronidae	*Dicentrarchus labrax*	N_BSS	Tripterygiidae	*Tripterygion tripteronotum*	N_TTP
Monacanthidae	*Stephanolepis diaspros*	L_KOY		*Tripterygion delaisi*	N_TDA
Mugilidae	*Chelon auratus*	N_MGA	Uranoscopidae	*Uranoscopus scaber*	N_UUC
	Mugil cephalus	N_MGS			

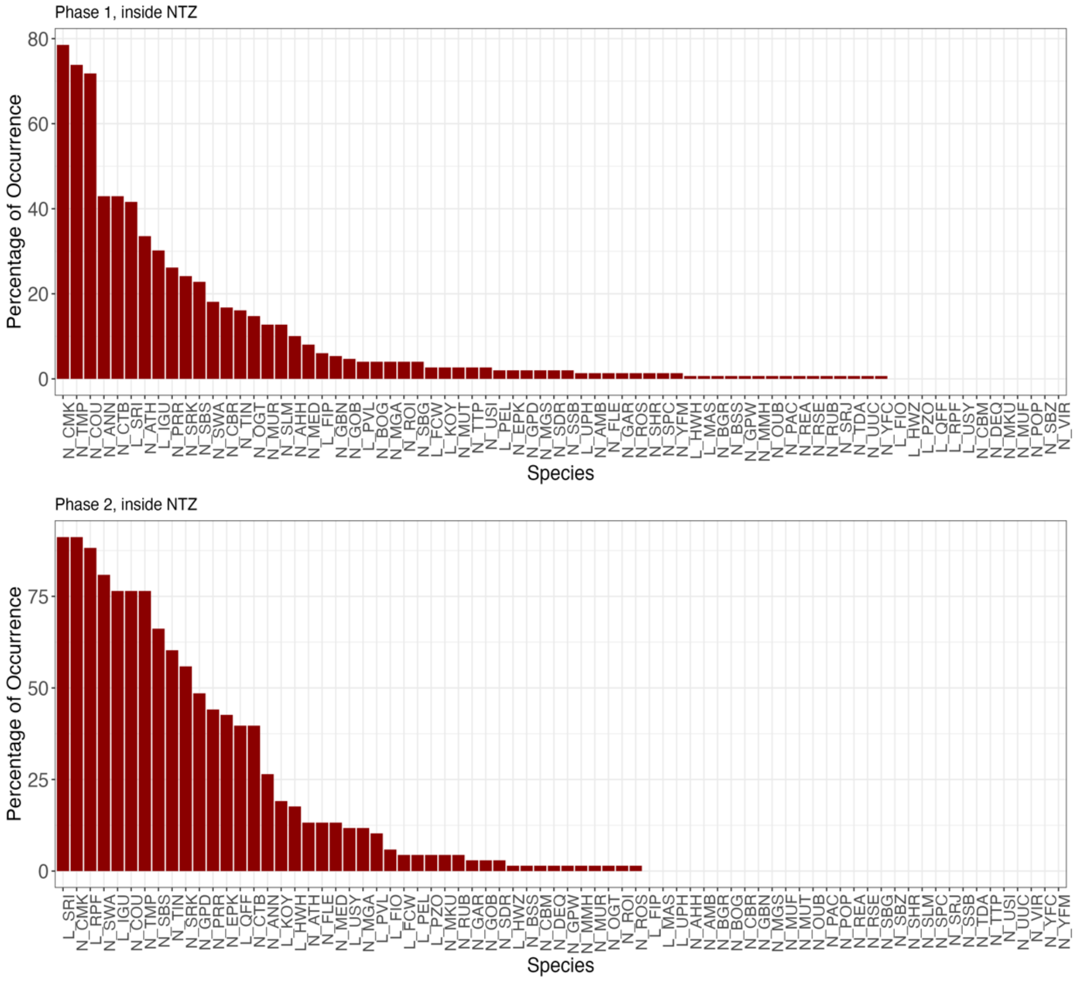

Figure 3. 4 The results of the percentage of occurrence analysis of all species in total transects sampled by Observer 1 and Observer 2 inside the NTZ are presented in the rank abundance curves. The top graph represents Phase 1 and the below graph represents Phase 2. The scientific names and codes of the species are presented in Table 3. 1.

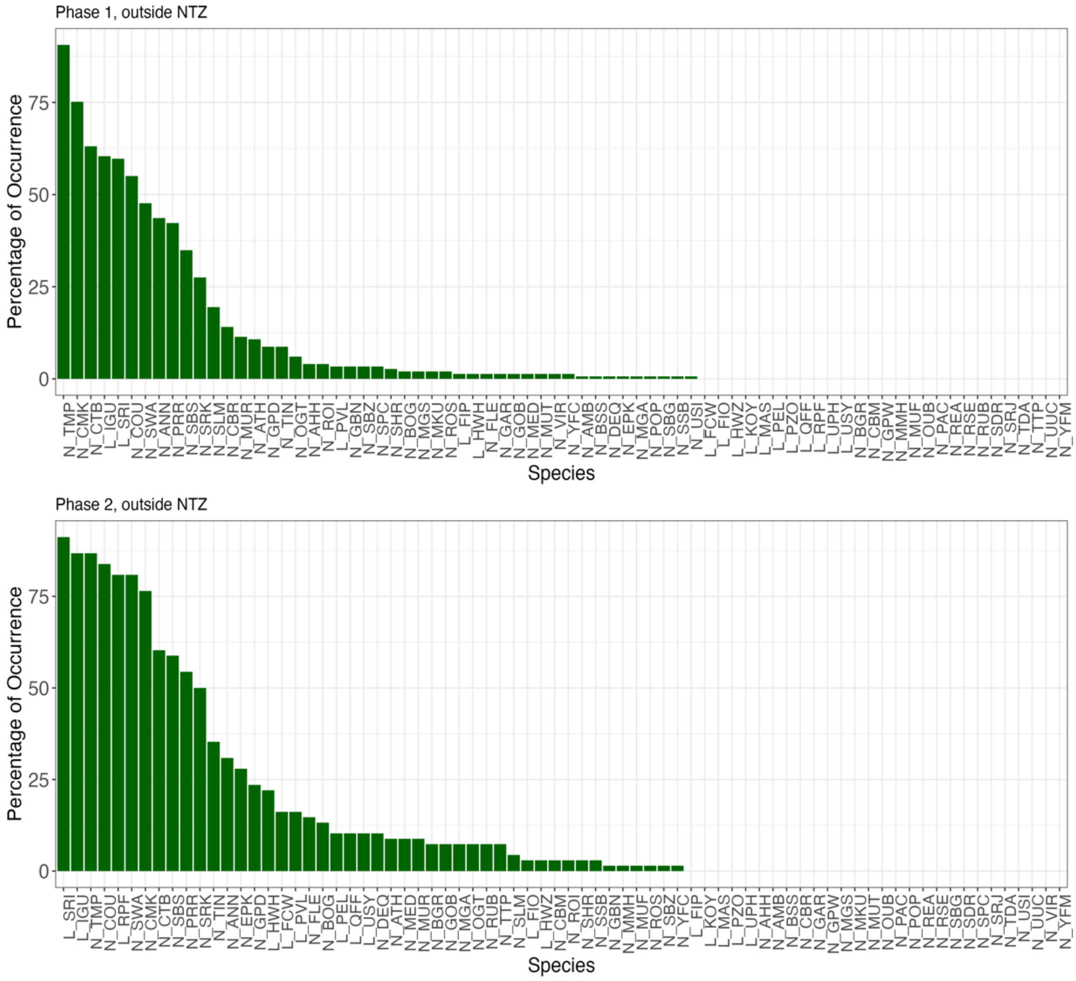

Figure 3. 5 The results of the percentage of occurrence analysis of all species in total transects sampled by Observer 1 and Observer 2 outside the NTZ are presented in the rank abundance curves. The top graph represents Phase 1 and the below graph represents Phase 2. The scientific names and codes of the species are presented in Table 3. 1.

In the evaluation of cumulative numbers of species and biodiversity performance, the results for each observer have been evaluated and interpreted separately. All calculated *curv*, *asym*, and *Bioperf* values are presented in Table 3. 2. Other biodiversity performance values results according to the transect habitats are presented in Table 3. 3 and Table 3. 4.

Table 3. 2 Indigenous and Non-Indigenous Species biodiversity performance values of all transects and observers.

Indigenous Species						Non-Indigenous Species					
Observers	**Phases**	**Region**	**Bioperf**	**asym**	**curv**	**Observer**	**Phase**	**Region**	**Bioperf**	**asym**	**curv**
Ob.1	1	NTZ	0.03	46.31	0.07	Ob.1	1	NTZ	0.01	13.82	0.02
Ob.1	1	Out	0.02	31.64	0.06	Ob.1	1	Out	0.03	4.09	0.08
Ob.1	2	NTZ	0.08	21.88	0.21	Ob.1	2	NTZ	0.10	12.42	0.27
Ob.1	2	Out	0.05	30.23	0.14	Ob.1	2	Out	0.06	10.15	0.15
Ob.2	1	NTZ	0.02	34.81	0.05	Ob.2	1	NTZ	0.01	8.18	0.03
Ob.2	1	Out	0.02	33.91	0.04	Ob.2	1	Out	0.11	3.65	0.28
Ob.2	2	NTZ	0.04	23.69	0.11	Ob.2	2	NTZ	0.08	10.80	0.19
Ob.2	2	Out	0.09	27.52	0.25	Ob.2	2	Out	0.06	9.81	0.14
Ob.3	2	NTZ	0.06	15.51	0.14	Ob.3	2	NTZ	0.05	7.33	0.12
Ob.3	2	Out	0.05	17.19	0.11	Ob.3	2	Out	0.02	9.96	0.05
Ob.4	2	NTZ	0.06	23.36	0.16	Ob.4	2	NTZ	0.17	9.22	0.48
Ob.4	2	Out	0.05	26.10	0.12	Ob.4	2	Out	0.05	8.51	0.13

3.3.1 Observer 1

The biodiversity and biodiversity performance of IS inside and outside the NTZ during two different phases were compared. In Phase 1, the biodiversity performance values (*asym*, *curv* and *Bioperf*), and the number of IS observed within the NTZ were higher than those outside the NTZ (Figure 3. 6 and Table 3. 2). However, in Phase 2, the opposite was observed, with the number of species observed outside the NTZ being higher compared to within. Despite this, the *Bioperf* and *curv* values remained higher within the NTZ (Table 3. 2). Comparing the phases, the number of IS recorded outside the NTZ remained similar in Phase 1 to Phase 2. However, the number of IS recorded within the NTZ in Phase 2 was lower than in Phase 1 (Figure 3. 6).

In the assessment of NIS, the number of species, and *asym* value were higher during Phase 1 within the NTZ than unprotected areas (Figure 3. 7). In Phase 2, similar results were observed. The number of NIS, and biodiversity performance values were higher within the NTZ than outside (Table 3. 2). In comparison of phases, there was an increase in the number of NIS, the *curv*, and *Bioperf* values, both within and outside the NTZ in Phase 2.

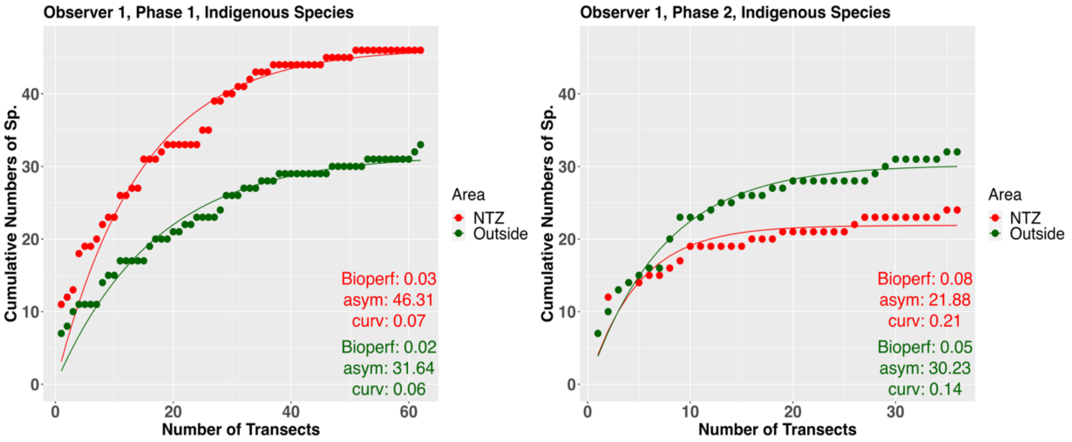

Figure 3. 6 Cumulative numbers of IS observed by Observer 1 in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 2

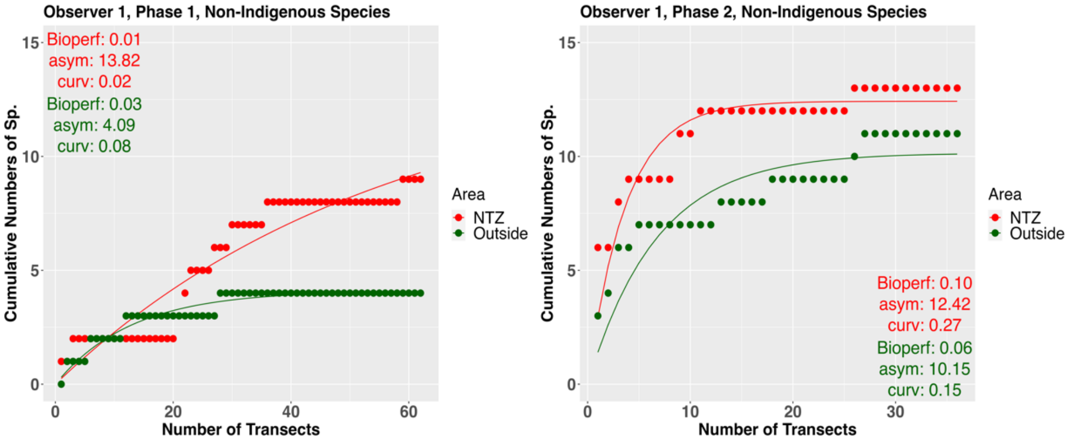

Figure 3. 7 Cumulative numbers of NIS observed by Observer 1 in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 2

P. oceanica and rocky substrate transect lines were compared across different phases and areas. For the *P. oceanica* transect of the assessment of IS, in Phase 1, the number of IS (Figure 3. 8), *Bioperf,* and *curv* values (Table 3. 3) recorded inside the NTZ were higher than those outside. In Phase 2, the opposite was observed, with a higher number of species recorded outside the NTZ. However, *Bioperf* and *curv* values remained higher inside the NTZ (Table 3. 3). When comparing phases, the number of IS observed in Phase 1 within the NTZ decreased in Phase 2, while the number of species observed outside increased (Figure 3. 8). Additionally, the *Bioperf* and *curv* values of both areas showed an increase in Phase 2.

For the rocky substrate transect, Phase 1 results showed higher number of IS, biodiversity performance values within the NTZ compared to outside (Figure 3. 8 and Table 3. 3). In Phase 2, the opposite was observed, with higher species numbers and *asym* values recorded outside the NTZ. However, *curv* and *Bioperf* values were still higher inside the NTZ. In comparing the phases, the results mirrored those obtained for the *P. oceanica* transect.

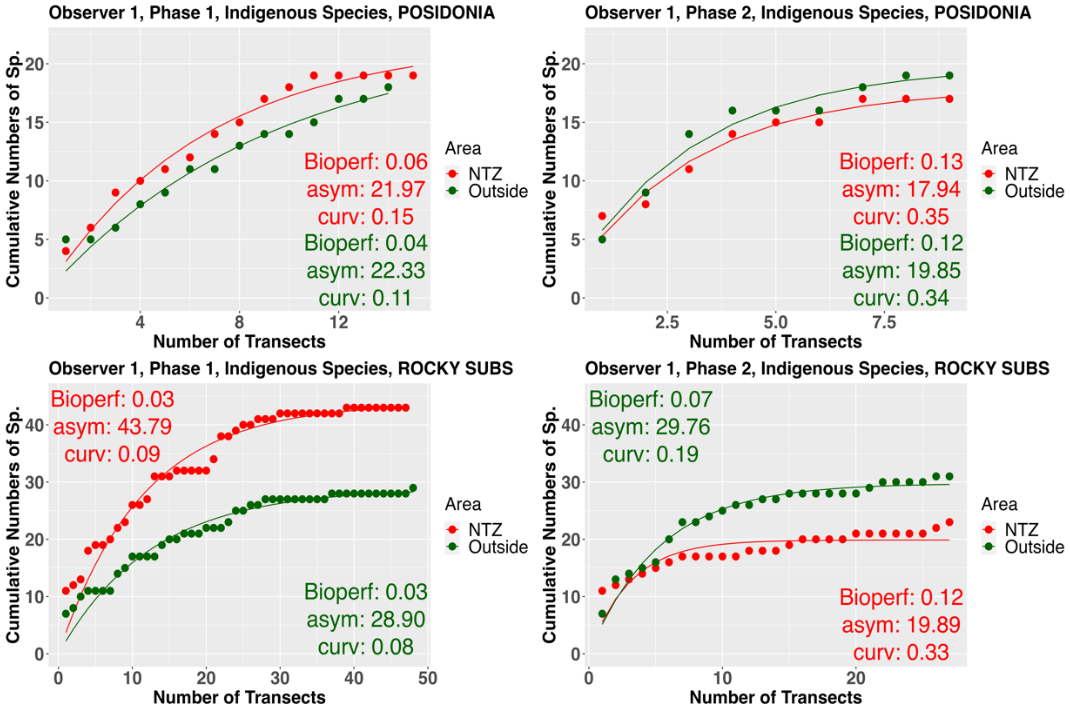

Figure 3. 8 Cumulative numbers of IS observed by Observer 1 on *Posidonia oceanica* transect (Top) and rocky substrate transect (Below) in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 3

In the assessment of NIS on the *P. oceanica* transect in Phase 1, a higher number of species and *asym* values were observed inside the NTZ than outside (Figure 3. 9 and Table 3. 4). However, *Bioperf* and *curv* values were higher outside the NTZ. In Phase 2, all biodiversity performance values, and NIS numbers were higher within the NTZ. When comparing phases, the number of NIS, all biodiversity performance values in both areas increased in Phase 2 compared to Phase 1.

For the rocky substrate transect, results similar to those for IS were obtained. In Phase 1, the number of NIS and *asym* values inside the NTZ were higher than outside

(Figure 3. 9 and Table 3. 4). Similarly, *Bioperf* and *curv* values were higher outside the NTZ. In Phase 2, the number of NIS and all values were higher inside the NTZ compared to outside. When comparing phases, the results were similar to those observed for the *P. oceanica* transect.

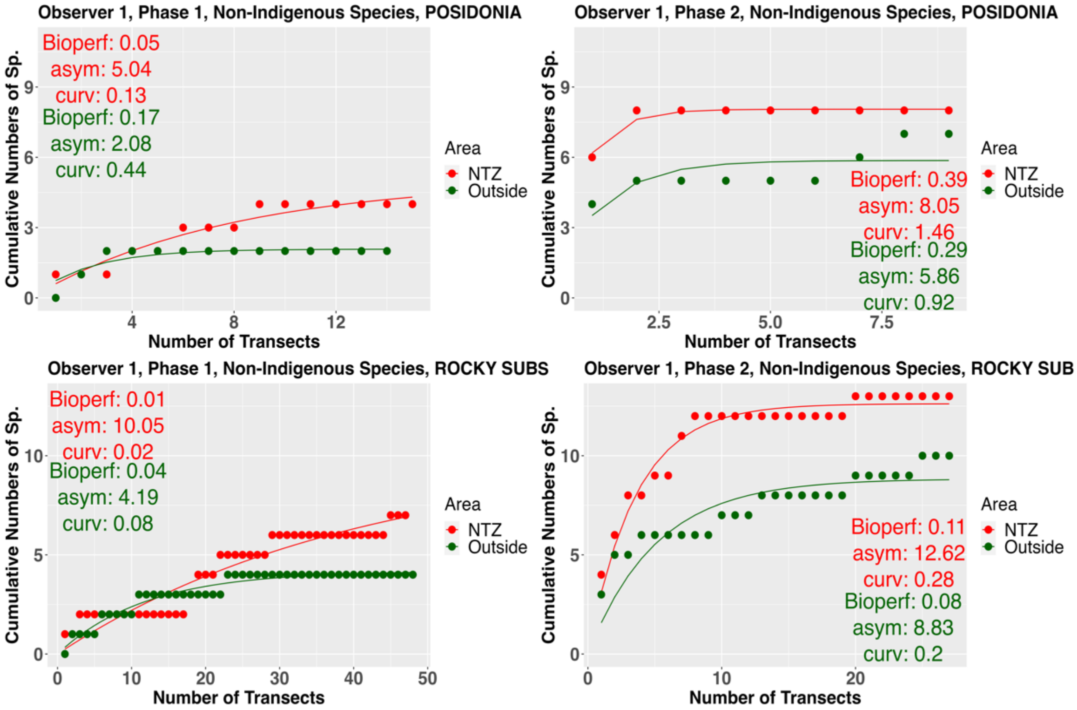

Figure 3. 9 Cumulative numbers of NIS observed by Observer 1 on *Posidonia oceanica* transect (Top) and rocky substrate transect (Below) in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 4

Table 3. 3 Biodiversity performance values of indigenous species according to transects.

Indigenous Species											
***Posidonia oceanica* Transect**						**Rocky Substrate Transect**					
Observers	**Phases**	**Region**	***Bioperf***	***asym***	***curv***	**Observers**	**Phase**	**Region**	***Bioperf***	***asym***	***curv***
Ob.1	1	NTZ	0.06	21.97	0.15	Ob.1	1	NTZ	0.03	43.79	0.09
Ob.1	1	Out	0.04	22.33	0.11	Ob.1	1	Out	0.03	28.90	0.08
Ob.1	2	NTZ	0.13	17.94	0.35	Ob.1	2	NTZ	0.12	19.89	0.33
Ob.1	2	Out	0.12	19.85	0.34	Ob.1	2	Out	0.07	29.76	0.19
Ob.2	1	NTZ	0.06	18.12	0.16	Ob.2	1	NTZ	0.03	32.58	0.07
Ob.2	1	Out	0.06	20.06	0.15	Ob.2	1	Out	0.03	28.70	0.08
Ob.2	2	NTZ	0.11	17.17	0.30	Ob.2	2	NTZ	0.05	23.79	0.13
Ob.2	2	Out	0.12	16.99	0.32	Ob.2	2	Out	0.13	26.43	0.37
Ob.3	2	NTZ	0.13	9.40	0.34	Ob.3	2	NTZ	0.09	13.99	0.24
Ob.3	2	Out	0.25	9.23	0.77	Ob.3	2	Out	0.05	17.80	0.14
Ob.4	2	NTZ	0.16	17.69	0.45	Ob.4	2	NTZ	0.09	21.28	0.24
Ob.4	2	Out	0.10	17.45	0.27	Ob.4	2	Out	0.06	24.71	0.15

3.3.2 Observer 2

In Phase 1, the number of IS and all the biodiversity performance values were not different between areas (Figure 3. 10 and Table 3. 2). In Phase 2, consistent with Observer 1's results (Figure 3. 6), the number of IS was higher outside the NTZ. A higher value of *Bioperf*, *asym*, and *curv* was also seen outside the NTZ (Table 3. 2). Comparing the phases, the number of IS observed during Phase 1 in both areas showed a minor decrease in Phase 2. Despite this decreasing, *Bioperf* and *curv* values increased in Phase 2.

In the analysis of NIS, the species numbers were higher within the NTZ during Phase 1 (Figure 3. 11). Conversely, the *Bioperf* and *curv* values were higher outside the NTZ (Table 3. 2). In Phase 2, the NIS values within the NTZ exhibited higher numbers and biodiversity performance values than outside. In addition, a comparison of phases revealed that the NIS numbers and all biodiversity performance values observed during Phase 1 increased in Phase 2.

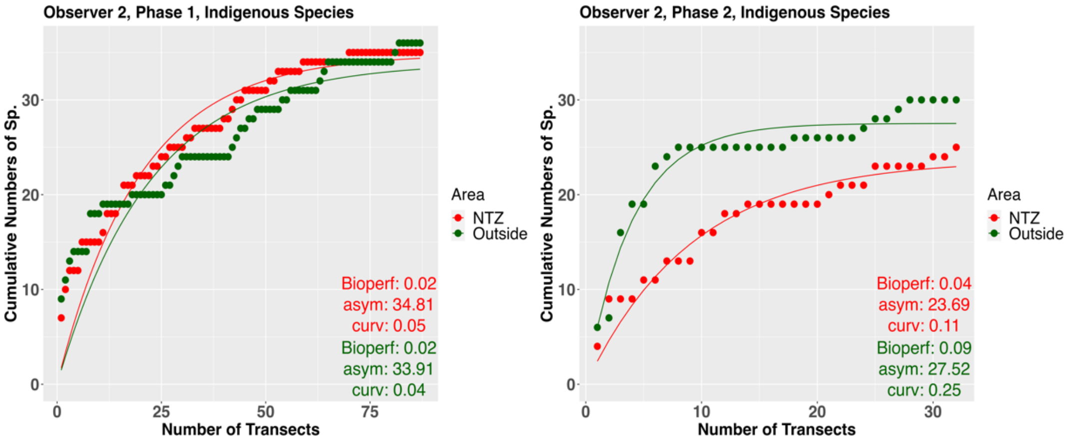

Figure 3. 10 Cumulative numbers of IS observed by Observer 2 in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 2

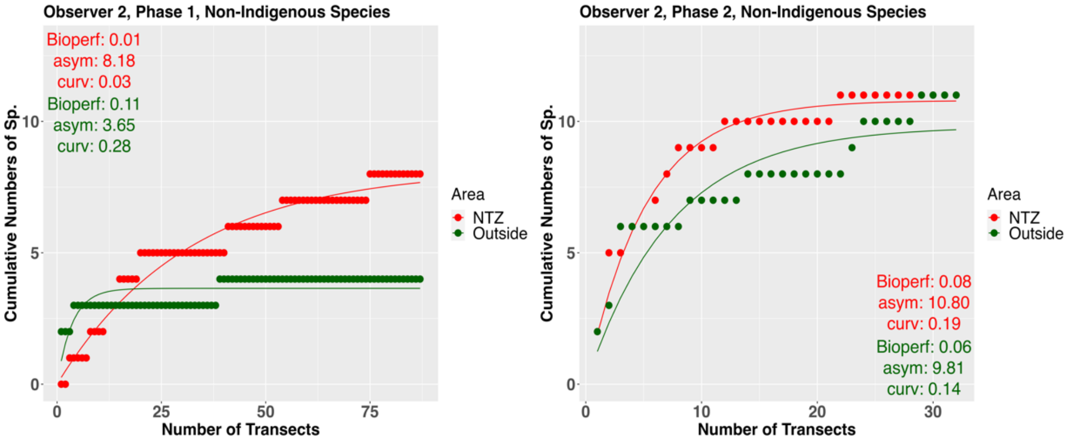

Figure 3. 11 Cumulative numbers of NIS observed by Observer 2 in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 2

Transect lines were compared across different phases and areas. In the *P. oceanica* transect, outside the NTZ displayed a higher number of IS and *asym* values than inside during Phase 1. However, both areas showed similar *Bioperf* and *curv* values (Figure 3. 12 and Table 3. 3). In Phase 2, the species numbers and biodiversity performance values were nearly close in both areas. When comparing the phases, the number of IS in both areas decreased from Phase 1 to Phase 2. Nonetheless, *Bioperf* and *curv* values increased.

On the rocky substrate, during Phase 1, there were higher counts of IS and *asym* values inside the NTZ compared to outside. However, the *Bioperf* and *curv* values showed little difference between the areas (Figure 3. 12 and Table 3. 3). In Phase 2, the trend reversed, with higher numbers of IS and higher biodiversity performance values outside the NTZ. When comparing the phases, the number of species within the NTZ in Phase 1 decreased in Phase 2, while the number outside remained unchanged (Figure 3. 12). Additionally, the biodiversity values in both areas increased in Phase 2.

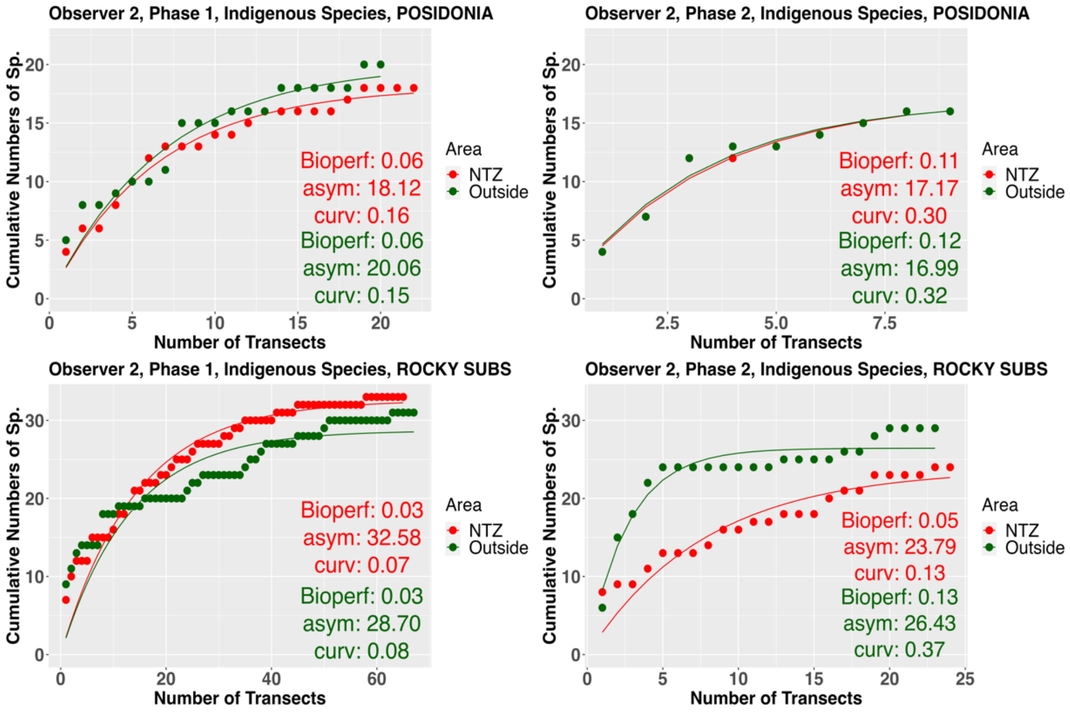

Figure 3. 12 Cumulative numbers of IS observed by Observer 2 on *Posidonia oceanica* transect (Top) and rocky substrate transect (Below) in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 3

In the assessment of NIS on the *P. oceanica* transect in Phase 1, a higher number of NIS were observed inside the NTZ compared to outside (Figure 3. 13). However, *Bioperf* and *curv* values were higher outside the NTZ (Table 3. 4). The same results were obtained in Phase 2. The assessment of both phases showed an increase in the number of species and all biodiversity performance values in Phase 2 compared to Phase 1 in both areas.

For the rocky substrate, similar results to those observed for the *P. oceanica* transect were obtained. In Phase 1, the number of species and *asym* values were higher inside the NTZ, while *Bioperf* and *curv* values were higher outside (Figure 3. 13 and Table

3. 4). In Phase 2, however, the number of species and all biodiversity performance values were higher inside the NTZ. Additionally, when comparing the phases, all values observed inside the NTZ increased in Phase 2 compared to Phase 1. Outside the NTZ, the number of species increased in Phase 2, while *Bioperf* and *curv* values decreased.

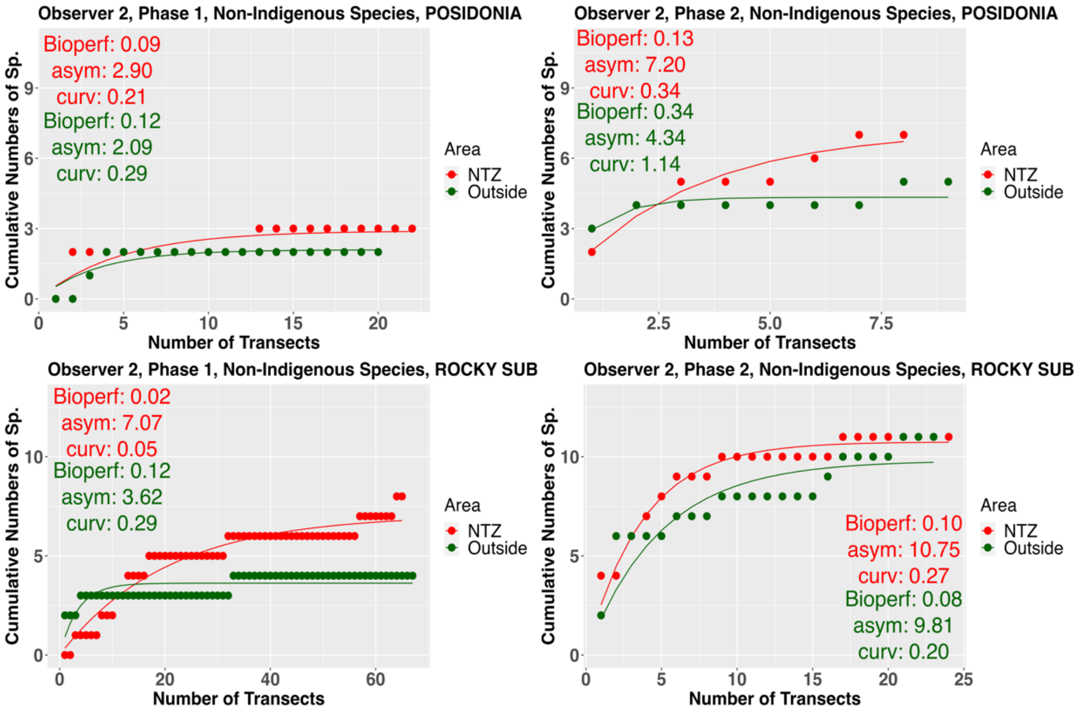

Figure 3. 13 Cumulative numbers of NIS observed by Observer 2 on *Posidonia oceanica* transect (Top) and rocky substrate transect (Below) in Phase 1 (Left) and Phase 2 (Right). The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 4

Table 3. 4 Biodiversity performance values of Non-Indigenous Species according to transects.

Non-Indigenous Species											
***Posidonia oceanica* Transect**						**Rocky Substrate Transect**					
Observers	**Phases**	**Region**	***Bioperf***	***asym***	***curv***	**Observers**	**Phases**	**Region**	***Bioperf***	***asym***	***curv***
Ob.1	1	NTZ	0.05	5.04	0.13	Ob.1	1	NTZ	0.01	10.05	0.02
Ob.1	1	Out	0.17	2.08	0.44	Ob.1	1	Out	0.04	4.19	0.08
Ob.1	2	NTZ	0.39	8.05	1.46	Ob.1	2	NTZ	0.11	12.62	0.28
Ob.1	2	Out	0.29	5.86	0.92	Ob.1	2	Out	0.08	8.83	0.2
Ob.2	1	NTZ	0.09	2.90	0.21	Ob.2	1	NTZ	0.02	7.07	0.05
Ob.2	1	Out	0.12	2.09	0.29	Ob.2	1	Out	0.12	3.62	0.29
Ob.2	2	NTZ	0.13	7.20	0.34	Ob.2	2	NTZ	0.10	10.75	0.27
Ob.2	2	Out	0.34	4.34	1.14	Ob.2	2	Out	0.08	9.81	0.20
Ob.3	2	NTZ	0.64	3.15	2.97	Ob.3	2	NTZ	0.08	7.22	0.19
Ob.3	2	Out	0.08	4.79	0.19	Ob.3	2	Out	0.02	10.41	0.05
Ob.4	2	NTZ	0.16	8.19	0.44	Ob.4	2	NTZ	0.30	7.00	0.98
Ob.4	2	Out	0.26	4.03	0.76	Ob.4	2	Out	0.06	7.69	0.14

3.3.3 Observer 3 and Observer 4

The observations of Observer 3 and Observer 4 were collected solely during a single phase, reflecting the results of Phase 2. Consequently, the evaluations and comparisons were only made between the NTZ and unprotected areas.

In the Observer 3 and Observer 4's results, a higher number of IS was observed outside NTZ than within (Figure 3. 14). However, except for the *asym* value, other biodiversity performance values were higher within the NTZ (Table 3. 2). For NIS, a higher number of species and biodiversity performance values were observed within NTZ than outside (Figure 3. 15 and Table 3. 2).

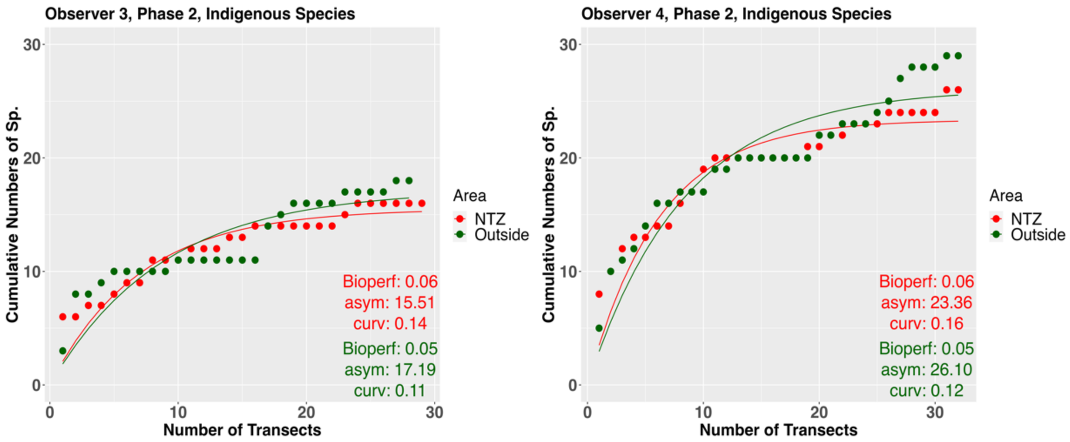

Figure 3. 14 Cumulative numbers of IS observed by Observer 3 (Left) and Observer 4 (Right) in Phase 2. The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 2

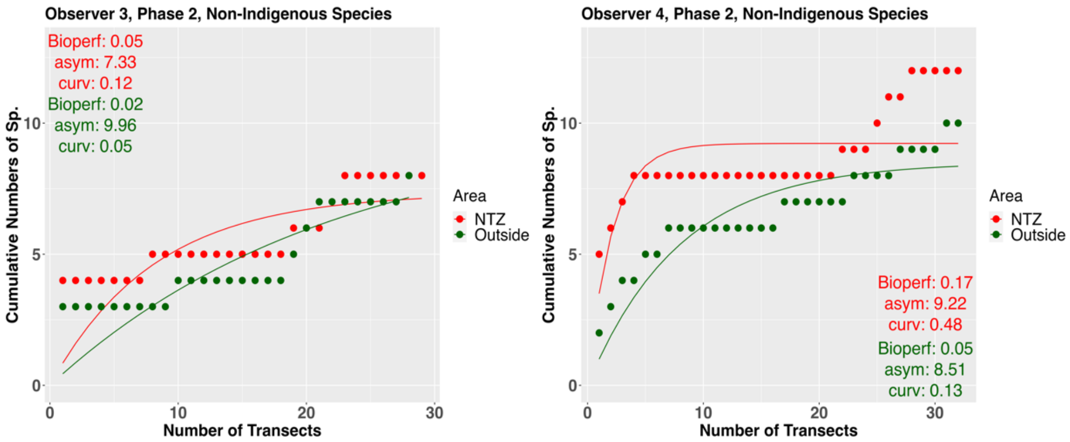

Figure 3. 15 Cumulative numbers of NIS observed by Observer 3 (Left) and Observer 4 (Right) in Phase 2. The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 2

Two transect lines were compared based on the areas. Observer 3’s results for the *P. oceanica* transect indicated that the number of IS and all biodiversity performance values were higher outside the NTZ (Figure 3. 16 and Table 3. 3). Similarly, in the rocky substrate transect, a higher number of species and *asym* values were observed outside the NTZ. However, *Bioperf* and *curv* values were higher within the NTZ.

In contrast, Observer 4’s results for the *P. oceanica* transect showed that the number of IS and all biodiversity performance values were higher inside the NTZ compared to outside (Figure 3. 17 and Table 3. 3). For the rocky substrate, the opposite trend was observed: the number of IS and *asym* values were higher outside the NTZ, while *Bioperf* and *curv* values were higher inside the NTZ.

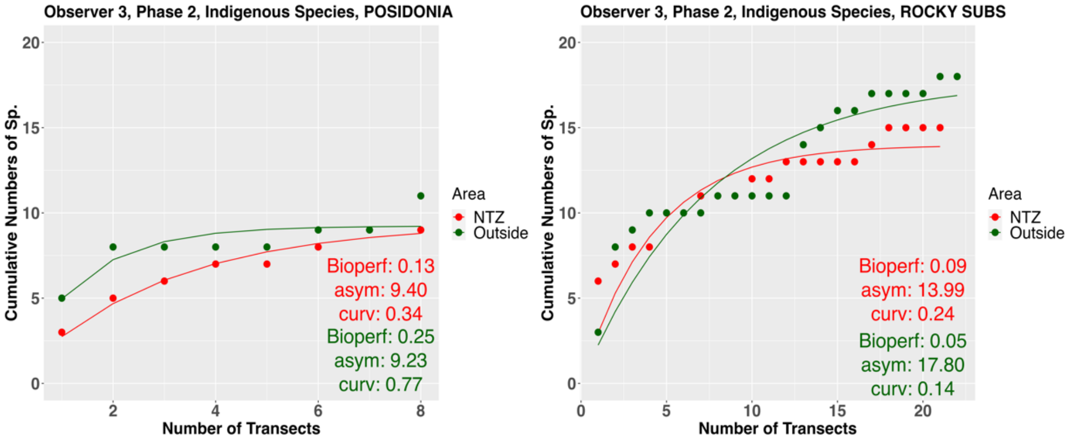

Figure 3. 16 Cumulative numbers of IS observed by Observer 3 on *Posidonia oceanica* transect (Left) and rocky substrate transect (Right) in Phase 2. The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 3

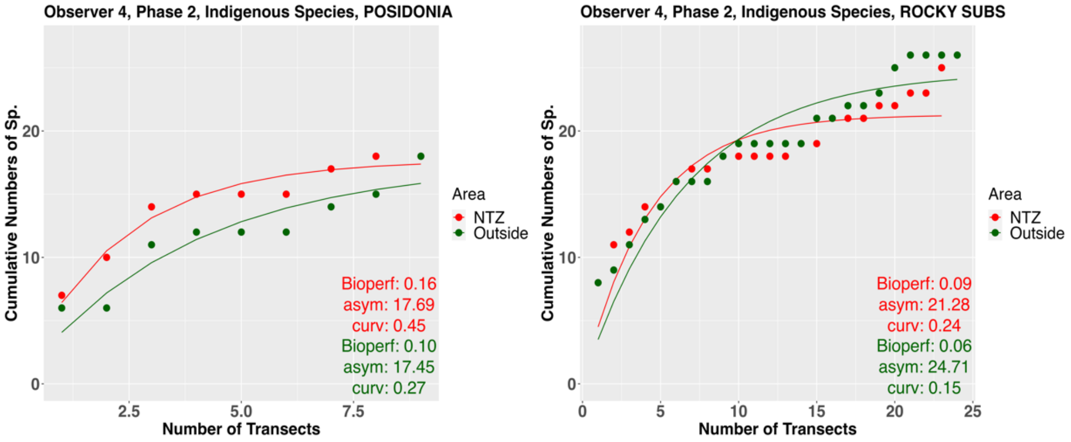

Figure 3. 17 Cumulative numbers of IS observed by Observer 4 on *Posidonia oceanica* transect (Left) and rocky substrate transect (Right) in Phase 2. The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 3

Based on the assessment for NIS, Observer 3's *P. oceanica* results indicated that the number of NIS and *asym* values were higher outside the NTZ (Figure 3. 18). However, *curv* and *Bioperf* values were higher inside the NTZ (Table 3. 4). In the rocky substrate transect, the number of NIS observed inside and outside the NTZ was similar, but *Bioperf* and *curv* values were higher inside the NTZ.

Observer 4's results for the *P. oceanica* transect showed a higher number of species inside the NTZ compared to outside (Figure 3. 19), but *Bioperf* and *curv* values were higher outside the NTZ (Table 3. 4). For the rocky substrate transect, the results were similar to those obtained in Observer 3's results.

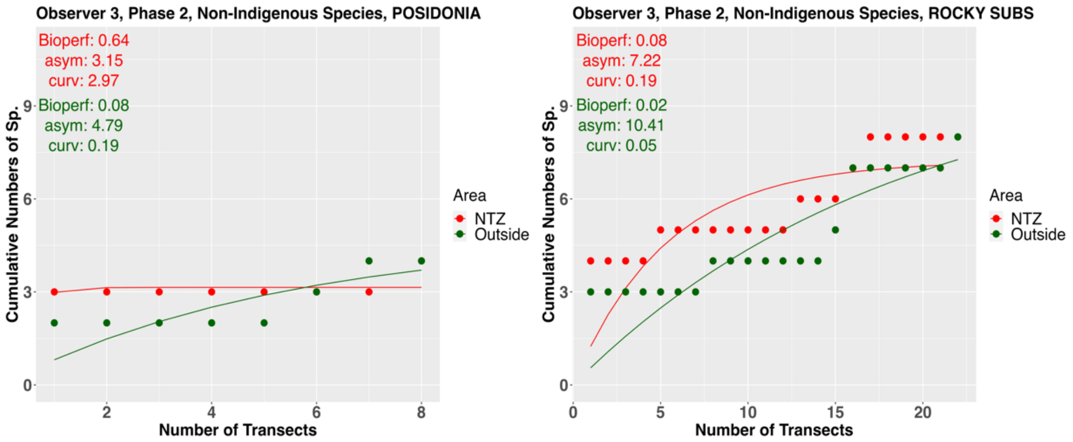

Figure 3. 18 Cumulative numbers of NIS observed by Observer 3 on *Posidonia oceanica* transect (Left) and rocky substrate transect (Right) in Phase 2. The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 4

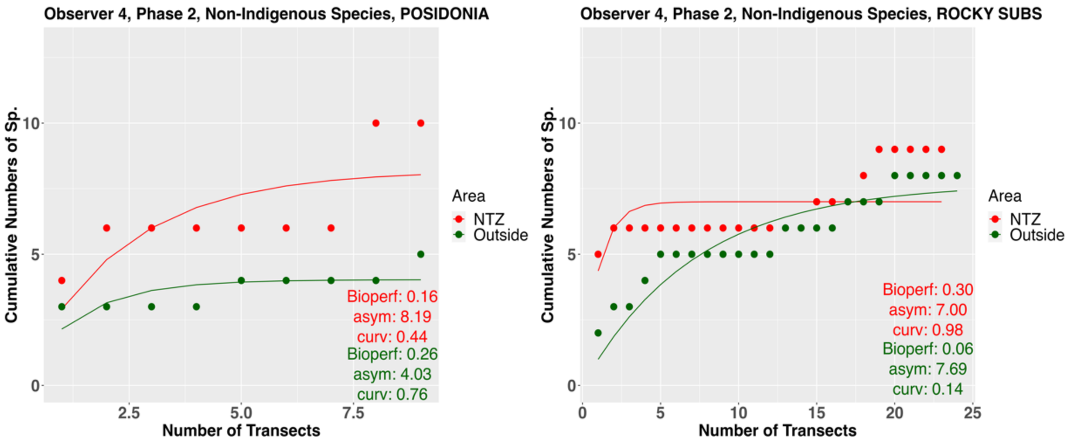

Figure 3. 19 Cumulative numbers of NIS observed by Observer 4 on *Posidonia oceanica* transect (Left) and rocky substrate transect (Right) in Phase 2. The red color represents inside the NTZ, and the green color represents outside the NTZ. Dots indicate the number of species observed, and lines represent the estimated average number of species. Biodiversity performance values are provided in the corner of the graphs and in the Table 3. 4

3.4 Discussion

Considering that inside and outside areas of the NTZ were previously subjected to similar pressures and are located close to each other, higher IS diversity within the NTZ observed two years after the establishment of the NTZ indicates that IS were more successful in the protected area. The success of IS was also reflected in their high percentage of occurrence in the sampled transects. Hence, NIS were not as successful within the NTZ as they were in unprotected areas (indicated by low Bioperf value), probably due to the high diversity of IS during these years. Their low success can be an indication that NIS experienced difficulties in ecosystems in which IS are more dominant in terms of species diversity (Stachowicz et al., 2002).

On the other hand, observations conducted 20 years later revealed a starkly different scenario from the initial findings. The disappearance of previously present species and the increase in NIS and their emergence as the most frequently observed species (Figure 3. 4 and Figure 3. 5) shows the rapid ecological change occurring in the

Eastern Mediterranean Sea (Galil, 2007). Although this situation might be perceived as an increase in biodiversity, the newly arriving NIS pose a threat (Galil, 2007), indicating that protection efforts in coastal areas in this region have not effectively preserved indigenous fish species and the native ecosystem dynamics.

The ineffectiveness of these efforts has also been apparent in the changes in IS diversity. The decrease in IS diversity within the NTZ contradicted the expectations that the effects of conservation measures would increase over time and that protected areas would harbor higher species diversity than unprotected areas. While it is commonly expected that protected areas have higher species richness (Côté et al., 2001; Halpern, 2003; Lester et al., 2009; Guidetti et al., 2014), a study conducted in the Red Sea found no significant difference between areas where fishing is allowed and those where it is banned (Roberts & Polunin, 1992). Also, Rogers & Beets (2001) determined no significant differences in fish species richness between the areas inside and outside the protected area (Caribbean). Similarly, Watson et al. (1996) reported that fishing pressure did not impact the species diversity between fished and non-fished areas along the Kenyan coast (Indian Ocean), consistent with the presented results.

The observed decrease in IS richness in this study can be attributed to the NIS in the NTZ. It is known that the complex trophic interactions often influence how communities respond to protection efforts, causing species diversity to change in various ways (Willis & Anderson 2003; Takashina et al., 2012). For that reason, the changes in trophic interactions over the years following the introduction of NIS into the coastal infralittoral ecosystem might have contributed to the decline in IS diversity within the NTZ. Additionally, the shallow infralittoral zones are recognized as a nursery area for numerous species, including NIS. In these shallow zones, species from the *Sparidae*, *Serranidae*, and *Labridae* families, typically associated with the littoral zone, concurrently share this area, and probably to reduce competition, Labrids and Sparids recruit at different times of year in the northwest Mediterranean (García-Rubies & Macpherson, 1995). Considering these dynamics and the disappearance of *Serranus cabrilla* (*Serranidae* family) and *Sarpa salpa* (*Labridae* family) within the NTZ in recent years, it can be linked that the habitat dynamics shared by IS in the NTZ have been disrupted by the increase of NIS in this area, negatively affecting their success and diversity.

The constant increasing trend in NIS richness in both areas was particularly caused by the rising Mediterranean seawater temperatures in recent times, which accelerated the entry of these species into the Mediterranean (Raitsos et al., 2010; Mavruk et al., 2017). In comparing the areas, high NIS richness and performance were seen within the NTZ. These outcomes contrast with the findings of Mumby et al. (2011) in the Bahamas, Hackerott et al. (2013) in the Caribbean, Malherbe & Samways (2014) in southern Africa, and Giakoumi & Pey's (2017) global review, which all reported that protected areas had a deterrent effect on the success of NIS. This effect was not observed in the present study conducted in the Eastern Mediterranean. Parallel results were obtained from other studies performed in the Eastern Mediterranean's MPAs. In the study by Kleitou et al. (2024), it was found that the protected area did not have a suppressive effect on NIS and even increased the abundance of the invasive species *Pterois miles*. At the onset of the conservation measures, the percentage of occurrence of NIS within the NTZ indicated that the area was already experiencing pressure from NIS. The health state of the ecosystem and the presence of NIS that had entered the coastal ecosystem before the establishment of the NTZ may have given these species an advantage for their introductions. Prohibiting fishing in an ecosystem already under pressure from NIS might inadvertently have created favorable conditions for NIS. On the other hand, NIS' lower numbers and performance in unprotected areas compared to the protected area can be attributed to small-scale commercial or sports fishing activities that help control their populations. NIS like *Siganus luridus* and *Siganus rivulatus*, targeted by the local anglers and small-scale fishers deploying traps, have gained economic value and entered the fish markets in the Eastern Mediterranean in recent years (EastMed, 2010). Fishing for these species in unprotected areas might be helping to control their numbers. Without such control within the NTZ and with the catalyzing effect of climate change and warming (Raitsos et al., 2010; MedECC, 2020), these NIS might use protected areas as shelters and have a possibility to increase their population success and disrupt the natural ecosystem dynamics, such as the sequential recruitment of indigenous species over the infralittoral habitats, which have evolved over millions of years, thereby suppressing the IS.

Following the general assessment, observations on seagrass (*Posidonia oceanica*) and rocky substrate habitats were evaluated to understand the impact of conservation efforts on species diversity over time in habitats with different structures and associated species. It was observed that IS and NIS in these habitats were affected differently. For IS in the seagrass habitat, although their diversity declined, their performance (Bioperf) increased over time, but this increase was similar in both protected and unprotected areas. This lack of difference suggests that IS associated with the seagrass habitat have not been affected by the protection measures, whereas IS in the rocky substrate habitat was affected positively. Conversely, the opposite trend was observed for NIS. NIS associated with the seagrass habitat responded better to conservation measures compared to those in the rocky substrate habitat. These contrasting results clearly indicate that the success of NIS in the seagrass habitat has negatively affected IS and prevented them from benefiting from conservation measures. Since *P. oceanica* beds are known as crucial areas for settlement, nursery, and feeding for numerous species across all trophic levels, playing a vital role in fulfilling these needs (Guidetti, 2000; Appolloni et al., 2023), the decrease in IS richness in these areas has become a significant concern for the ecosystem of the region. In a study conducted by Kalogirou et al. (2010) around the coast of Rhodes, on the farther west of the present study area, high biodiversity was recorded on *P. oceanica* in an area without any protection status, with IS contributing more to this recorded biodiversity. Although these results are consistent with the findings observed outside the protected area in this study, higher IS diversity was expected within the NTZ. Instead, the opposite outcome was observed, which can again be attributed to NIS. Most NIS depend on *P. oceanica* meadows during their early stages, emphasizing the importance of this habitat not only at the time of their initial arrival but also throughout their establishment process (Kalogirou et al., 2010). *Siganidae* species, which use *P. oceanica* beds for juvenile migration (Kalogirou et al., 2010), might have benefited from the fishing ban, thereby negatively affecting IS associated with this habitat.

3.5 Conclusion

The findings of the present study revealed that eliminating fishing pressure on NIS in the coastal infralittoral zone through protection measures inadvertently created favorable conditions for NIS. This allowed them to use the protected area as a shelter, leading to their increased success and diversity within the NTZ. Consequently, their success disrupted the established habitat dynamics of IS, negatively affecting IS diversity and preventing them from benefiting from the protection measures. These results indicate that protection efforts can inadvertently advantage NIS in ecosystems already under NIS pressure. The impact of fishing on NIS and the potential of utilizing fishing as a tool to slow down the process of change in the Eastern Mediterranean ecosystem need to be investigated in more detail to understand how fishing activities can control NIS populations. Such knowledge would provide valuable insights into managing and mitigating the rapid ecological changes in the region.

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CHAPTER 4

GENERAL CONCLUSION

This thesis was performed to search for a response to how Marine Protected Areas (MPAs) affect the ecosystem, particularly non-indigenous species (NIS), in areas under intense NIS pressure, such as the Eastern Mediterranean. Within this scope, the hypothesis that MPAs not only contribute to biodiversity conservation and the recovery of fish populations in degraded ecosystems but also have the potential to mitigate the spread of NIS was tested in two different habitats with different protection levels. First, the state within the FRA, where only trawl bans were enforced, was focused. Commercially exploited fish populations within the FRA showed a relatively prompt response to the conservation measures, and the recovery in their biomass has been observed. Furthermore, the NIS population within the FRA could not show success in terms of biomass and proportion. Protection measures have also slowed the increase of NIS percentages in the FRA. These observed results indicated that, along with the positive effects of conservation measures on biomass, there was also a slowing effect on NIS, verifying the hypothesis that conservation measures have a negative impact on NIS.

The second focus was on the shallow infralittoral zone within the fully protected No-Take Zone (NTZ), where most of the population consists of juveniles and young-of-the-year fish. Results showed that although a high indigenous species diversity was observed at the beginning of the conservation in the NTZ, this favorable protection effect reversed, and indigenous species' diversity declined within the NTZ but remained stable outside. In contrast, NIS were more successful within the NTZ. The reduced success of indigenous species within the NTZ was attributed to the negative impact of NIS on indigenous species. Additionally, the relatively lower number of NIS outside the NTZ was linked to fishing activities targeting NIS in these areas, thereby controlling their populations. These observed results within the NTZ did not support the hypothesis that MPAs promote biodiversity and effectively control and mitigate NIS. The difference between these two cases arose from the fact that the targeted fish groups represented different parts of the population and the varying levels of protection measures within the areas. Reduced fishing pressure on the fishing grounds, consisting mainly of adult fish and commercially exploitable species, supported these populations in terms of biomass and resilience. However, it has been observed that the prohibition of fishing to protect sensitive essential habitats, such as nursery and recruitment areas, inadvertently benefits NIS. This indicates that the Eastern Mediterranean ecosystem's resilience to NIS may be critically vulnerable in these coastal areas, a vulnerability that may be intensified by the warming Mediterranean waters.

In conclusion, these observed results in areas where fishing is not allowed (No-Take Zone) and restricted (within FRA) demonstrate that the impact of conservation efforts on the NIS may vary depending on the habitat structure being protected, the structure of fish groups within the protected area (juvenile or adult fishes), and the level of protection measures implemented (fully protected or partially protected). These results may also suggest that in regions under intense NIS pressure, such as the Eastern Mediterranean, fishing may be used as a tool to slow down the process of change caused by NIS. Additionally, as stated at the beginning of this study, the primary reason for the protection of the Kızılliman area was to safeguard the breeding caves and feeding grounds of monk seals. However, if a Marine Protected Area is to be established in the Mediterranean, especially in the eastern part, with the sole goal of increasing marine biodiversity or recovering fish stocks, it is crucial to consider the impact of NIS in the planning process.

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APPENDICES

A. The results of species-specific assessment

Table A1. The results of species-specific assessment conducted outside the FRA over the years. The species codes and their corresponding scientific names are in the Table A3 below. Biomass values are represented using symbols: (+) for biomass between 1-10 kg, (++) for 10-100 kg, and (+++) for 100-800 kg.

Years	**1983**	**1984**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2014**	**2019**	**2022**	**2023**
Species																			
LSJ																	+		
OGF															+	++	+	+	+
QSM																	++	+	
QCN						+			+		+	+					+		
MGR																	+		
RLI																			

CMK	CTZ	CSP	HMY	YBR	OUB	BOG	NUO	TRG	RNH	MSF
					++		+	+		++
	++				++	+	+			++
	+					+		+		
					+					
	+				+	+		+		
+	+				+			+		
++					+	+				
+	++				++					+
	++				+++		+		+	+
+	++				+++	+	+	++		+
+	++			+	++	+	+		+	+
+	+				+	+		+	+	+
+	+		+	+	++	+				+
	+	++			+	+		+		
	+	+			++	+		+		
	+				++					
+	+	+	+		+	++		+		
	+	+			+	++				
	+	+			++	+				

SWA	ANN	DEL	DEP	DEC	DEQ	DYL	YOC	COU	COE	CIL
				+						++
++	+	+	+	++						++
	++		+			+				
+	++		++			+				+
+	++		+			+			+	
++	++					+				+
+	+					+				
++	++			+		+				+
+	+		+	+			+			++
++	+		+		+	+				++
++	++		+					+		++
+	++		+	+	++					+
+	++		+	+						+
+	++		+							+
+	++		+							++
				+						+
+	++		++					+		
+	+		+							+
										+

GTH	GBN	FIP	FIO	RRH	KZK	EPK	GPW	ANE	AOM	CTB
	+						++			
							+			++
										++
							++			++
			+			+	++			++
			+				+			++
			+							+
			+			+	+			+
			+				++			+
	+		++				++			++
	+		+				+			+
			++				++			
			+			+	++			++
			+				++	+		
							++			
++			++				++	+		++
		++	+				++			++
		++					++			

MMH	MUR	MUT	HKE	SNS	SSB	LDV	PON	SUI	KZP	LFZ
	+	++	+		+	++	++		+	
	+	+++	++	+		+++	++			
	++	++								
		++								
+	+	+								
+	++	++								
+	++	++								
	+	++					+			
+	+	++			+	+		+	+	
	+	++	+		+	+	+	+++	+	
	+	++			+	+	++	++		+
++	+	++			+	+	++	++	+	++
	+	++			++	+	++	++	+	+
		++			++		++	++	+	+
	+	++			++	+	++	++	+	+
	+	++				+		++		
	++	++		+	++	+	++	++		+
+		+++				+	++	++		
		++			+	+	++	++	+	

TRI	PKS	BGR	PII	RPF	RPG	REA	PAC	SBA	NYW	NNZ
					++		++			
		+			++		++			
					++		++	+		
+		++			+		++			
					+		++			
		++			+		++	+		
		++					++			
		+			+		+			
					+		++			
+					+		+++	+		
++		+			++		++	+		
+							++			
					+	+	++	+		+
		+					++	+		+
				+			++			++
							++	+		++
	+++			+	++		++	+++		++
+	+++			++			++	++	+	++
+	+++			++	+		++	+		++

RSE	BBS	SNQ	VMA	LIB	HWH	SAE	SAA	REO	SHR	PZO
	++			+++						
+	+	++		+++				+		
+	+	+		+						
++				++					+	
+	+			+						
++	+	+		++						
+	+			++						
+		+		++						
+	+			+++						
+	+			++						
+	++			+++						
+		+		++						
+	+	+		++						
				++						
				+++				+		
				++						
	++		++	++	+					+
+	+	+		++	+	+	+			+
				++	+					+

YRS	YRC	SBG	PRR	SOL	SRI	IGU	SRK	SRJ	CBR	AMB
		+		++				++		
		++		+	+			+	++	
								+	++	
		+			+		+++		+	
			+		+	++	++		++	
		+	+		+	+	++	+	++	
		+				+	++	+	+	
		+	+		+	+	++	+	+	
					+		+	+	++	
		+	++		++		+	++	++	
		+	+		++		+	++	++	
					++		++	++	++	
		+	+		++	+	++	++	++	
+	++	+						+	++	
		+						+	++	
								+	++	
		+	+			++		++	+	++
+	+	+	+		+		++	+	+	
	+	+			++			+	+	

QFF	SDR	TIN	ROS	ROI	MED	KOY	BRB	SPC	BPI	FLE
	+					++			+	
	+					++			++	
								++		++
+		+				++				
					+	++		++	+	+
	++				+	++				
						++		+		
+		+				++				
	+					++				+
	++					+		+		++
	+					++		++		+
						+			+	+
		+			+	+		++	+	++
								+		+
+	+					+				+
+		+				+	+	++		+
+						+				+
+	+					+				+

JOD	XYN	UUC	UPH	UPM	GUU	LHT	HOM	HMM	WEG	TZA
+		++		+	++		+		++	
+	+	++	+	++	+				+	
			+	++						
+				+					+	
			++	+						+
			++	+						+
				+						+
+		+	+		+				+	
+		+	+		+				+	
		+	++	+	+				+	+
+		+	+++	+	+				+	
		+	++	++			+			
			+++	++				+		
			++	++				+		
		+	++	++				+		
		+	+++	++						
		+	+	+		+		++		
		+	+	++		+		+		
			++	++		+		+		

Table A2. The results of species-specific assessment conducted inside the FRA over the years. The species codes and their corresponding scientific names are in the Table A3 below. Biomass values are represented using symbols: (+) for biomass between 1-10 kg, (++) for 10-100 kg, and (+++) for 100-800 kg.

Years	1983	1984	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2014	2019	2022	2023
Species																			
LSJ																			
OGF																+			
QSM																			
QCN											+					+			+
MGR																			
RLI										+									

CMK	CTZ	CSP	HMY	YBR	OUB	BOG	NUO	TRG	RNH	MSF
					+		+			++
	++				+		+			+
+	+							+		
	+					+		+		
+	+				+	+		+		
+	+				+					
+	+				+	+				
++	++				++					
	++				++		+			
++	++		+		++	+	+		++	+
+	++				++	+	+		++	+
	+				+	++	++	+	+	+
	++				+	++		+	+	
	+				++	++			+	
	+	+		+	+	+			+	+
+	+				+	+			+	
	+	+			++	++				
	+				+	+				
	+				++	+				

SWA	ANN	DEL	DEP	DEC	DEQ	DYL	YOC	COU	COE	CIL
	+			++						++
	+	++	+	+						+
	++					+		+		
+	+		+			+				+
	+		+			+				
+	++			+		+		+		
+	+								+	+
+	+			+						+
+	+			+						
+	+		+						+	++
+	++		+	+				+		++
+	++		++							+
+	++		+	+						+
	+		+	+						+
	+									+
++	+++		+					+		+
			+++							+
										+
+	++							+		+

GTH	GBN	FIP	FIO	RRH	KZK	EPK	GPW	ANE	AOM	CTB
										++
										+
										+
						+	++			+
										+
			+							+
			+				+			+
			+			++				+
			+				+			+
			+							+
							++			+
			++	+			++			++
			++	+			++			+
							+			+
							++			
							++			+
			++		++		+++			+
		++		++	+		++			
		++	+		++		++			

MMH	MUR	MUT	HKE	SNS	SSB	LDV	PON	SUI	KZP	LFZ
	+	+++	+			++				
	++	+++	+	+		+	+			
+	++	++								
	++	+++				+				
++	++	+								
++	++									
+	+	+								
++	+	+								
	+	+				+				
	++	+++				++		+		
+	+	++				++	+	+		+
	++	+++				++	++	++		++
+	++	+++				+	+	++		
	+	+++				+	+	++		+
	+	+++				+	++	++		
	++	+++			+	++	+	+		
		+++				++	+	++		
+		+++				+	++	+		
	+	+++				+		+		

TRI	PKS	BGR	PII	RPF	RPG	REA	PAC	SBA	NYW	NNZ
					++		++			
					++	+	++	+		
					+		++			
					+		++			
					++		++			
					+		+			
					+		++			
+					++		++			
+					+		++			
+					+		++	++		
++					+		++	+		
+					+	++	++	++		
					+		+++	+++		
					+		+++	+++		+
							++	++		++
++							++	++		++
	+				++		++	++		+
+	+			+	+		++	++		+
+			+	++	+		++	++		++

RSE	BBS	SNQ	VMA	LIB	HWH	SAE	SAA	REO	SHR	PZO
	+			++						
		+		++						
	+			+						
+				+						
+	+									
+										
+				++						
+	++	+		++						
+	+			+						
+	+			++						
	++			++						
		+		++	+		+			
+	+	+		++						
+		+		+++			+			
				+++						
++		+		++						
			+	++						
				++						+
+		+		++						+

YRS	YRC	SBG	PRR	SOL	SRI	IGU	SRK	SRJ	CBR	AMB
								+		
					+				+	
						+	++		+	
					+	+	++		+	
					+	++	++		+	
					+	++	++	+	++	
					+	+	++	+	+	
					+	+	++	+	+	
+					+	++	+	+	++	
			+		++		+	+	++	
			+		++		++	++	++	+
+	+				++		++	+	++	
		+	+		+		++	+	++	+
+	+	+						+	++	
	+							+	++	
	+		++		+		+++	++	++	
								+	++	
					+		+	+	+	
		+	+		++		++	+	++	

QFF	SDR	TIN	ROS	ROI	MED	KOY	BRB	SPC	BPI	FLE
									++	
	+					++			++	
	++					+		+		++
+	+					++		+		+
	+					+		+		
	++				+	++				
	+					++		++		+
	+	+				++		+	+	+
	+					++				
	++				+	+		++	+	++
	+	+			+	++		+		++
						+		++	+++	++
	+					+		+++	++	++
	+							++		++
								++	+	++
			+	+		+	+	+		+
+								+		+
+	+					+		+		+
+	++		+		+	+		+		+

JOD	XYN	UUC	UPH	UPM	GUU	LHT	HOM	HMM	WEG	TZA
+		++					++			
+	+	+	+	++	+		+		++	
+		+	+						+	+
+		+	+	+					+	
			++						+	+
+		+			+				+	+
+		+	++	+					+	+
+		+	+		+				+	
+		+	+		+				+	+
+		+	++						+	+
+		+	++		+					+
		+	++	++			+			
			++	++				+	+	
			++	++		+		++		
			++	++				+		
		+	++	+						
			+	++				++		
			++	+				+		
			++	+				+		

Table A3. Families and scientific names of the analyzed species based on the Fishbase database (Froese & Pauly, 2024) and their corresponding species codes used in the species-specific biomass assessment.

Family	Species	Code	Family	Species	Code
Apogonidae	*Jaydia smithi*	QSM	Plotosidae	*Plotosus lineatus*	PII
	Ostorhinchus fasciatus	OGF	Pomacentridae	*Chromis chromis*	CMK
	Apogonichthyoides nigripinnis	QCN	Serranidae	*Epinephelus aeneus*	GPW
Balistidae	*Balistes capriscus*	TRG		*Epinephelus costae*	EPK
Blenniidae	*Blennius ocellaris*	NUO		*Serranus cabrilla*	CBR
Bothidae	*Arnoglossus imperialis*	RLI		*Serranus hepatus*	SRJ
	Arnoglossus laterna	MSF		*Serranus scriba*	SRK
	Arnoglossus thori	RNH	Scaridae	*Sparisoma cretense*	PRR
	Bothus podas	OUB	Sciaenidae	*Argyrosomus regius*	MGR
Callionymidae	*Callionymus filamentosus*	YBR	Scorpaenidae	*Pterois miles*	PZO
Carangidae	*Alepes djedaba*	LSJ		*Scorpaena notata*	SNQ
	Caranx rhonchus	HMY		*Scorpaena porcus*	BBS
	Seriola dumerili	AMB		*Scorpaena scrofa*	RSE
	Trachurus mediterraneus	HMM	Scombridae	*Scomber colias*	VMA
	Trachurus trachurus	HOM	Siganidae	*Siganus luridus*	IGU
Centriscidae	*Macroramphosus scolopax*	SNS		*Siganus rivulatus*	SRI

Champsodontidae	*Champsodon nudivittis*	CSP	Soleidae	*Solea solea*	SOL
Citharidae	*Citharus linguatula*	CIL	Sparidae	*Boops boops*	BOG
Congridae	*Conger conger*	COE		*Dentex dentex*	DEC
Cynoglossidae	*Cynoglossus sinusarabici*	YOC		*Dentex gibbosus*	DEP
Dactylopteridae	*Dactylopterus volitans*	DYL		*Dentex macrophthalmus*	DEL
Dorosomatidae	*Sardinella aurita*	SAA		*Diplodus annularis*	ANN
	Sardinella maderensis	SAE		*Diplodus sargus*	SWA
Dussumieriidae	*Etrumeus sadina*	RRH		*Diplodus puntazzo*	SHR
Echeneidae	*Remora remora*	REO		*Diplodus vulgaris*	CTB
Engraulidae	*Engraulis encrasicolus*	ANE		*Lithognathus mormyrus*	SSB
Fistulariidae	*Fistularia commersonii*	FIO		*Pagellus acarne*	SBA
	Fistularia petimba	FIP		*Pagrus auriga*	REA
Gobiidae	*Deltentosteus quadrimaculatus*	DEQ		*Pagellus erythrinus*	PAC
	Gobius niger	GBN		*Pagrus pagrus*	RPG
	Oxyurichthys papuensis	NYW		*Sparus aurata*	SBG
Haemulidae	*Pomadasys incisus*	BGR		*Spicara flexuosum*	FLE
	Pomadasys stridens	PKS		*Spicara maena*	BPI
Holocentridae	*Sargocentron rubrum*	HWH		*Spicara smaris*	SPC
	Coris julis	COU		*Spondyliosoma cantharus*	BRB
	Pteragogus trispilus	TRI	Sphyraenidae	*Sphyraena chrysotaenia*	YRC

Labridae	*Symphodus mediterraneus*	MED		*Sphyraena sphyraena*	YRS
	Symphodus roissali	ROI	Synodontidae	*Saurida lessepsianus*	LIB
	Symphodus rostratus	ROS	Tetraodontidae	*Lagocephalus guentheri*	GTH
	Symphodus tinca	TIN		*Lagocephalus sceleratus*	LFZ
	Xyrichtys novacula	XYN		*Lagocephalus spadiceus*	KZP
Leiognathidae	*Equulites elongatus*	KZK		*Lagocephalus suezensis*	SUI
	Equulites klunzingeri	PON		*Torquigener hypselogeneion*	QFF
Merlucciidae	*Merluccius merluccius*	HKE	Trachinidae	*Trachinus araneus*	TZA
Monacanthidae	*Stephanolepis diaspros*	KOY		*Trachinus draco*	WEG
Mullidae	*Mullus barbatus*	MUT	Triglidae	*Chelidonichthys lastoviza*	CTZ
	Mullus surmuletus	MUR		*Lepidotrigla cavillone*	LDV
	Parupeneus forsskali	RPF		*Chelidonichthys lucerna*	GUU
	Upeneus moluccensis	UMP	Trichiuridae	*Trichiurus lepturus*	LHT
	Upeneus pori	UPH	Uranoscopidae	*Uranoscopus scaber*	UUC
Muraenidae	*Muraena helena*	MMH	Zeidae	*Zeus faber*	JOD
Nemipteridae	*Nemipterus randalli*	NNZ			
Ophichthidae	*Echelus myrus*	AOM			