MOVE ON

Report for marine ecosystem condition and ecological reference status assessment methods in ORs and OCTs - Assessment of reference condition of coral reefs, mangroves, seagrass beds and kelps forests in overseas territories

Deliverable n° D.2.1.c

July 2023

Citation: MOVE-ON project (2023), European Commission Directorate General Environment Grant Agreement no. 07.027735/2019/808239/SUB/ENV.D2. Deliverable D.2.1.c – Report for marine ecosystem condition and ecological reference status assessment methods in ORs and OCTs - Assessment of reference condition of coral reefs, mangroves, seagrass beds and kelps forests in overseas territories.

Coordinated by:

GOVERNO DOS AÇORES
FRCT

Partners:

Leibniz Universität Hannover
Universidad Rey Juan Carlos
Nexa
Université de La Réunion
Biodiversidad Atlantica y Sostenibilidad
University of Portsmouth
WWF
IRD Institut de Recherche pour le Développement France
South Atlantic SAERI
University of Trento
Nova Blue Environment

Supported by:

This project has received funding from the European Union represented by European Commission Directorate General Environment under grant agreement N° 07.027735/2019/SI2.808239/SUB/ENV.D2. This document only reflects the views of its authors. The Commission is not responsible for any use that may be made of the information it contains.

Project Acronym	**MOVE-ON**
Project Title	**From case studies to anchor projects - setting the ground to advance MAES in Europe's overseas.**
Grant Agreement n°	07.027735/2019/808239/SUB/ENV.D2
Start of the project	May 2020
Duration	36 months
Project coordinator	Regional Fund for Science and Technology, Regional Government of the Azores (Portugal)
Website	www.moveon-project.eu

Deliverable title	D.2.1.c – Report for marine ecosystem condition and ecological reference status assessment methods in ORs and OCTs - Assessment of reference condition of coral reefs, mangroves, seagrass beds and kelps forests in overseas territories.
Deliverable n°	**D.2.1.c**
Activity title	Activity 2 - Method development and implementation support
Task title	Task 2.1 – Method development and assessment of marine coastal ecosystem reference Condition in EU Overseas territories
Task Leader(s)	URJC and NBE
Lead authors	Jean-Philippe Maréchal (NBE) – Ewan Trégarot (UOP)
Contributing authors	Cindy Cornet (UOP)
Due date of deliverable	04/2023
Actual submission date	28/07/2023
Dissemination Level:	Public

Version	**Status**	**Date**	**Author(s)**
1.0	Draft	18/07/2023	Maréchal (NBE) & Trégarot (UOP)
1.2	Revision	25/07/2023	FRCT, all partners
1.3	Final version	28/07/2023	FRCT

Summary

The concept of ecosystem condition reference is vital for assessing the current state of marine ecosystems, serving as a benchmark for evaluating their health, integrity, and functioning. Defining ecosystem condition reference requires practicality, scalability, and ecological relevance. Practical indicators that are readily available and easily measurable should be used for consistent assessments. Scalability ensures widespread adoption and data integration, fostering collaboration and reliability. Ecological relevance considers various ecosystem components and processes to understand functioning and resilience comprehensively.

In the MOVE-ON project, marine and coastal ecosystems in EU and UK overseas territories were studied, including coral reefs, seagrass meadows, kelp forests, and mangroves. Coral reefs are diverse ecosystems supporting marine biodiversity, fisheries, tourism, and coastal protection. Seagrass meadows are crucial for marine life, carbon sequestration, and water purification, but they face global threats. Kelp forests provide habitat, carbon storage, and coastal protection. Mangroves are unique trees offering coastal protection, biodiversity support, and livelihoods for local communities.

We emphasised the importance of defining ecosystem condition reference to understand changes and impacts on ecosystems over time. Historical data helps prevent the shifting baseline syndrome and serves as a reference point for future assessments, despite not always reflecting pristine conditions.

An appropriate baseline is essential for nature conservation, restoration, and management, particularly when assessing ecosystem services. Near pristine ecosystems in the European and British Overseas Territories serve as examples for conservation and references for nearby territories lacking historical data.

Data limitations in EU overseas territories pose challenges in assessing ecosystem condition. Traditional indicator-based approaches may not fully capture the complexity and resilience of marine ecosystems. To address this, integrating functional traits of species into assessment frameworks is suggested, providing valuable insights into ecosystem health even with limited data.

Developing indicators based on functional traits offers a more holistic understanding of ecosystem health, including information on species' roles and responses to environmental changes. Collaborations between scientists, policymakers, and local communities are crucial for comprehensive ecosystem assessments, incorporating traditional ecological knowledge to enhance understanding.

Innovative approaches and interdisciplinary collaborations are vital for sustainable management and conservation efforts in EU overseas territories' unique and valuable marine ecosystems, moving beyond traditional indicators. The assessment of ecological reference conditions is essential for effective management and conservation of these ecosystems.

Table of Contents

List of Figures

List of Tables

INTRODUCTION

Ecosystem condition reference refers to the baseline or reference state against which the current state of an ecosystem is assessed. It serves as a benchmark for evaluating the health, integrity, and functioning of marine ecosystems. Defining ecosystem condition reference is crucial for understanding the extent of changes and impacts on ecosystems over time.

Several considerations should be taken into account when defining ecosystem condition reference. Firstly, it should be practical, meaning that the indicators used to measure ecosystem condition should be readily available and easily measurable. This allows for efficient and consistent assessments across different regions and organisations.

Secondly, the definition should be scalable, which means it can be widely adopted and standardised. This allows for comparisons and integration of data from various sources, facilitating collaboration and improving the reliability of assessments.
Thirdly, the ecological relevance of the definition is essential. It should encompass the various components and processes that define an ecosystem, including physical characteristics, species composition, biological communities, and ecological interactions. This holistic approach provides a comprehensive understanding of ecosystem functioning and resilience.

In the absence of comprehensive baseline data, certain ecological reference points can be utilized. This may include data from efficient marine protected areas or regional averages, providing a starting point for comparing patterns across space and time. These reference points help in identifying trends, changes, and potential impacts on marine ecosystems.

Developing a robust definition of ecosystem condition reference requires collaboration among scientists, policymakers, and stakeholders. It should consider the specific characteristics and dynamics of each ecosystem, while also aligning with international frameworks and directives, such as the Marine Strategy Framework Directive (MSFD) that addresses environmental status and Good Environmental Status (GEnS).

1. Marine and Coastal Ecosystems in MOVE-ON

In the MOVE-ON project, our attention was directed towards the primary marine and coastal ecosystems present within EU and UK overseas territories. These ecosystems include coral reefs (Reunion Island, French Polynesia), seagrass meadows (Canary Islands), kelp forests (Falkland Islands) and mangrove forests (French Guiana).

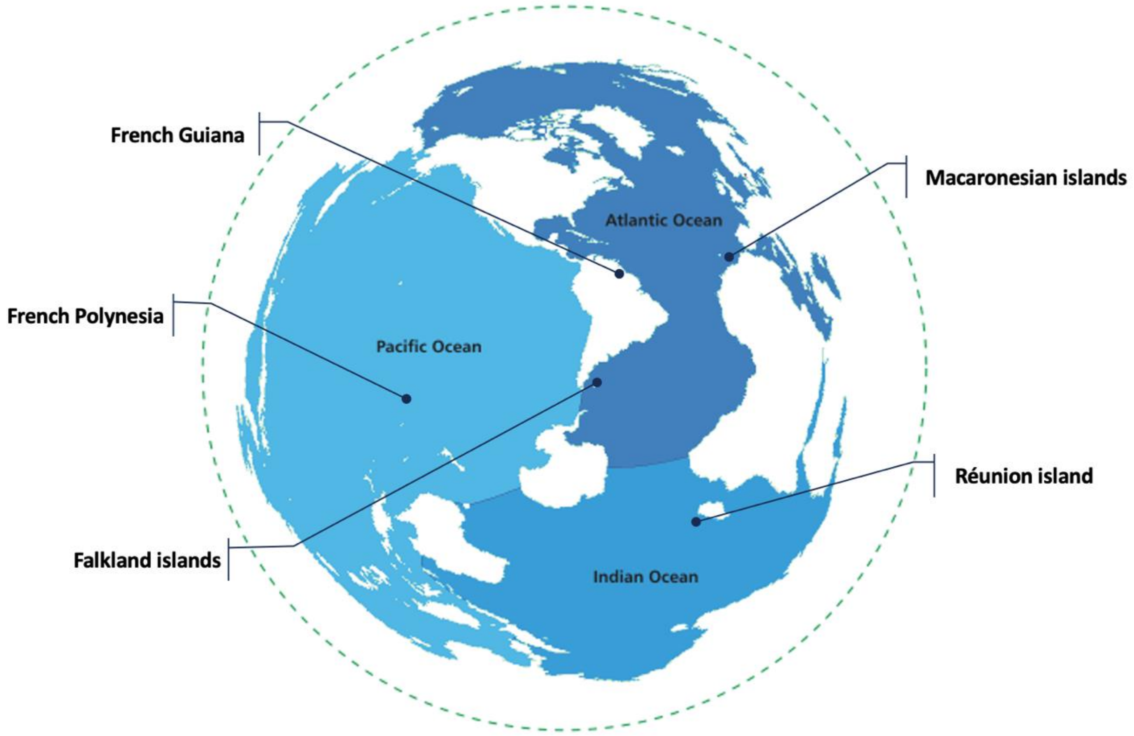


Figure 1. Location of the MOVE-ON project case study for reference condition assessment.

a. Coral Reefs

Coral reefs are vibrant ecosystems formed by colonies of hundreds to thousands of individual corals, known as polyps. These marine invertebrates possess sturdy exoskeletons composed of calcium carbonate and have a mutualistic relationship with algae called zooxanthellae. Corals construct intricate and diverse three-dimensional reef structures, including fringing reefs, barrier reefs, patch reefs, platform reefs, and atolls. Furthermore, coral reefs support a rich array of marine biodiversity, providing habitat and shelter for countless species. These ecosystems harbour intricate food webs and trophic interactions, supporting the survival and reproduction of various organisms. The loss of coral reefs would have far-reaching ecological consequences, impacting not only the corals themselves but also the diverse communities of fish, invertebrates, and other marine organisms that depend on them.

Coral reef ecosystems provide a wide range of essential ecosystem services. One of their primary services is the provision of food and livelihoods for millions of people worldwide. Coral reefs support diverse fish populations, serving as important fishing grounds that contribute to local economies and food security. Additionally, they offer valuable opportunities for tourism and recreation, attracting visitors who appreciate the beauty and biodiversity of these vibrant marine habitats.

Coastal protection is another crucial service provided by coral reefs. Their complex structures act as natural barriers, dissipating wave energy and reducing the impacts of storms and erosion on coastlines. By buffering against wave action, coral reefs help safeguard coastal communities, infrastructure, and valuable coastal ecosystems.

Coral reefs also play a significant role in carbon sequestration and climate regulation. The calcification process of corals and the growth of reef structures result in the storage of substantial amounts of carbon dioxide, mitigating the impacts of climate change. This "blue carbon" capacity of coral reefs contributes to the global carbon cycle and helps offset greenhouse gas emissions.

i. Coral Reefs in Reunion Island

Reunion Island, situated in the southwestern region of the Indian Ocean approximately 684 km east of Madagascar, is a French overseas department. Although the development of reef formations around the island is relatively limited, they play a crucial role in supporting tourism, recreational activities (Cillauren and David, 2019), commercial fishing, and coastal protection. However, the coral reefs have been significantly impacted by population growth and the consequent coastal development (Mirault, 2007). Additionally, they are now facing the increasing frequency and intensity of hurricanes, powerful swells, and heatwaves associated with climate change (e.g., bleaching events) (Quod, 1999). To safeguard the reefs, the Marine Nature Reserve of Reunion (RNMR) was established in 2007, covering an area of 35 km^2 from Saint-Paul to Etang-Salé (Thomassin *et al.*, 2010).

ii. Coral Reefs Tikehau Atoll, French Polynesia

Tikehau, located in the Tuamotu Islands, lies approximately 340 km northeast of Tahiti. It is closely situated to the neighbouring atoll, Rangiroa, only 12 km to the east. The atoll features an oval-shaped lagoon measuring 27 km in length and 19 km in width, encompassing a lagoon area of approximately 461 km^2. Consisting of two main islands and numerous islets, the entire atoll is encircled by an almost continuous coral reef. Tuheiava Pass, a single deep and wide passage located on the western shore, serves as a focal point for recreational activities, artisanal fisheries, and maritime traffic. Due to its physical characteristics, such as low altitude, small size, and exposure to distant and storm-induced swells, Tikehau atoll faces significant coastal risks, primarily flooding and erosion.

b. Seagrass beds

Seagrasses are flowering marine plants that inhabit shallow waters across various regions, ranging from the tropics to the Arctic Circle. They form extensive underwater meadows, encompassing over 300,000 km^2 and representing one of the most widespread coastal habitats on our planet. These seagrass meadows create complex and highly productive ecosystems, supporting a remarkable diversity of marine life. From small invertebrates to large fish, crabs, turtles, marine mammals, and birds, seagrasses provide crucial shelter and food for a diverse community of organisms.

The importance of seagrasses extends beyond their ecological value. They offer a multitude of services to human communities, including climate regulation, coastal protection, water purification, and a source of food. Seagrass beds play a vital role in mitigating climate change by capturing and storing significant amounts of carbon dioxide. They also act as natural buffers, reducing the impacts of coastal erosion and storm events, thus safeguarding vulnerable coastlines. Furthermore, seagrasses enhance water quality by filtering sediments and nutrients, contributing to clearer and healthier marine environments.

When assessing the ecological condition of seagrass beds, various indicators and parameters are commonly utilized. These include shoot density, benthic cover, leaf length, biomass/cover of epiphytes, as well as the abundance and/or biomass of macroinvertebrates and fish assemblages.

Unfortunately, seagrass meadows face threats on a global scale, stemming from natural and anthropogenic stressors. Nearly 30% of the global seagrass area has been lost since the late nineteenth century, with 22 out of the world's 72 seagrass species experiencing a decline. Pressures such as urban, industrial, and agricultural runoff, coastal development, dredging, unregulated fishing and boating activities, and the impacts of climate change contribute to the degradation of seagrass habitats.

The decline in seagrass cover has significant implications for human well-being due to their essential ecosystem services. However, there is hope for reversing this trend through seagrass conservation, rehabilitation, and restoration efforts. By actively addressing the threats and implementing appropriate management strategies, we can restore lost seagrass habitats and the valuable ecosystem services they deliver.

i. Seagrass beds in the Canary Islands

The Macaronesian regions of the Canary Islands and Madeira are home to various seagrass genera, including Cymodocea, Halophila, Ruppia, Thalassia, and Zostera. These seagrass meadows, found along the sheltered eastern and southern coasts, are vital in providing several ecological services, such as primary production, habitats, nurseries, and coastal protection. The level of primary productivity can vary, influenced by factors like meadow density, geographic area, and hydrological conditions. These ecosystems serve as essential habitats for numerous marine organisms, which rely on them at different life cycle stages for food and shelter from predators.

Coastal development, including port expansions, and waste disposal, especially sewage discharges, pose significant pressures on this habitat. The exact causes of the epiphytic growth of *Lyngbya sp* over *C. nodosa* communities are still not fully understood. The blooms of this cyanobacteria are temporary and likely have a combination of natural and human-induced origins. Implementing regulations for activities such as coastal development, dredging, and waste disposal, as well as establishing zoning for aquaculture facilities away from seagrass habitats, are effective management measures. In certain cases, these measures can be implemented within protected areas to safeguard seagrass beds.

c. Kelp forests

Kelp forests, consisting of large brown algae, thrive in cool and relatively shallow coastal waters. These underwater forests provide a vital habitat and nourishment for a wide range of fish, invertebrates, and marine mammals. Monitoring kelp forests' health of, particularly the canopy's understorey, can be challenging. However, indicators such as kelp density and size, as well as the abundance of mobile macro-invertebrates (such as urchins and crustaceans) and sessile macro-invertebrates and macroalgae, are commonly used.

Kelp forests face increasing threats globally due to various human activities. Climate change, resulting in rising ocean temperatures, poses a significant risk to these cold-water ecosystems. The elevated temperatures associated with climate change increase the likelihood of massive kelp forest die-offs, raising concerns about their ability to recover. Destructive fishing practices, such as bottom trawling, have also been implicated in dramatic declines in kelp populations, as seen in the case of the United Kingdom. Additionally, removing predators through fishing and hunting has likely altered the structure of many kelp forests.

Beyond their ecological importance, kelp forests provide valuable ecosystem services. These services include carbon sequestration, as kelps absorb carbon dioxide from the atmosphere and store it in their tissues, contributing to mitigating climate change. Kelp forests also offer coastal protection, acting as natural buffers against waves and reducing erosion of shorelines. Furthermore, they support local economies through activities such as tourism, recreational fishing, and commercial harvesting of kelp for various purposes, such as food and cosmetics.

i. Falkland Island kelp forest

The Falkland Islands, located in the South West Atlantic between the southern tip of South America and the Sub-Antarctic region, boast a collection of over 700 small islands surrounding the main islands of East and West Falkland. Along the coastline, one can find pristine kelp forest ecosystems that remain intact, showcasing their ecological significance (van Tussenbroek, 1993; Friedlander et al., 2020; Mora-Soto et al., 2021). These kelp forests exhibit higher biodiversity and species richness compared to the neighbouring regions of South America and the Sub-Antarctic island of South Georgia (Figuerola et al., 2017; Beaton et al., 2020). Notably, these kelp forests provide habitat for commercially valuable species and play a vital role in carbon cycling within the surrounding waters.

d. Mangroves

Mangroves are unique trees that thrive in tropical tidal waters, where their growth is enabled by their ability to tolerate high salinity and regular submergence, unlike most other tree species. They can be found in tropical regions worldwide, extending to subtropical areas up to the 20°C winter sea water isotherm (Duke et al., 1998). Mangroves flourish along calm shorelines such as deltas, estuaries, lagoons, and open coasts, as they are unsuitable for surf shores (Worthington et al., 2020). However, their intertidal distribution makes them particularly vulnerable to sea-level rise and coastal changes.

Monitoring mangrove forests can be challenging, but various indicators are commonly used. These include tree height, diameter at breast height (DBH), abundance, sapling heights, density, canopy cover, as well as the density and species richness of macro-invertebrates. Remote sensing techniques often utilize the Normalized Difference Vegetation Index (NDVI) to assess the vitality of mangrove forests.

Mangroves provide numerous ecosystem services that are invaluable to both the environment and human populations. One of their primary services is coastal protection, as mangrove forests act as natural buffers against storm surges, waves, and erosion, safeguarding coastlines and nearby communities. They also play a crucial role in carbon sequestration, helping to mitigate climate change by capturing and storing significant amounts of carbon in their biomass and sediments. Furthermore, mangroves contribute to water purification by filtering pollutants and trapping sediments, improving water quality in coastal areas.

Mangroves serve as important habitats for a diverse range of species, including fish, birds, and invertebrates. They provide nesting sites, feeding grounds, and nurseries for various marine organisms, supporting the biodiversity and productivity of coastal ecosystems. Additionally, mangrove forests are often utilized by local communities for traditional livelihood activities such as fishing and gathering of forest products, including timber, fuelwood, and medicinal plants.

However, mangrove forests face multiple threats. Sea-level rise, changes in precipitation patterns, and global warming can increase salinity levels to a point unsuitable for mangrove seedlings. The intensification and frequency of extreme weather events, such as storms and hurricanes, pose a significant risk to mangrove forests and hinder their recovery between consecutive events. Coastal development, associated pollution, alterations in the hydrological cycle, and changes in sedimentation patterns exert chronic stress on mangroves, limiting their adaptive capacity to move inland or regenerate after disturbances.

i. Mangrove forests - French Guiana

Guiana French Guiana is host to diverse mangrove tree species, including the red mangrove (*Rhizophora mangle*), black mangrove (*Avicennia germinans*), and white mangrove (*Laguncularia racemosa*). These mangroves have successfully adapted to thrive in the challenging estuarine zone, characterized by fluctuating salinity levels, tidal movements, and nutrient-rich sediments. The extensive coverage of mangroves along the coast of French Guiana is remarkable. Overall, the ecological condition of these mangroves is highly favourable. However, they face common threats such as deforestation, pollution, and the impacts of climate change. These mangroves play a vital role in supporting the region's biodiversity by serving as nursery grounds and shelters for a wide array of marine species, including fish, crustaceans, and molluscs. The intricate root systems of mangroves provide essential habitat and protection for juvenile organisms, enabling their growth and development before they venture into the open ocean. Beyond their ecological significance, the mangroves act as natural buffers, safeguarding the coastline against erosion and storm surges. The dense network of roots effectively dissipates wave energy and stabilizes sediments, thereby reducing the impact of coastal erosion. This protective function holds particular importance in French Guiana, where coastal areas are vulnerable to storms and rising sea levels. Moreover, the estuarine mangroves of French Guiana deliver various socio-economic benefits to local communities. They support artisanal fishing activities, serving as a vital source of livelihood and sustenance. Additionally, the mangroves play a pivotal role in attracting tourism, drawing visitors who appreciate the unique beauty and biodiversity offered by these coastal ecosystems.

2. Objectives (assessment of ecological condition of reference state) & ES. Assessment of Ecological Reference Condition and Ecosystem Services in Tropical Marine Coastal Ecosystems of EU Overseas Territories

Marine coastal ecosystems found in EU overseas territories, including coral reefs, seagrass beds, mangroves, and kelp forests play a vital role in providing essential ecosystem services. Understanding the assessment of ecological reference conditions and the link between ecosystem condition and ecosystem services is crucial for effective management and conservation. Coral reefs support a wide range of ecosystem services, including fisheries production, tourism, and shoreline protection. Hughes et al. (2018) emphasize the importance of assessing ecological reference conditions in coral reef ecosystems and their relationship to ecosystem services. Seagrass beds are productive ecosystems that provide important ecosystem services such as carbon sequestration, nutrient cycling, and habitat provision. (Duarte et al. 2020) discuss the assessment of ecological reference conditions in seagrass ecosystems and their implications for ecosystem services. Mangrove forests serve as vital coastal buffers, supporting biodiversity, carbon storage, and providing nursery habitats for numerous marine species. Alongi (2015) provides insights into the assessment of ecological reference conditions in mangrove ecosystems and their relationship to ecosystem services. Kelp forests contribute to carbon sequestration, provide habitat for various species and support fisheries. Krumhansl et al. (2016) examine the assessment of ecological reference conditions in kelp forest ecosystems and their association with ecosystem services.

Selection of reference condition assessment methods and sites

Following the guidelines for the assessment of ecosystem condition developed in "MOVE-ON D.2.1.a - *Report for marine ecosystem condition and ecological reference status assessment methods in ORs and OCTs - Technical guidelines for assessment*", we identified the context of ecosystem condition assessment for each region in the MOVE-ON project and applied a BEF framework on each case study.

1. REUNION ISLAND - CORAL REEF

a. Decision tree:

Following the decision tree elaborated in MOVE-ON D.2.1.a, we analysed the context of Reunion Island and selected the most appropriate method for defining reference conditions:

Historical data available

Quantitative data + Complete data sample

Historical data used to describe reference condition and assess natural variability

Historical reference condition

b. Habitat mapping

Detailed habitat data are available for the Reunion Island to produce maps at several resolution and information details, compatible with ecosystem condition assessment (Fig. 2, Fig. 3). Coral reefs extent reaches 18.6 km². The Marine Reserve protects 80% of the coral reefs since 2007, and 6% are fully protected. The total area of the RMNR is 35 km².

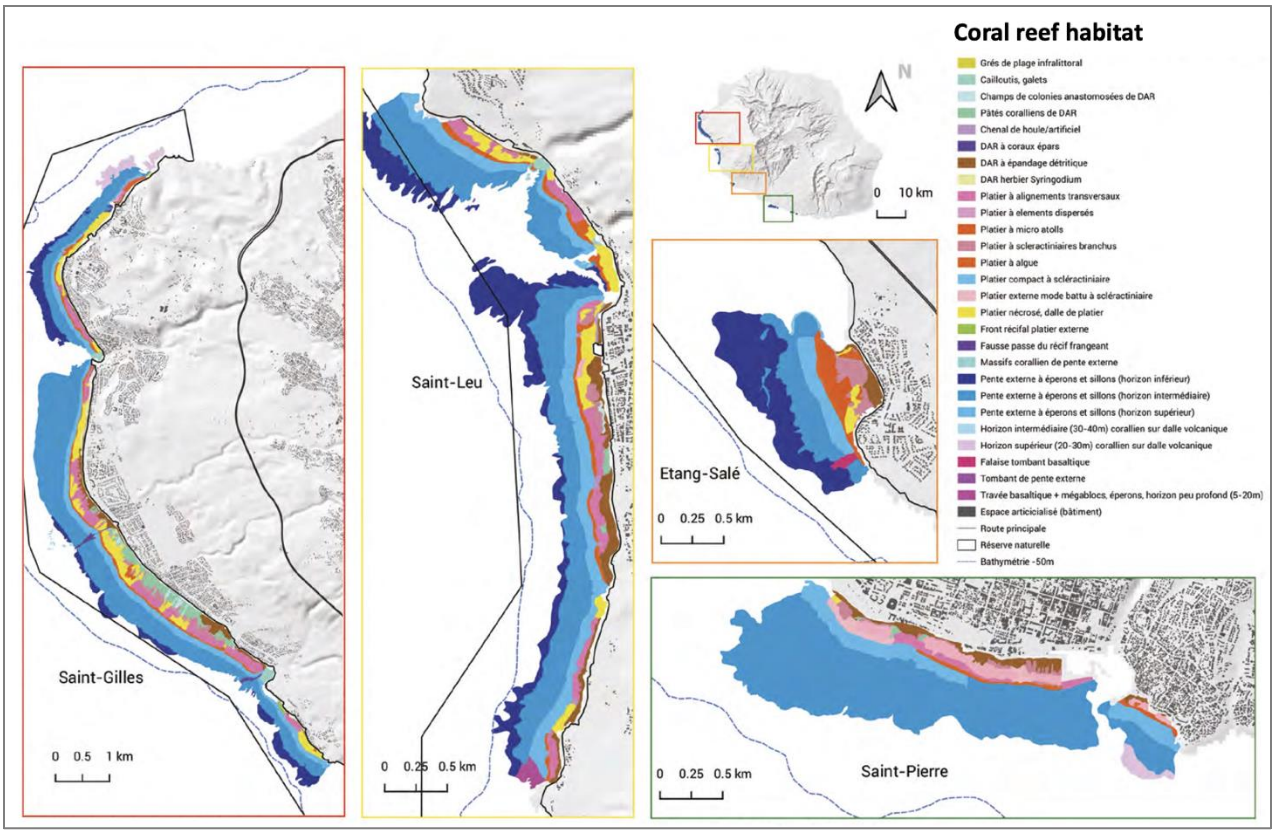


Figure 2. Map of coral reef extent in the Reunion Island (Sources: Marex, BdThopoIGN, Ifrecor).

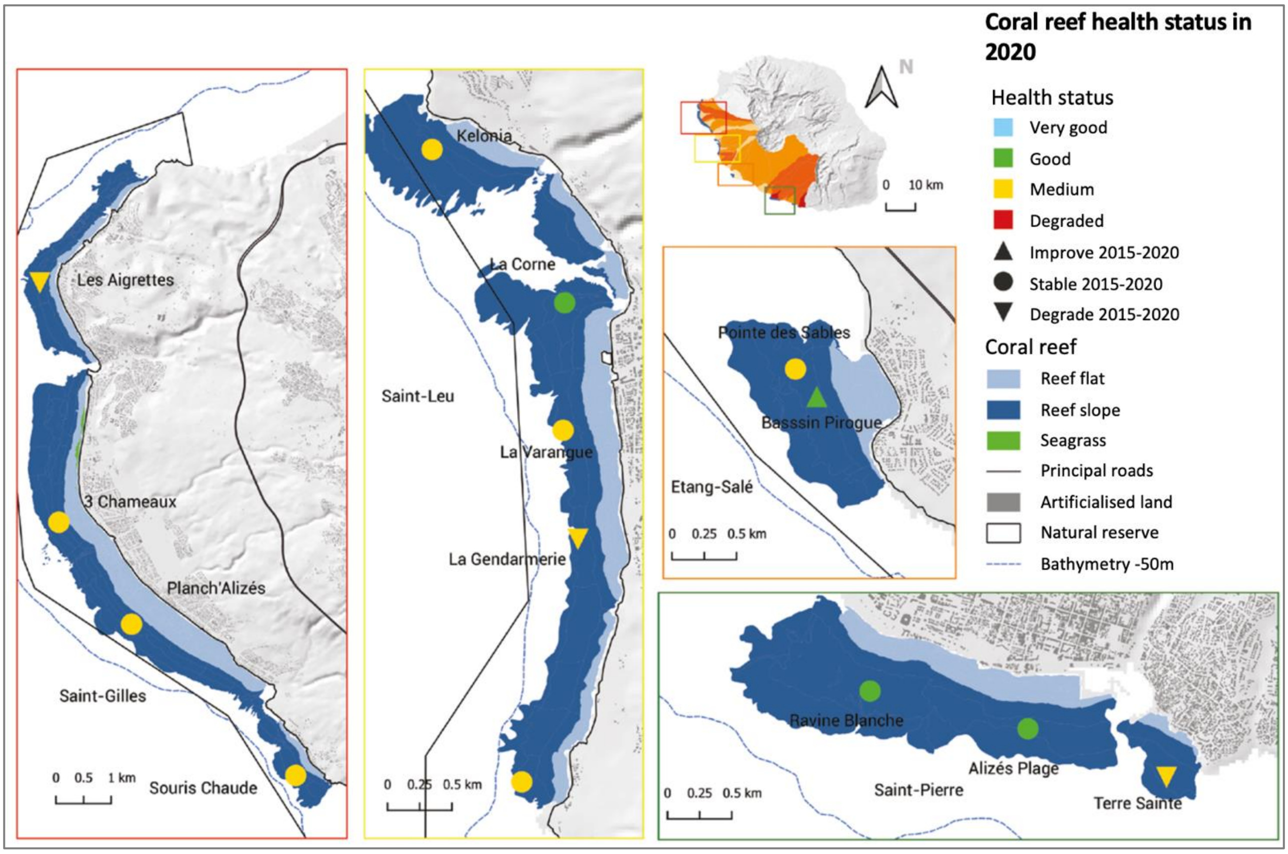


Figure 3. Map of monitoring site on coral reef in the Reunion Island and health status in 2020 (Sources: Marex, BdThopoIGN, Ifrecor).

c. Functional biodiversity

The marine fish species inventory for Reunion Island reefs reached a total number between 850 and 900 species of bony fish (Bourmaud, 2005; Letourneur et al. 2004), 66% of them (~580) associated with coral reefs (Letourneur et al. 2004). Additionally, 459 species of cnidarians, including 148 species of reef-building corals (hard corals), 2 500 Molluscs, 61 Echinoderms, and 181 plants were described on Reunion Island (Bourmaud, 2005). The list of fish species is available from Letourneur et al. (2004) and could represent the theoretical referent functional space of reef fish. However, this list represents a compilation of different surveys, with different approaches, to a maximum depth of 4220 m. Coral reefs are not present at this depth but in all likelihood, mesophotic reefs were sampled. To avoid any biases when comparing this inventory to modern ones, we used instead the list of fish species observed by Chabanet (1994) on the reef from Saint Gilles to La Saline. Sampling went from January 1991 to March 1992, sites are clearly identified and therefore, comparable with current fish assemblages. A total of 257 fish species were recorded for qualitative studies (presence/absence) performed during the day and at night. However, fewer species (103) were targeted for quantitative studies (including a measure of abundance). For practical reasons, and comparability with future surveys, the functional biodiversity approach will be based on this list of species. Note that the current monitoring programme targets only 50 species of fish.

All species were classified according to six different life-history traits using functional properties defined by Mouillot et al. (2014): (1) Species maximum body size: <7 cm, 8–15 cm, 16–30 cm, 31–50 cm, 51–80 cm or >80 cm; (2) Mobility: sedentary (including territorial species), mobile or very mobile; (3) Period of activity: diurnal, nocturnal, or both; (4) Schooling: solitary, pairing, small groups (3–20 individuals), medium groups (20–50 individuals) or large groups (>50 individuals); (5) Position in the water column: benthic (species associated with the bottom), bentho-pelagic, or pelagic and (6) Trophic group: herbivores-detritivores (feed upon turf and filamentous algae and/or detritus), macroalgae-feeders (large fleshy algae and/or seagrass), sessile invertebrate feeders (e.g., corals, sponges, ascidians), mobile invertebrate feeders (benthic prey, such as crabs and mobile mollusks), planktivorous (small organism in the water column), piscivores (fish and cephalopods) or omnivores (both vegetal and animal material). Traits values were retrieved from Quimbayo et al. (2021) database (for 20 species) and completed using FishBase.

The space with the best quality has the lowest Mean Absolute Deviation (MAD) value, meaning the best space has 6 dimensions – mad = 0.042). From this, we computed the functional space – for graphical reasons, we can only plot in 4-dimensional space (Fig. 4).

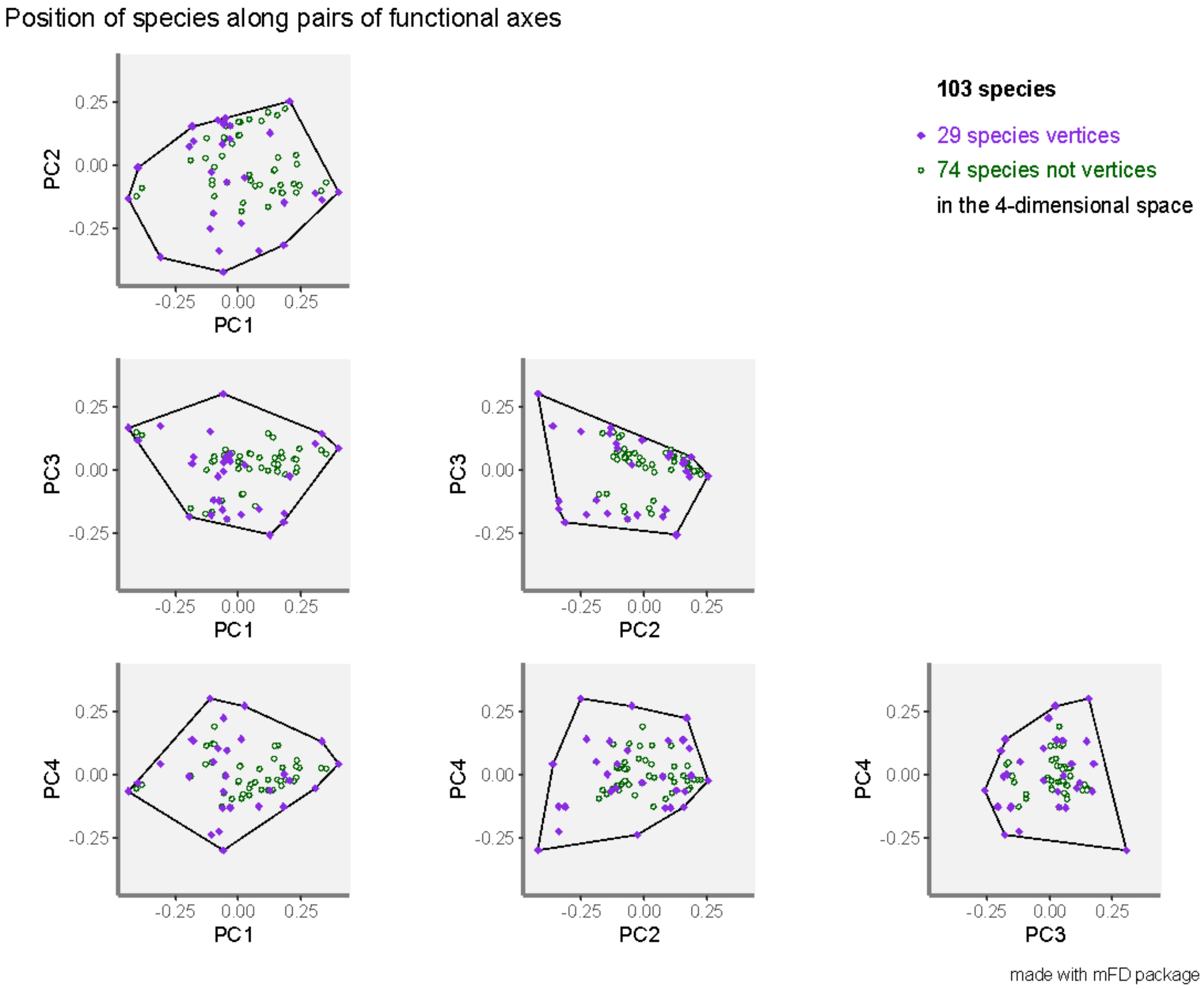


Figure 4. Functional space for fish species in coral reefs of the Reunion Island.

Here, we can assume that the reference functional space for Reunion Island includes 103 species of fish with their associated traits, projected in a multidimensional space. Or else, we can assume that for each site monitored between 1991 and 1992, the referent condition of each site corresponds to the multidimensional space occupied by the fish assemblages at that time.

Using the multidimensional space occupied by fish species, we can identify spatial changes in community composition using beta-diversity, whether changes in community are characterised by species replacement (turnover) or species loss (nestedness). Beta-diversity is an interesting descriptor of diversity for assessing the effects of habitat and landscape change because it measures the difference in species composition between communities (Anderson et al. 2011), expressed as dissimilarity. Taking the example of two different sites, Trois Chameaux_reef slope (TC_Pente) and Planch' Alyzée_reef slope (PA_Pente), the functional dissimilarity between the two sites reach 0.2686 (or 27%), from which almost the entire difference (26.6%) comes from the loss (or absence) of fish in Trois Chameaux_reef slope compared to Planch' Alysée (Fig. 5.). This approach can be used to compare different site, or the same site on a time series, as long as the protocol remains the same (target species).

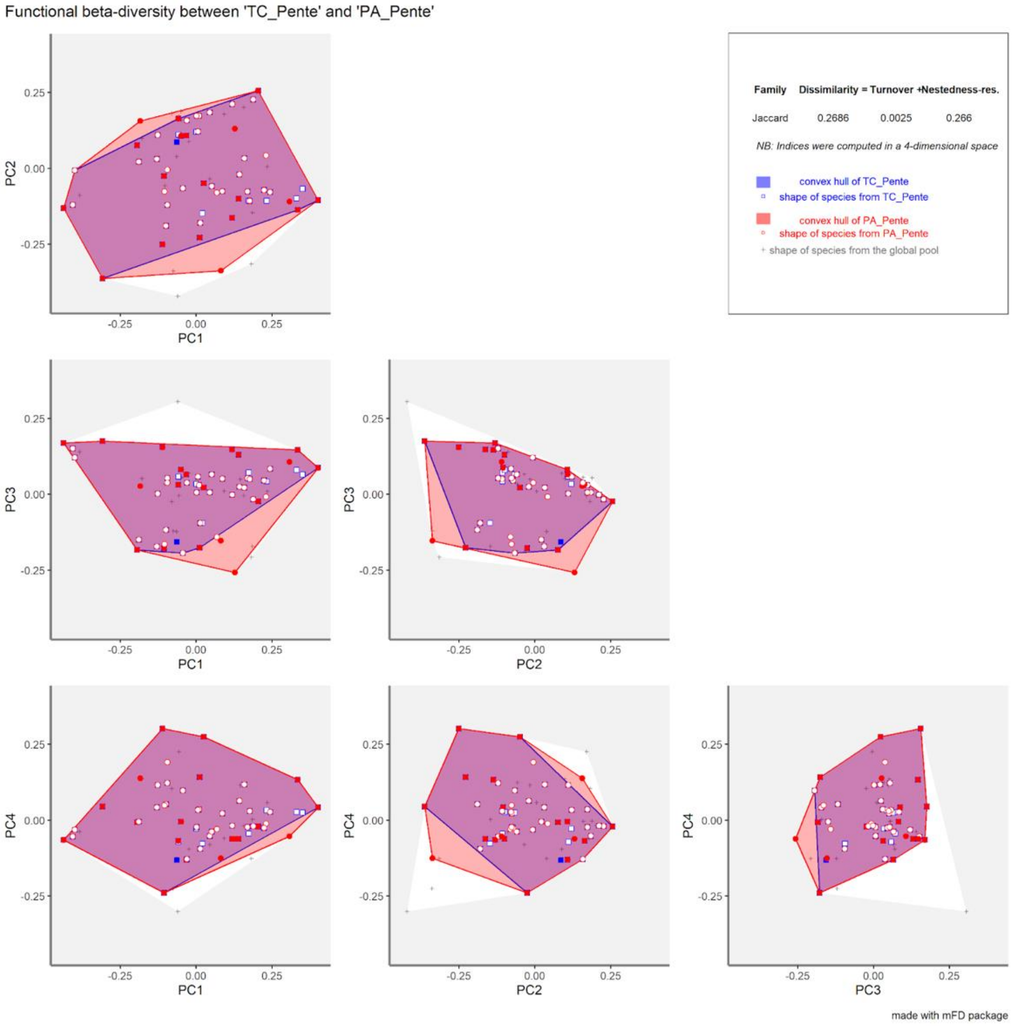


Figure 5. Functional beta-diversity between Trois Chameaux Reef slope and Planch' Alysée reef slope fish assemblages (Data: year 1991).

From this comparative overview between the two sites, the mFD package allows computing Alpha Functional Diversity indices (Alpha diversity - see Annexe 1), each providing a different facet of fish community. The functional richness represents the proportion of functional space filled by species of the studied assemblage, i.e. the volume inside the convex-hull shaping species.

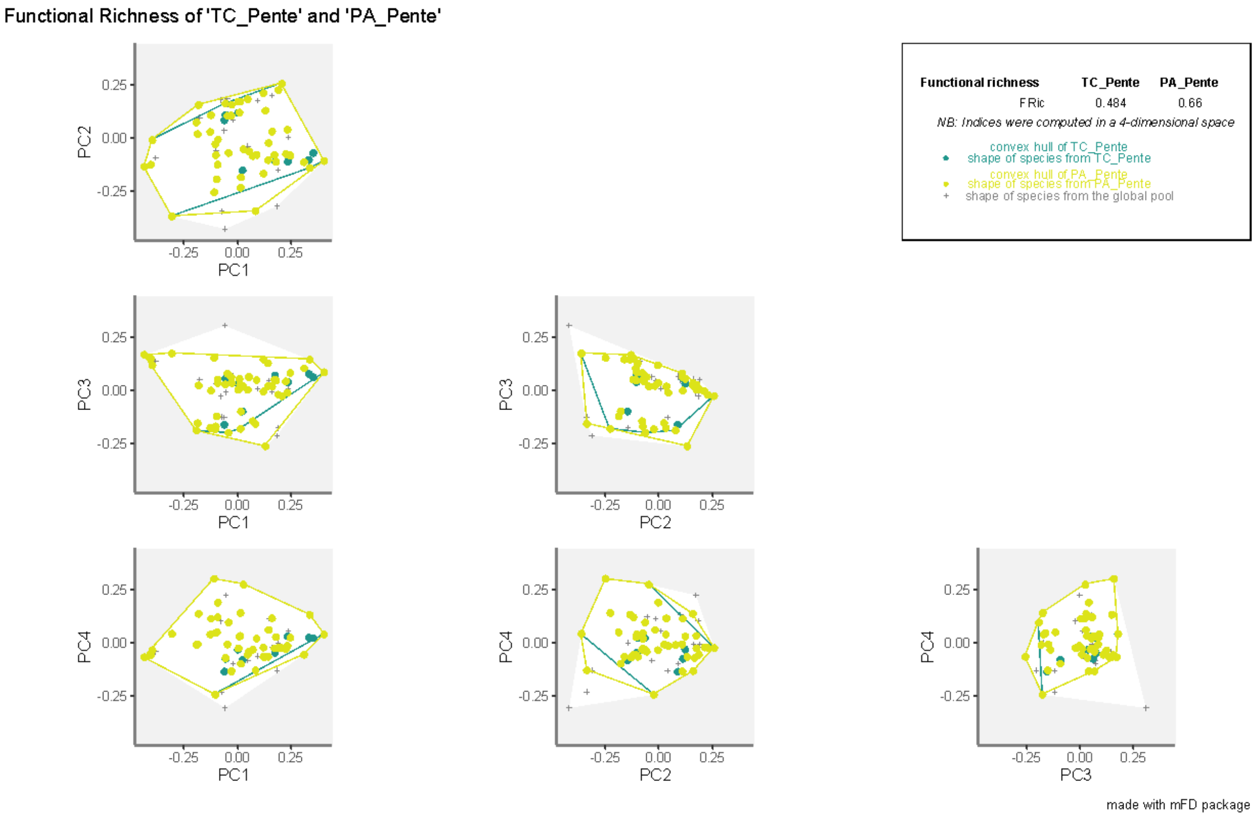


Figure 6. Functional richness of fish assemblages in Reunion Islands in 1991, between Trois Chameaux reef slope (TC_Pente) and Planch' Alizés reef slope (PA_Pente).

Fig. 6 shows the difference between the two assemblages. At this stage, we cannot say if Trois Chameaux (TC) was more degraded than Planch' Alyzés(PA). Despite the fact that TC had 2 more species (S=59) than PA (S=57), the functional richness of TC (0.484) is less than PA (0.660). It could be explained by PA having more species fulfilling the same function. Having high functional redundancy is also a good indicator of the resilience of the site to perturbation, meaning that if one species disappears, the function remains. A deeper analysis to look precisely at the species or function dissimilarity would be required but falls out of the scope of this report. However, such analyses highlight the importance of looking beyond the sole species richness of a site, which could be misleading.

We can either keep the referent functional biodiversity indices for each sites / reef formation, or average the indices per reef formation, so that the referent FRic for reef flat would be 0.439 ± 0.122 (mean ± standard error), 0.496 ± 0.090 for reef slope, and 0.435 ± 0.007 for reef pass (Table 1). Any significant difference from those values would characterise a significant change in fish communities due to perturbation (positive or negative).

Table 1. Functional indices for fish assemblages in Reunion Island.

Sites	**sp_richn**	**FDis**	**FEve**	**FRic**	**FDiv**	**FOri**	**FSpe**
TC_Platier	53	0,453	0,449	0,682	0,795	0,096	0,353
TC_Pente	59	0,549	0,434	0,484	0,892	0,079	0,452
CM_Platier	27	0,492	0,529	0,306	0,864	0,104	0,438
CM_Pente	46	0,558	0,478	0,345	0,884	0,071	0,447
PA_Platier	35	0,398	0,443	0,329	0,730	0,102	0,340
PA_Pente	57	0,519	0,456	0,660	0,863	0,090	0,449
SG_Passe	45	0,434	0,485	0,431	0,935	0,064	0,479
PH_Passe	37	0,561	0,530	0,425	0,870	0,088	0,435
TB_Passe	41	0,581	0,530	0,451	0,889	0,133	0,467

d. Community ecology

Drawing upon a comprehensive review of scientific literature and research findings, we examined various aspects of coral reef ecosystems, including coral diversity and composition, coral health and resilience and algal dynamics.

Coral diversity and composition: the coral reefs of Reunion Island exhibit a remarkable diversity of coral species (Table 2). Studies have identified a wide range of genera, including Acropora, Montipora, Pocillopora, and Porites and Astreopora among others. These coral communities contribute to the structural complexity of the reefs and provide crucial habitats for a diverse array of marine organisms. A shift in population structure has been globally observed from Acropora dominant communities to a mix of Pollicipora, Porites and Astreopora genera (Bigot 2008, Chabanet 1994) (Fig. 7).

Table 2. Species richness in the reef flat and external slope of coral reefs in the Reunion Island in 1994 (from Chabanet 1994).

Reef region	Component	Species matrix
Reef flat & external slope	Benthos Fish	86 coral species 98 fish species
Reef flat	Benthos Fish	29 coral species 53 fish species
External slope	Benthos Fish	73 coral species 77 fish species

Coral health and resilience: assessments of coral health and resilience in Reunion Island have revealed both positive and negative trends. While some areas exhibit healthy coral cover and vitality, others have experienced significant stress and degradation.

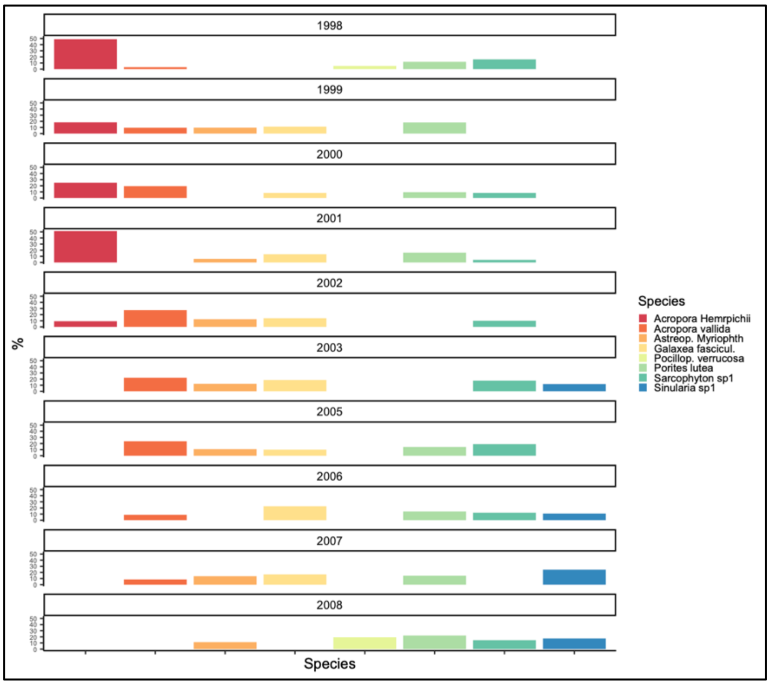


Figure 7. Relative contribution (in %) of the most abundant species at the "Planch Alizé PE" station between 1998 and 2008 and shift in major coral assemblages (Data source Bigot 2008).

Coral communities and algal dynamics: macroalgae, turf algae, and corals contribute to the overall dynamics and structure of the reefs. The global trend is a shift between coral dominant to algal dominant communities as shown on figure 8 on temporal change between 1998 and 2019 on several reefs at Reunion Island.

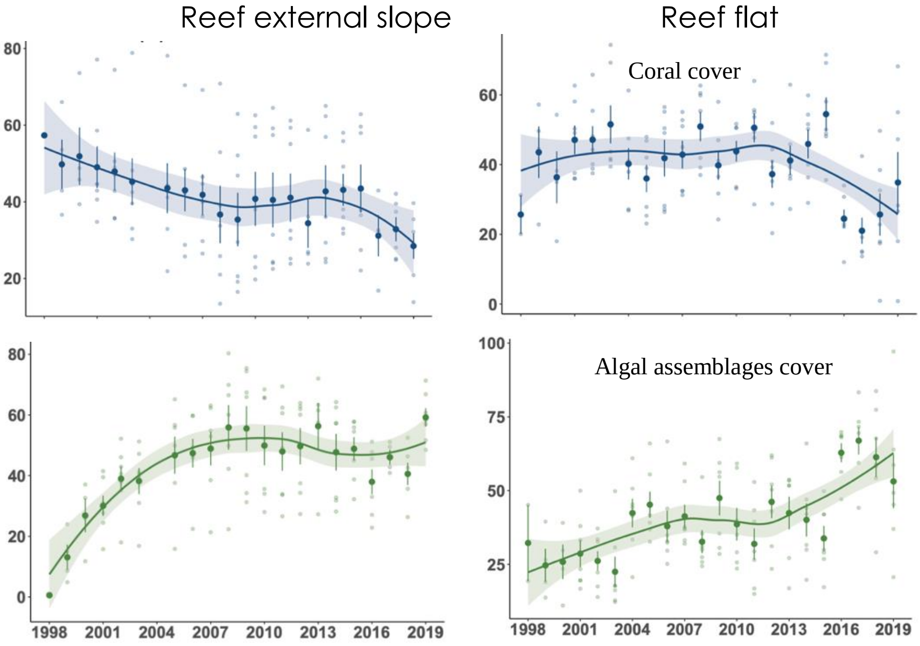


Figure 8. Temporal changes in coral cover and algal assemblages cover in reefs of the Reunion Island between 1998 and 2019. GCRMN dataset. Evolution of living coral (%) and algae population

(macroalgae and turf %). Mean value/year/station (•). Mean value/year, all stations combined and standard error (• with |). Evolution curves and uncertainty bands: LOESS type smoothing.

e. Summary table

	HABITAT			
Variable	**Spatial distribution** **1.4 Habitat distribution: 1.4.1**	**Fragmentation** **1.4 Habitat distribution: 1.4.2**	**Habitat loss/increase** **1.5 Habitat extent: 1.5.1 – 1.5.2**	**Connectivity** **1.4 Habitat distribution: 1.4.2**
Index	**Surface extent**	**Patch size**	**Loss/Increase**	**Connectivity**
Values	18.6 km^2	Small and Large	na	Patch connectivity/isolation

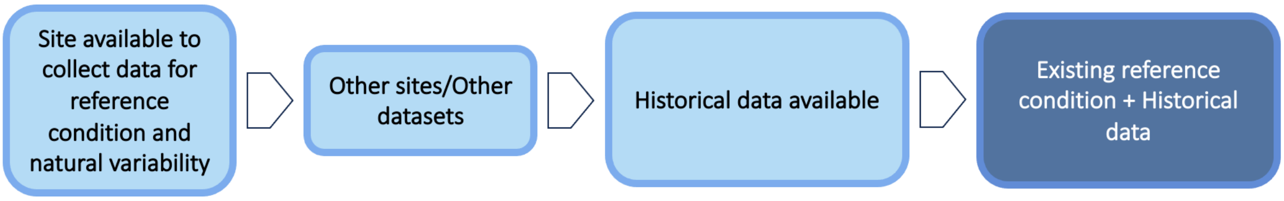
	BIODIVERSITY				
Variable	**Functional space** **6.2 Condition of benthic community: 6.2.1 – 6.2.2**	**Functional diversity** **6.2 Condition of benthic community: 6.2.2** **4.3 Abundance/distribution of key trophic groups/species: 4.3.1** **3.3 Population age/size distribution: 3.3.1 – 3.3.4**			
Index	**Functional space**	**FRic**	**FEve**	**FDiv**	**FOri**
Values	103 species of fish and associated traits. % change of functional space Beta diversity between referent site from 1991 and other years.	Reef flat: 0.439 ± 0.122	Reef flat: 0.474 ± 0.028	Reef flat: 0.796 ± 0.039	Reef flat: 0.100 ± 0.002
		Reef slope: 0.496 ± 0.090	Reef slope: 0.456 ± 0.013	Reef slope: 0.880 ± 0.009	Reef slope: 0.080 ± 0.005
		Pass: 0.435 ± 0.007	Pass: 0.515 ± 0.015	Pass: 0.898 ± 0.019	Pass: 0.095 ± 0.020

	COMMUNITY
Variable	**Community assemblages** **1.1 Species distribution: 1.1.3 ; 1.2 Population size: 1.2.1 ; 2.1 Invasive species: 2.1.1**
Index	**Community assemblages**
Values	MVCC not computable
	Corals Reef slope: 54,1% 1998 → 28 ± 8% 2019 coral cover change in community structure Acropora → Pollicipora, Porites, Astreopora Corals Reef flat: 43,5 % 1999 → 34,23% 2019 Algae Reef slope: 7,6% 1998 → 59 ± 8% 2019 Algae Reef flat: 24,7 % 1999 → 53,2% 2019 Fish: 30 indicator species → 7.8kg/100m^2 1998 → 2019 low biomass (1.5 kg/100m^2, decrease in size and abundance).

2. TIKEHAU FRENCH POLYNESIA - CORAL REEF

a. Decision tree:

Following the decision tree elaborated in MOVE-ON D.2.1.a, we analysed the context of the Tikehau atoll and selected the most appropriate method for defining reference conditions:



b. Habitat mapping:

Detailed habitat data are available for the Tikehau atoll to produce maps at several resolution and information details, compatible with ecosystem condition assessment (Fig. 9, Fig. 10). Benthic classes and geomorphic structure distribution is given in Fig. 11.

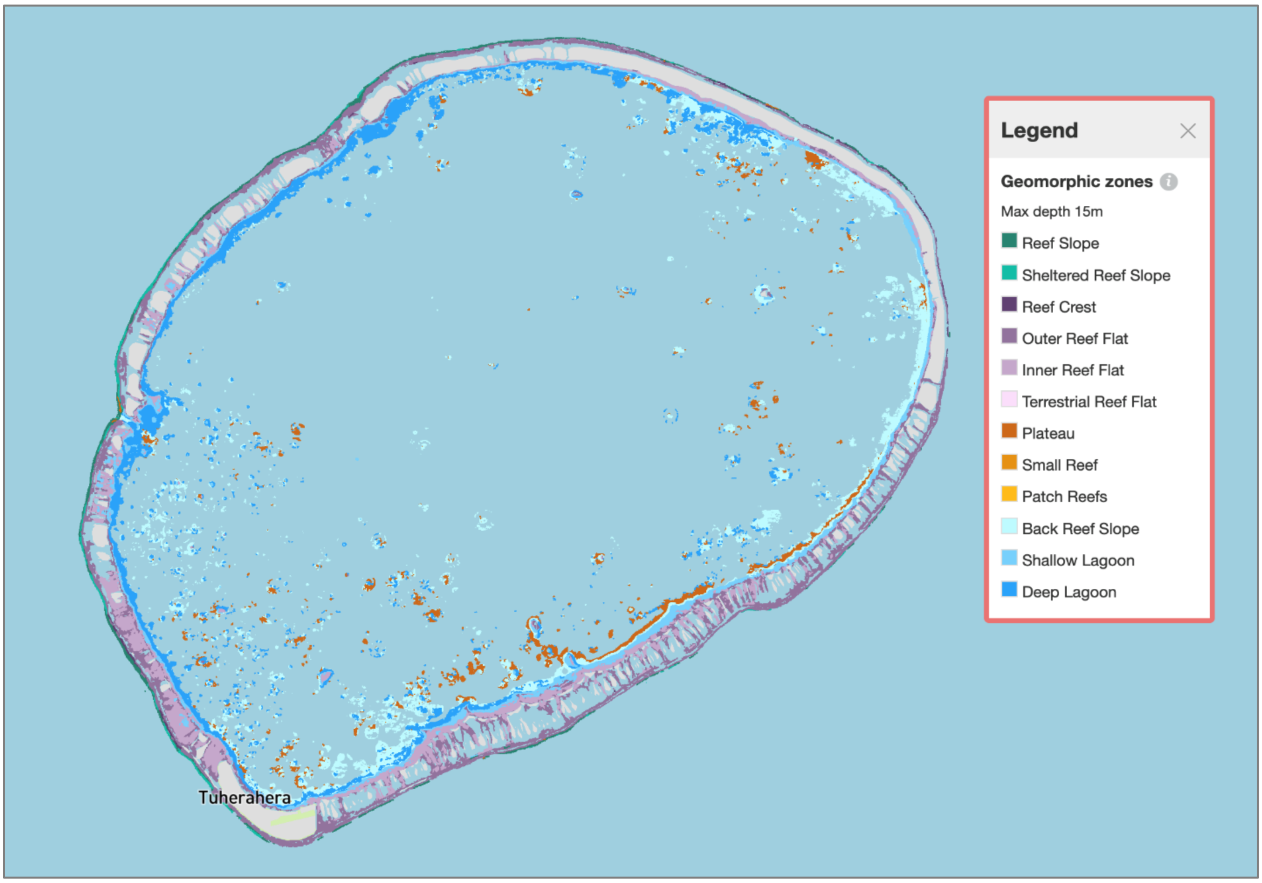


Figure 9. Reef geomorphology. Tikehau atoll in French Polynesia (Source: Allen Coral Atlas).

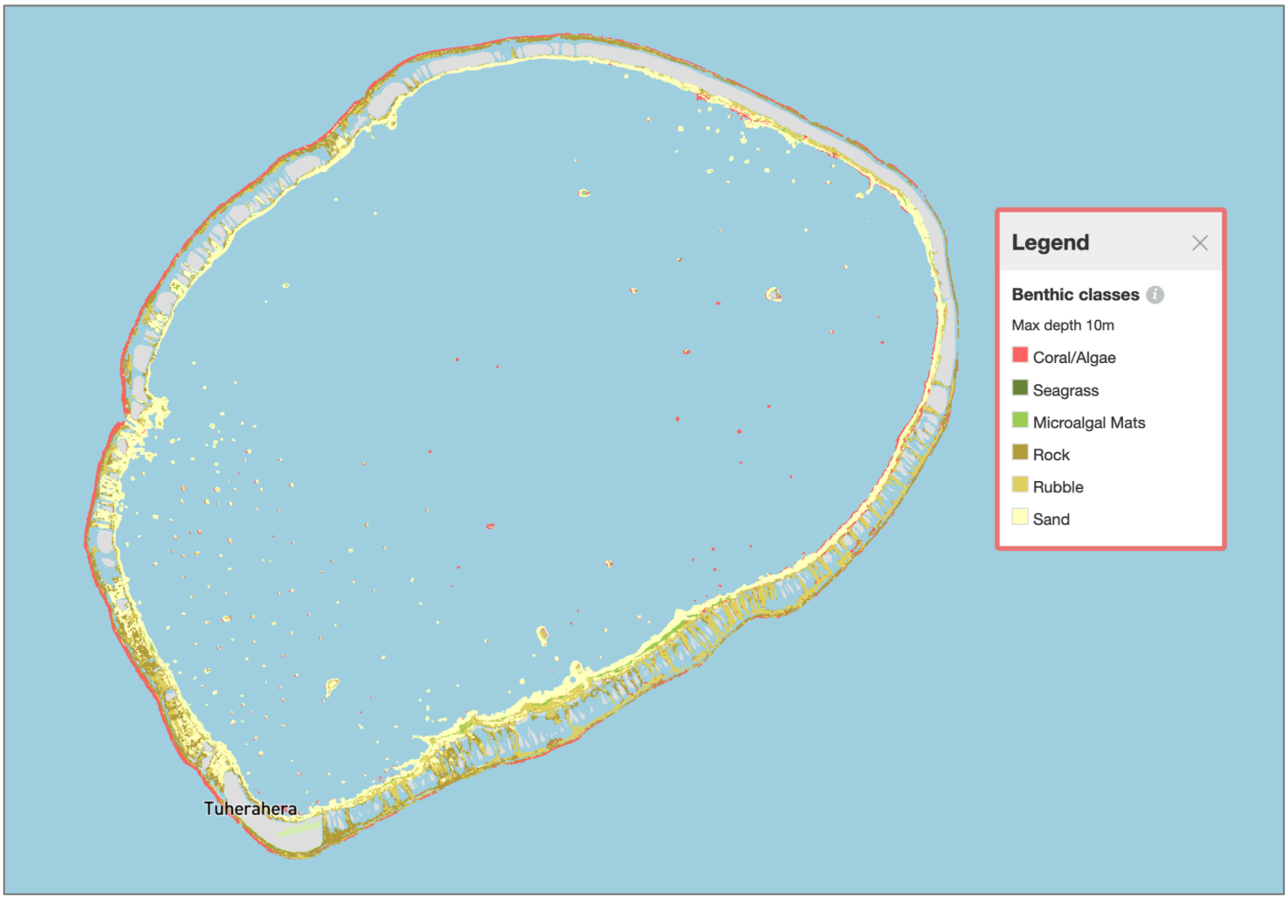
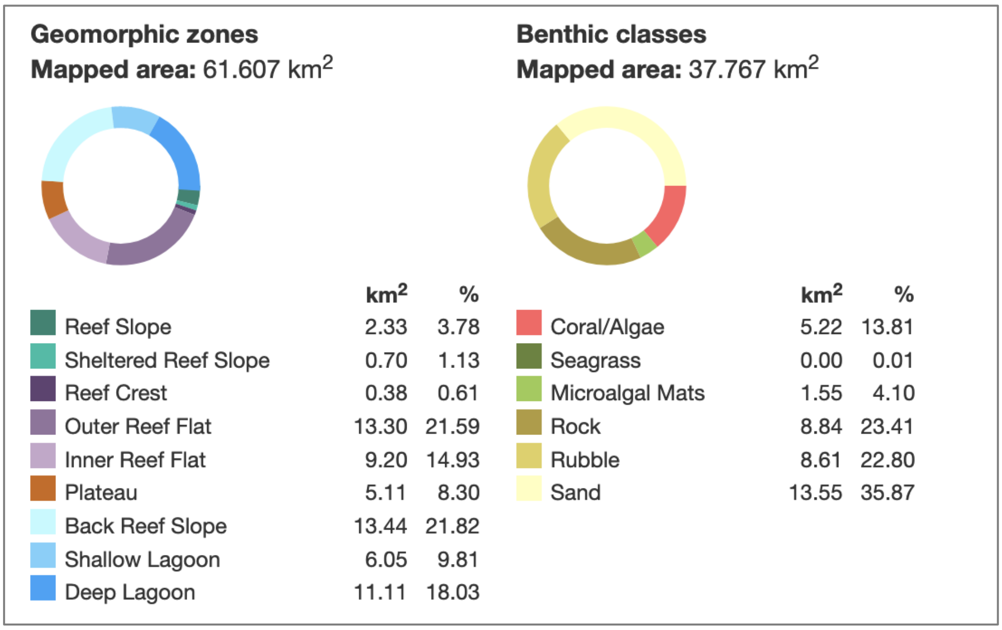


Figure 10. Reef habitat. Tikehau atoll in French Polynesia (Source: Allen Coral Atlas).



Geomorphic zones
Mapped area: 61.607 km^2

	km^2	%
■ Reef Slope	2.33	3.78
■ Sheltered Reef Slope	0.70	1.13
■ Reef Crest	0.38	0.61
■ Outer Reef Flat	13.30	21.59
■ Inner Reef Flat	9.20	14.93
■ Plateau	5.11	8.30
■ Back Reef Slope	13.44	21.82
■ Shallow Lagoon	6.05	9.81
■ Deep Lagoon	11.11	18.03

Benthic classes
Mapped area: 37.767 km^2

	km^2	%
■ Coral/Algae	5.22	13.81
■ Seagrass	0.00	0.01
■ Microalgal Mats	1.55	4.10
■ Rock	8.84	23.41
■ Rubble	8.61	22.80
■ Sand	13.55	35.87

Figure 11. Surface area of the geomorphic and benthic classes around the Tikehau atoll in French Polynesia. (Source: Allen Coral Atlas)

c. Functional biodiversity

Underwater Visual Census were performed from 2004 to 2022 on the reef monitoring station in Tikehau, recording species abundance. Fish abundances recorded on the three transects were summed to have an overall abundance on the site. A total of 186 species have been recorded.

All species were classified according to six different life-history traits using functional properties defined by Mouillot et al. (2014): (1) Species maximum body size: <7 cm, 8–15 cm, 16–30 cm, 31–50 cm, 51–80 cm or >80 cm; (2) Mobility: sedentary (including territorial species), mobile or very mobile; (3) Period of activity: diurnal, nocturnal, or both; (4) Schooling: solitary, pairing, small groups (3–20 individuals), medium groups (20–50 individuals) or large groups (>50 individuals); (5) Position in the water column: benthic (species associated with the bottom), bentho-pelagic, or pelagic and (6) Trophic group: herbivores-detritivores (feed upon turf and filamentous algae and/or detritus), macroalgae-feeders (large fleshy algae and/or seagrass), sessile invertebrate feeders (e.g., corals, sponges, ascidians), mobile invertebrate feeders (benthic prey, such as crabs and mobile mollusks), planktivorous (small organism in the water column), piscivores (fish and cephalopods) or omnivores (both vegetal and animal material). Traits values were retrieved from Quimbayo et al. (2021) database (for 41 species) and completed using FishBase.

The functional space with the best quality has 5 dimensions (mad = 0.050, see Annexe 2). From this, we represented the position of the 186 species based on their associated traits into the functional space, plotted in 4-dimensional space for graphical reasons. Here, we can assume that the reference functional space for Tikehau includes 186 species with their associated traits, projected in a multidimensional space (Fig. 12).

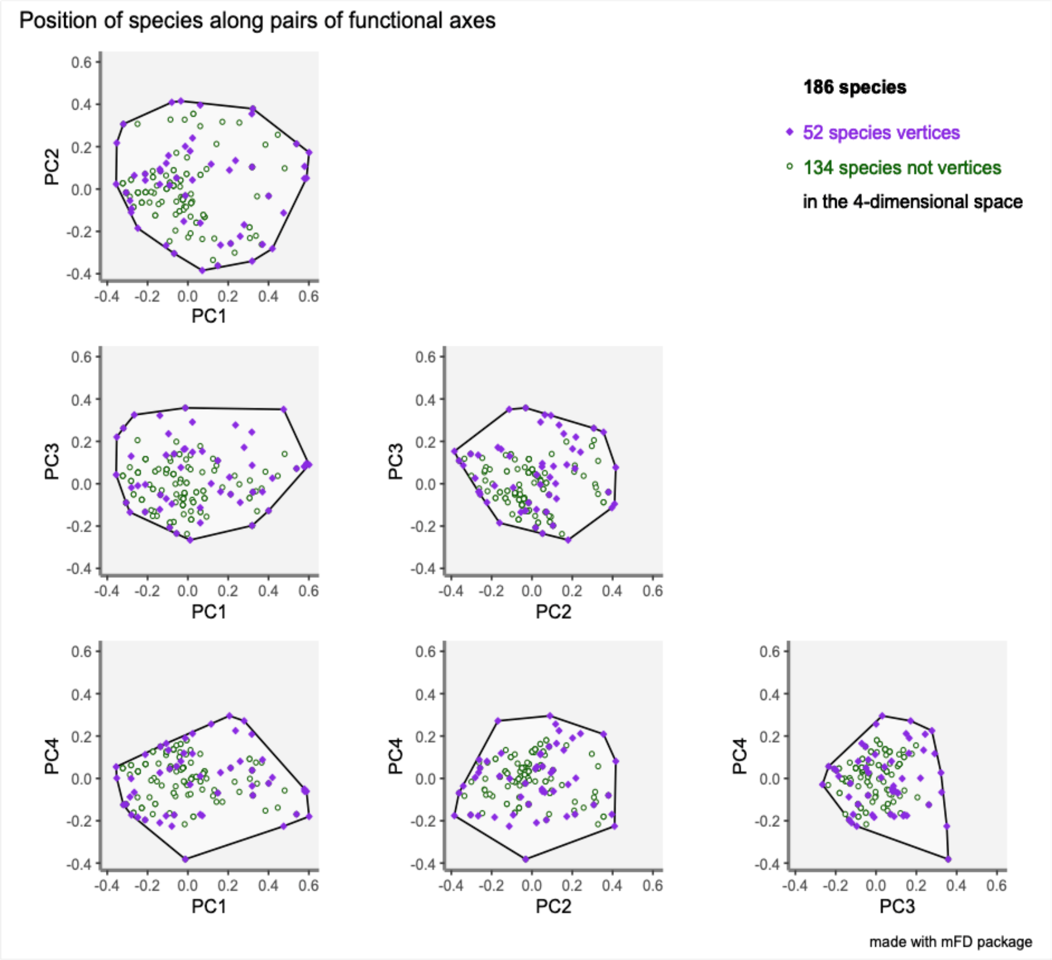


Figure 12. Functional space for fish species in coral reefs of the Tikehau atoll.

Using the multidimensional space occupied by fish species, we can identify changes in community composition along time using beta-diversity, whether changes in community result in species replacement (turnover) or species loss (nestedness) (Fig. 13 – Beta). Taking the example of two different years (2004 and 2022), the functional dissimilarity between the two years reaches 0.6271 (or 63%), from which almost the entire difference (59%) comes from the loss (or absence) of functions in 2004 (Fig. 14). Major community changes occurred in 2004 or earlier, with signs of strong resilience from this site.

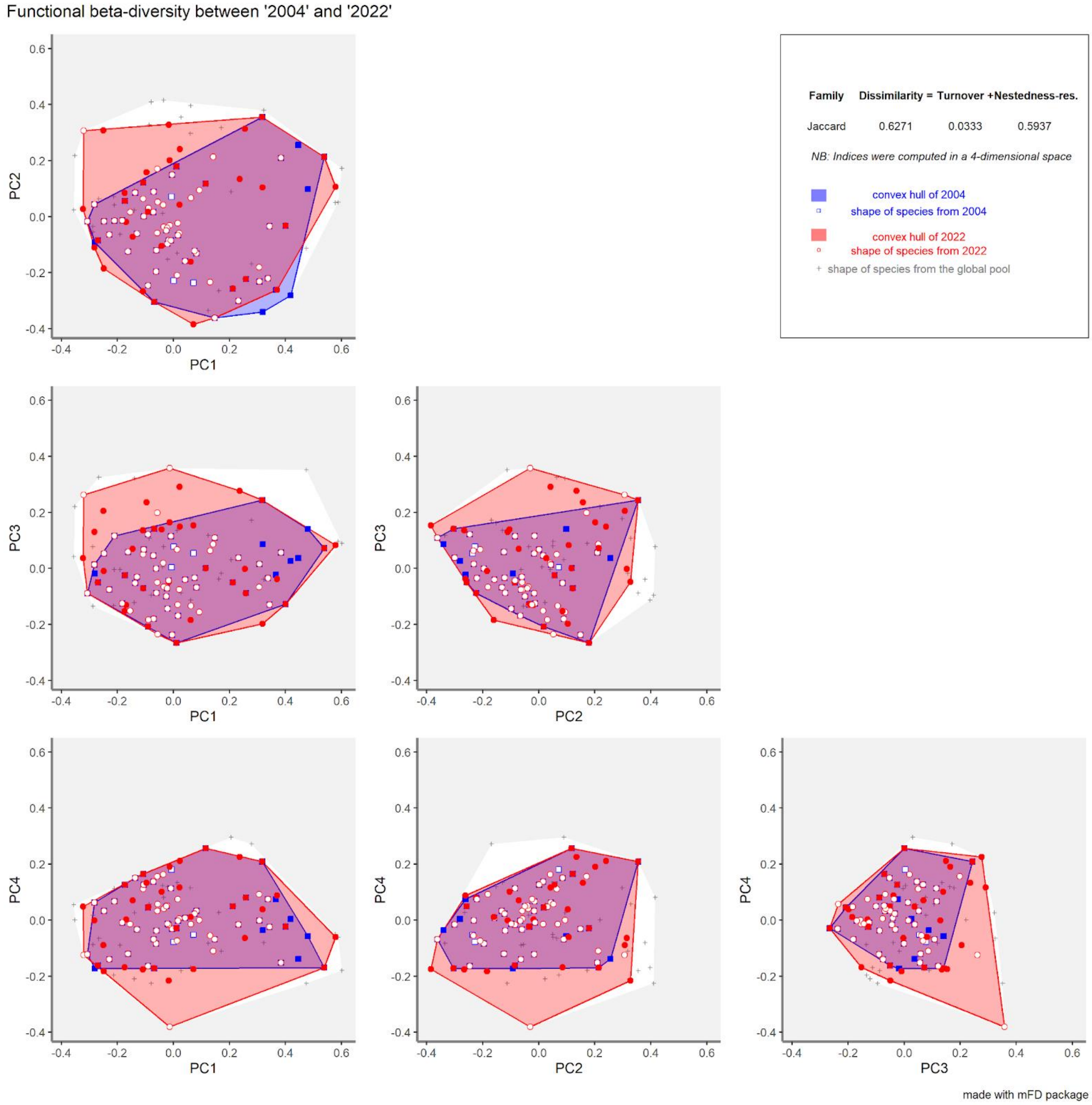


Figure 13. Reef fish functional beta-diversity between 2004 and 2022 for the monitoring site of Tikehau.

From this comparative overview between the two sites, the mFD package allows computing many Alpha Functional Diversity indices (Alpha diversity see Annexe 1), each providing a different facet of fish community. From the temporal analysis, the difference between the Functional richness between 2004 (0.264) and 2022 (0.691) is significant. The change of observers between 2004-2010 and 2010-2022 could partly explain the difference. 2012 and 2022 appeared to be the years when the functional richness was the highest (0.772 and 0.691, respectively) (Fig. 14). It has been observed that the reef in Tikehau is extremely dynamic and resilient to strong perturbations (hurricanes, crown-of-thorn starfish bloom, bleaching events), with fish populations certainly affected in some ways by those events as well.

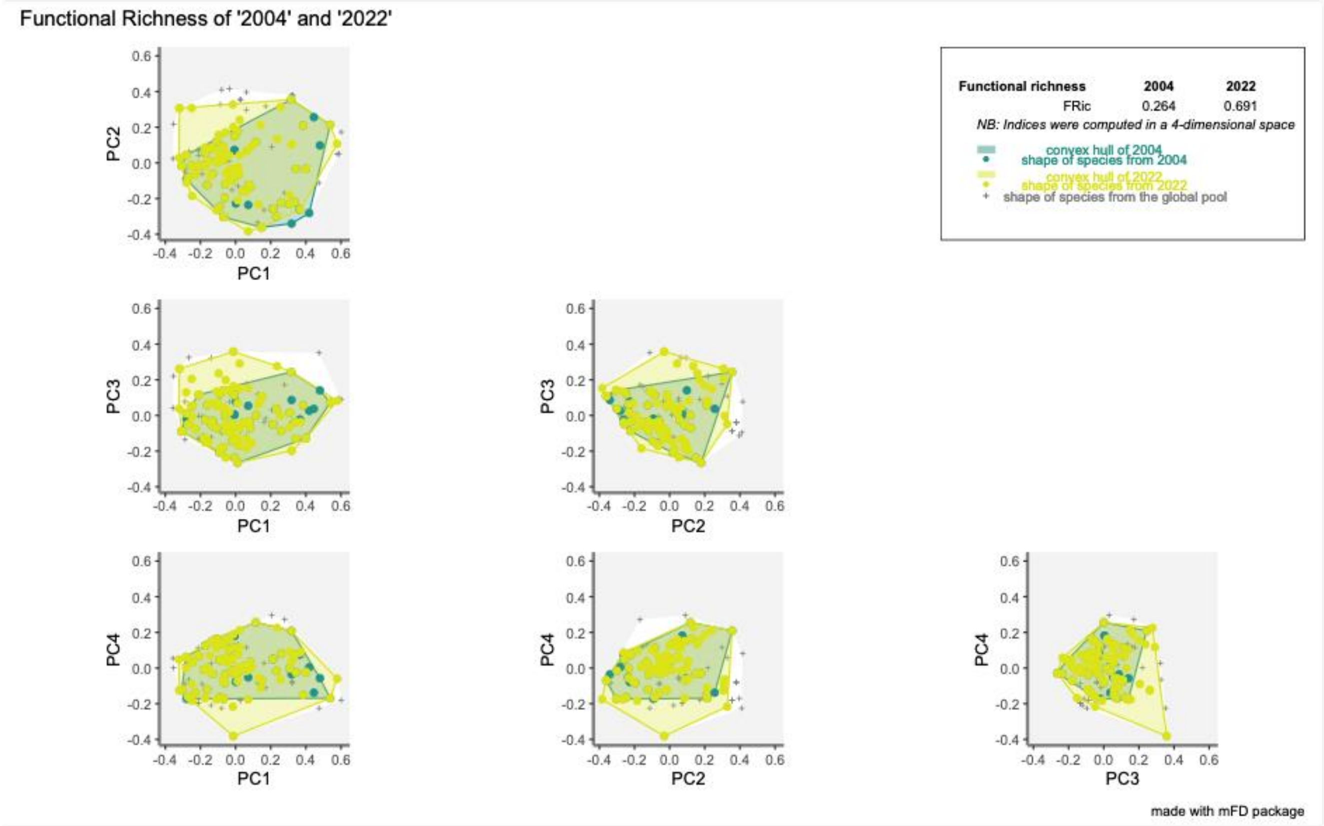


Figure 14. Functional richness of fish assemblages in Tikehau, French Polynesia, between 2004 and 2022.

Table 3 summarises the functional biodiversity indices of fish assemblages for each monitoring year. Looking more particularly at FRic, FDiv, FEve and FOri, we could consider 2022 as referent condition (High), 2010 as medium condition, and 2004 as degraded (low) condition (See summary table).

Table 3. Functional indices for fish assemblages in Tikehau, French Polynesia.

Year	Sp. richn	FDis	FEve	FRic	FDiv	FOri	FSpe
2004	67	0,388	0,394	0,264	0,783	0,085	0,444
2006	78	0,405	0,367	0,287	0,794	0,066	0,411
2008	104	0,484	0,388	0,424	0,837	0,054	0,430
2010	87	0,463	0,359	0,438	0,834	0,048	0,438
2012	121	0,548	0,395	0,772	0,913	0,098	0,532
2014	90	0,589	0,424	0,628	0,842	0,044	0,526
2016	90	0,539	0,426	0,553	0,888	0,048	0,520
2018	104	0,517	0,439	0,665	0,852	0,051	0,486
2020	86	0,397	0,406	0,426	0,840	0,044	0,472
2022	114	0,539	0,408	0,691	0,924	0,037	0,533

d. Community ecology

In French Polynesia, coral reef communities are subject to dynamic cycling changes, particularly in the aftermath of hurricane impacts or bleaching events. Scientific studies have highlighted the significance of hurricanes in shaping the resilience of these ecosystems. Blackwood *et al.* (2011) investigated the long-term effects of hurricanes on coral reefs in the region and found that intense storms can cause significant damage to coral communities, leading to a reduction in coral cover and diversity. However, Darling et al. (2017) and Alvarez-Filip et al. (2019) demonstrated that coral reefs with higher initial coral cover and genetic diversity exhibited greater resilience, with faster recovery rates after hurricanes. These findings suggest that the cycling changes in reef communities can be influenced by the frequency and intensity of hurricanes or other environmental events, as well as the pre-existing ecological condition of the reefs (Vercelloni *et al.* 2019). Coral bleaching events, primarily driven by elevated sea temperatures, have caused substantial coral mortality in certain regions (Penin et al. 2007).

The French Polynesian outer-reef system displays remarkable resilience, with coral cover fully recovering within 5-10 years after intense disturbances. Recovery trajectories of coral communities exhibit a symmetrical-sigmoid shape, indicating similar regulatory processes governing habitat colonization and saturation at a large spatial scale. Dominant coral taxa, such as Pocillopora, Acropora, and Porites, exhibit varying recovery rates based on their life history characteristics. Despite differing vulnerability levels of taxa to disturbances and diverse disturbance histories on reefs, the recovering coral populations and communities converge toward their pre-disturbed states, preserving community abundances and structures (Vercelloni *et al.* 2019).

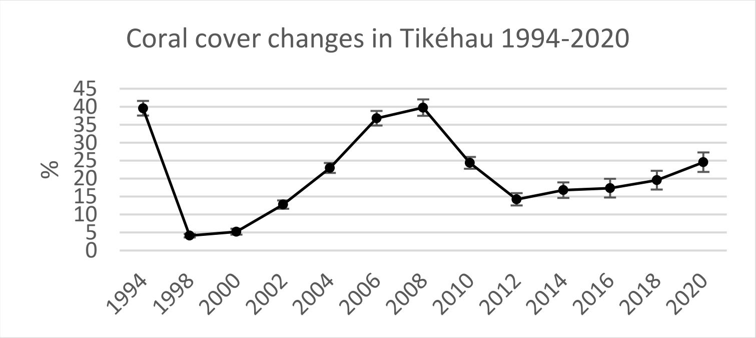


Figure 15. Changes in coral cover at Tikehau monitoring site between 1994 and 2020 showing dynamic cycling change effect, particularly after bleaching impacts. (Source data: CRIOBE IRCP).

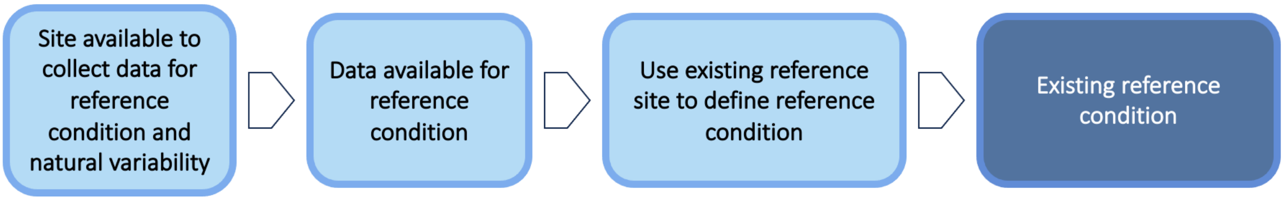
e. Summary table

<table>
<tr><td></td><td colspan="7">HABITAT</td></tr>
<tr><td>Variable</td><td>Spatial distribution
1.4 Habitat distribution: 1.4.1</td><td colspan="2">Fragmentation
1.4 Habitat distribution: 1.4.2</td><td colspan="2">Habitat loss/increase
1.5 Habitat extent: 1.5.1 – 1.5.2</td><td colspan="2">Connectivity
1.4 Habitat distribution: 1.4.2</td></tr>
<tr><td>Index</td><td>Surface extent</td><td colspan="2">Patch size</td><td colspan="2">Loss/Increase</td><td colspan="2">Connectivity</td></tr>
<tr><td>Values</td><td>37 km^2</td><td colspan="2">All Large</td><td colspan="2">na</td><td colspan="2">Highly connected</td></tr>
<tr><td></td><td colspan="7">BIODIVERSITY</td></tr>
<tr><td>Variable</td><td colspan="2">Functional space
6.2 Condition of benthic community: 6.2.1 – 6.2.2</td><td colspan="5">Functional diversity
6.2 Condition of benthic community: 6.2.2
4.3 Abundance/distribution of key trophic groups/species: 4.3.1
3.3 Population age/size distribution: 3.3.1 – 3.3.4</td></tr>
<tr><td>Index</td><td colspan="2">Functional space</td><td colspan="2">FRic</td><td>FEve</td><td>FDiv</td><td>FOri</td></tr>
<tr><td rowspan="3">Values</td><td colspan="2" rowspan="3">186 species of fish and associated traits
% change of functional space
Beta diversity between referent year in 2022 and other years.</td><td colspan="2">High 2022: 0.691</td><td>High 2022: 0.408</td><td>High 2022: 0.924</td><td>High 2022: 0.037</td></tr>
<tr><td colspan="2">Medium 2010: 0.438</td><td>Medium 2010: 0.359</td><td>Medium 2010: 0.834</td><td>Medium 2010: 0.048</td></tr>
<tr><td colspan="2">Low 2004: 0.264</td><td>Low 2004: 0.394</td><td>Low 2004: 0.783</td><td>Low 2004: 0.085</td></tr>
<tr><td></td><td colspan="7">COMMUNITY</td></tr>
<tr><td>Variable</td><td colspan="7">Community assemblages
1.1 Species distribution: 1.1.3
1.2 Population size: 1.2.1
2.1 Invasive species: 2.1.1</td></tr>
<tr><td>Index</td><td colspan="7">Coral reef cover</td></tr>
<tr><td>Values</td><td colspan="7">Corals Reef slope: 60% - 1984
Corals Reef flat: 10% - 1984
Dynamic cycling effect of coral assemblages recovery.</td></tr>
</table>

3. FRENCH GUIANA - MANGROVE FORESTS

a. Decision tree

Following the decision tree elaborated in MOVE-ON D.2.1.a, we analysed the context of French Guiana and selected the most appropriate method for defining reference conditions:



b. Habitat mapping

Detailed habitat data are available for French Guiana mangrove forests to produce maps at the global and regional scales, compatible with ecosystem condition assessment (Fig. 16).

Figure 16. Extent of mangrove forests in French Guiana (Source: CARNAMA 2020).

c. Functional biodiversity

Mangrove species present in the French Caribbean Outermost Regions (French Guiana, Martinique and Guadeloupe) were retrieved from Trégarot et al. (2020). A total of 12 species of mangrove trees, shrubs, and ferns have been recorded, which will represent the whole functional space for the Caribbean.

All species were classified according to 10 different life-history traits to account for their tolerance to salinity (tidal zone and estuarine position), their ability to attenuate wave energy (growth form and roots type), to oxygenate and stabilize the sediment (roots type), their productivity (leaf size) and their reproductive strategy (flowering and fruiting) (Table S1 in Supplementary Materials). The final traits selected are the following: (1) Tidal zone: high, mid, low; (2) Estuarine position (downstream, intermediate, upstream; (3) Size (height in m); (4) Growth type: fern, shrub, shrub-tree, tree; (5) Roots: pneumatophores, below-ground, knee roots, sturdy props lenticellate; (6) Leaf size (length in cm); (7) Inflorescence type: spike, raceme, panicle, dichasia, solitary, absent, branching; (8) Fruit: capsule, drupe, nutlet, pod, legume; (9) Release: spore, seeds, propagule, ordered; (10) Germination: unknown, spore, epigeal, hypogeal, viviparous, cryptoviviparous. Traits values were retrieved from Duke *et al.* (2014).

The next step toward the computation of functional diversity indices is to estimate the functional trait-based distance between species in order to build the functional space in which indices will be computed. We used the Gower Distance metric to compute distances as we have non-continuous traits. To generate a multidimensional space in which diversity indices are computed, we performed a PCoA using the trait-based distances and evaluated the quality of PCoA-based multidimensional spaces according to the deviation between trait-based distances and distances in the functional spaces. From this, we computed the functional space in 4 dimensions for graphical reasons (Fig. 17).

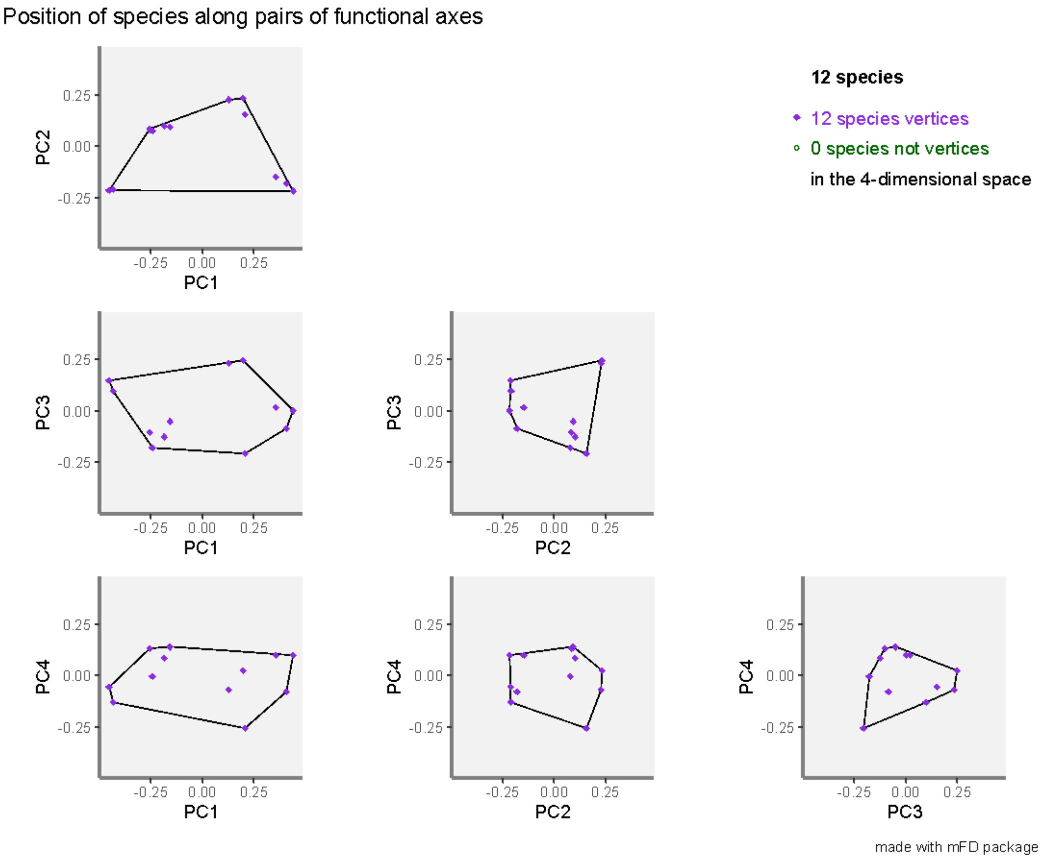
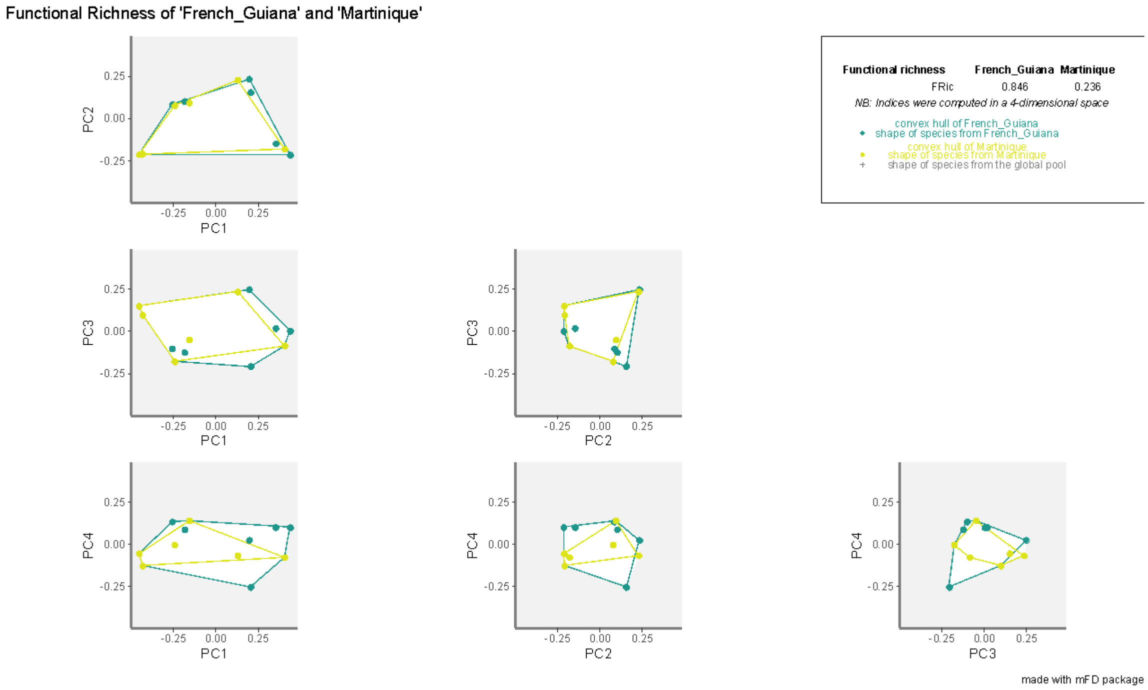


Figure 17. Functional space for tree species in mangrove forests in French Guiana.



Functional richness	French_Guiana	Martinique
FRic	0.846	0.236

NB: Indices were computed in a 4-dimensional space

Figure 18. Comparison of functional richness of mangrove assemblages in the Caribbean, between French Guiana and Martinique.

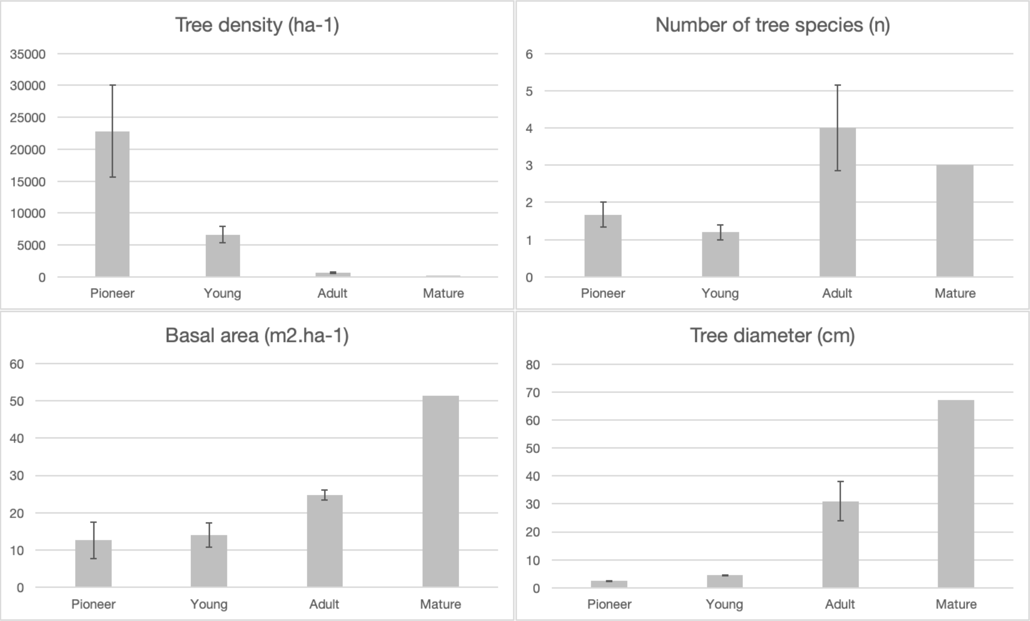
Table 4 provides functional biodiversity indices and species richness of mangrove trees for French Guiana, Martinique and Guadeloupe.

Table 4. Functional indices for true mangrove species assemblages in the French Overseas Regions of the Caribbean.

Site	sp_richn	FDis	FEve	FRic	FDiv	FOri	FSpe
French Guiana	11	0,851	0,640	0,846	0,817	0,320	0,747
Martinique	6	0,815	0,709	0,236	0,814	0,270	0,759
Guadeloupe	8	0,858	0,714	0,675	0,839	0,360	0,762

d. Community ecology

Given the strong dynamism of the mangrove forests of French Guiana, it would not be representative to provide a single value for tree density, basal area, stand height, biomass above ground and number of species (Fig. 19). In fact, between the different stages of development (Pioneer, Young, Adult, Mature, Mixed, or Cemetery Stand), we can expect very different forest structures retrieved from Fromard et al. (2004). Considering the mangroves of French Guiana are still today in a very good condition, we can consider the value in Table 5, representative of a reference condition, at least for the Sinnamary area, but this condition status could be extended further given the pristine condition and low pressure along the coastline of French Guiana.



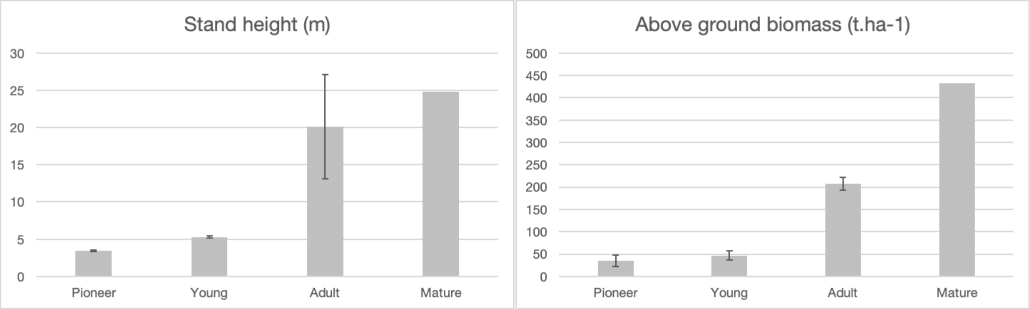


Figure 19. Average values of parameters describing the ecological condition of mangrove forests in French Guiana.

e. Summary table

	HABITAT			
Variable	**Spatial distribution 1.4 Habitat distribution: 1.4.1**	**Fragmentation 1.4 Habitat distribution: 1.4.2**	**Habitat loss/increase 1.5 Habitat extent: 1.5.1 – 1.5.2**	**Connectivity 1.4 Habitat distribution: 1.4.2**
Index	**Surface extent**	**Patch size**	**Loss/Increase**	**Connectivity**
Values	500-700 km^2	All Large	Natural fluctuations	Highly connected

	BIODIVERSITY				
Variable	**Functional space 6.2 Condition of benthic community: 6.2.1 – 6.2.2**	**Functional diversity 6.2 Condition of benthic community: 6.2.2 4.3 Abundance/distribution of key trophic groups/species: 4.3.1 3.3 Population age/size distribution: 3.3.1 – 3.3.4**			
Index	**Functional space**	**FRic**	**FEve**	**FDiv**	**FOri**
Values	11 species of trees and associated traits % change of functional space Beta diversity between territories, sites or years	High: 0.846	High: 0.640	High: 0.817	High: 0.320

	COMMUNITY
Variable	**Community assemblages 1.1 Species distribution: 1.1.3 1.2 Population size: 1.2.1 2.1 Invasive species: 2.1.1**
Index	**Community assemblages**
Values	Cf Table 5 for descriptor average values of stage development of mangrove forests.

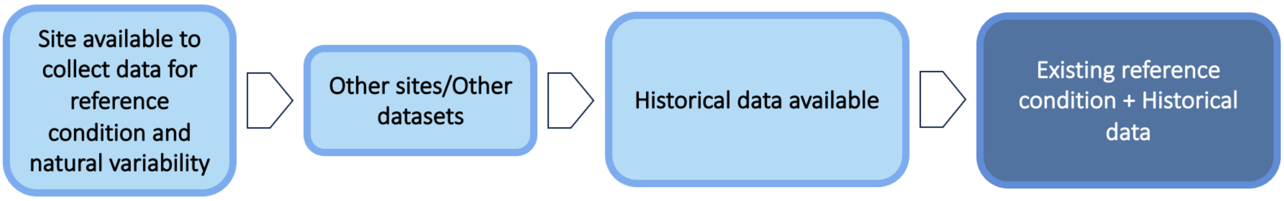
Table 5. Mangrove forest description parameters for four development stages (calculated from Fromard et al. 2004)

Mangrove stage	N of tree species (n)	Tree density (ha^{-1})	Basal area ($m^2.ha^{-1}$)	Tree diameter (cm)	Stand height (m)	Above ground biomass ($t.ha^{-1}$)
Pioneer (avg±stdEr)	1.67±0.3	22833.3±7242.1	12.6±4.9	2.5±0.1	3.4±0.8	34.4±13
Young (avg±stdEr)	1.2±0.2	6630.2±1258.9	14.0±3.3	4.5±0.1	5.3±0.2	46.3±10.4
Adult (avg±stdEr)	4.0±1.15	676.7±135	24.7±1.3	30.9±7	20.1±1.1	207.7±14.5
Mature (single value)	3.0	162.0	51.4	67.1	24.8	431.9

4. CANARY ISLANDS - SEAGRASS

a. Decision tree

Following the decision tree elaborated in MOVE-ON D.2.1.a, we analysed the context of the Canary Islands and selected the most appropriate method for defining reference conditions.



b. Habitat mapping

Potential distribution maps based on suitability analysis are available for the Canary Islands *Cymodocea* meadows, compatible with ecosystem condition assessment (Fig. 20).

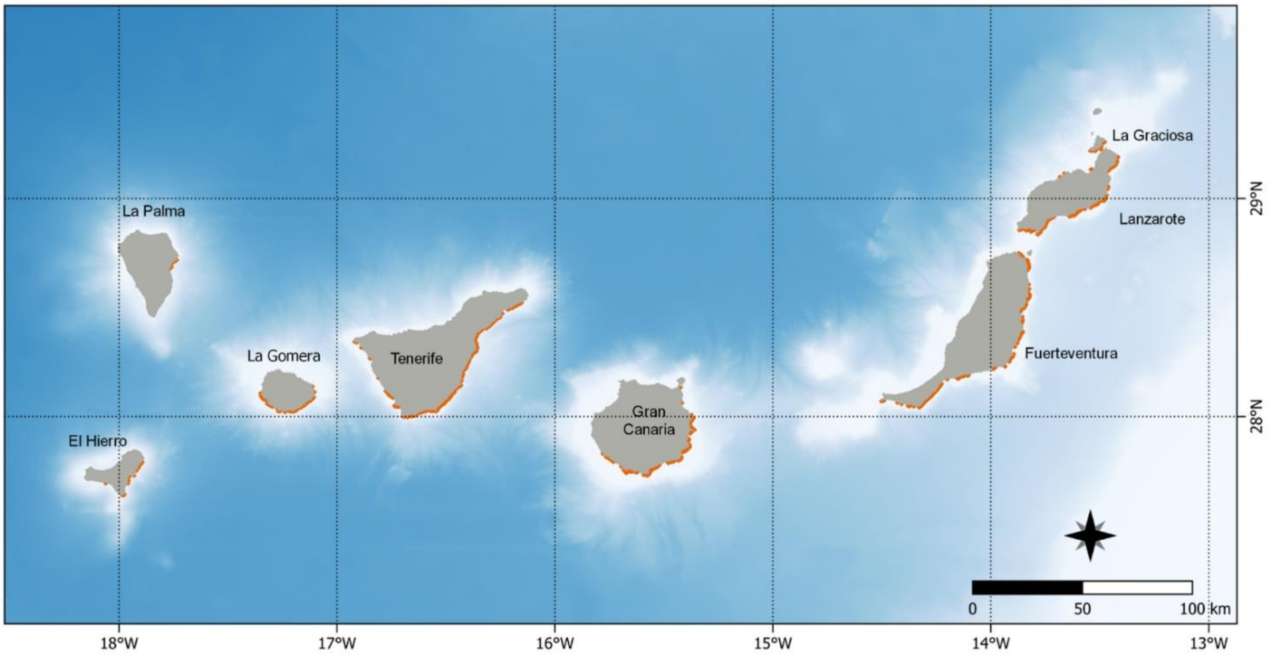


Figure 20. Potential distribution of *Cymodocea nodosa*, retrieved from Casas et al. (2021).

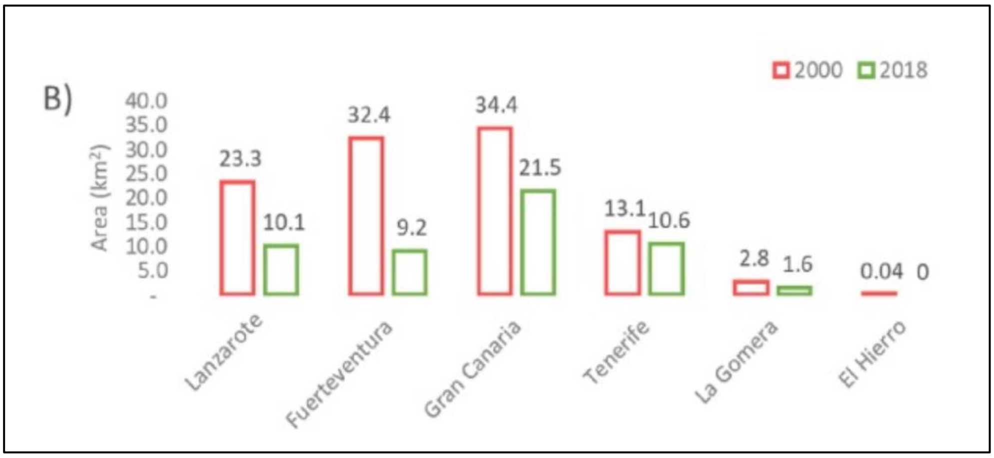


Figure 21. Seagrasses meadow area by islands between 2000 and 2018 in the Canary Islands (from Montero-Hidalgo *et al.* 2023).

Cymodocea seagrass beds have declined over the past 20 years by 50%, from 106 km^2 in 2000 to 53 km^2 in 2018, with a regression rate of 3 km^2 y^{-1} (Fig. 21). For the remaining 53 km^2 in 2018, only 7 km^2 has a high seagrass cover (Montero-Hidalgo et al. 2023).

c. Functional biodiversity

Fish assemblage data were retrieved from Tuya *et al.* (2014) and Espino Rodríguez (2020). In 2003, fish were sampled from June to September in three islands (Fuerteventura, Lanzarote and Gran Canaria) using a 6 m long, 4 m wide, 0.5 m high seine net with a mesh size of 1 mm. The net was towed over the seabed by two SCUBA divers following a 25 m transect. This technique captures small fishes that have reduced swimming capacities.

In 2011, fish were sampled in Gran Canaria through an annual cycle (February, May, August, November). Juveniles were sampled through the seine net; adults were sampled through underwater visual census (Tuya *et al.* 2014).

A total of 58 fish species have been recorded, which will represent the whole functional space. All species were classified according to six different life-history traits using functional properties defined by Mouillot et al. (2014): (1) Species maximum body size: <7 cm, 8–15 cm, 16–30 cm, 31–50 cm, 51–80 cm or >80 cm; (2) Mobility: sedentary (including territorial species), mobile or very mobile; (3) Period of activity: diurnal, nocturnal, or both; (4) Schooling: solitary, pairing, small groups (3–20 individuals), medium groups (20–50 individuals) or large groups (>50 individuals); (5) Position in the water column: benthic (species associated with the bottom), bentho-pelagic, or pelagic and (6) Trophic group: herbivores-detritivores (feed upon turf and filamentous algae and/or detritus), macroalgae-feeders (large fleshy algae and/or seagrass), sessile invertebrate feeders (e.g., corals, sponges, ascidians), mobile invertebrate feeders (benthic prey, such as crabs and mobile mollusks), planktivorous (small organism in the water column), piscivores (fish and cephalopods) or omnivores (both vegetal and animal material). Traits values were retrieved from Quimbayo *et al.* (2021) database and completed using FishBase.

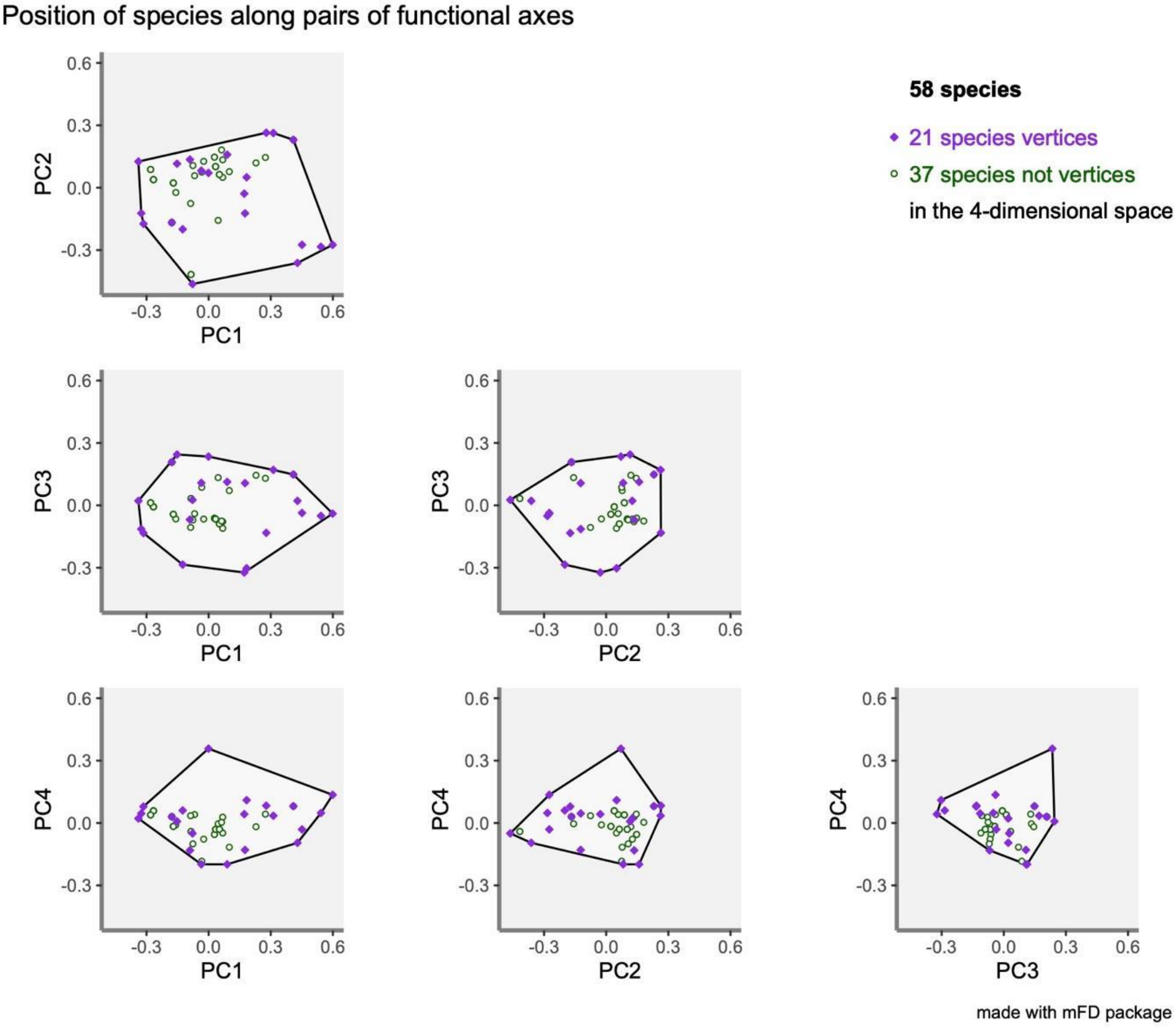


Figure 22. Functional space of fish assemblages in seagrass meadows of Canary Islands. Position of species along pairs of functional axes.

Based on the fish surveys conducted in *Cymodocea nodosa* meadows across the Canary Islands, back in 2003, we can theoretically consider that all 58 species and their associated traits, represent the referent functional space (Fig. 22). Using more historical surveys, this functional space could change slightly. Then, using the multidimensional space occupied by fish species, we can identify changes in community composition between islands, through beta-diversity, whether changes in community result in species replacement (turnover) or species loss (nestedness) (Fig. 22 – Beta). Taking the example of two different islands, namely Fuerteventura and Gran Canaria, the functional dissimilarity of fish assemblages associated with seagrass beds reaches 0.5675 (or 57%), from which 39% comes from turnover (replacement of species occupying the same function) and 18% from nestedness (Fig. 23).

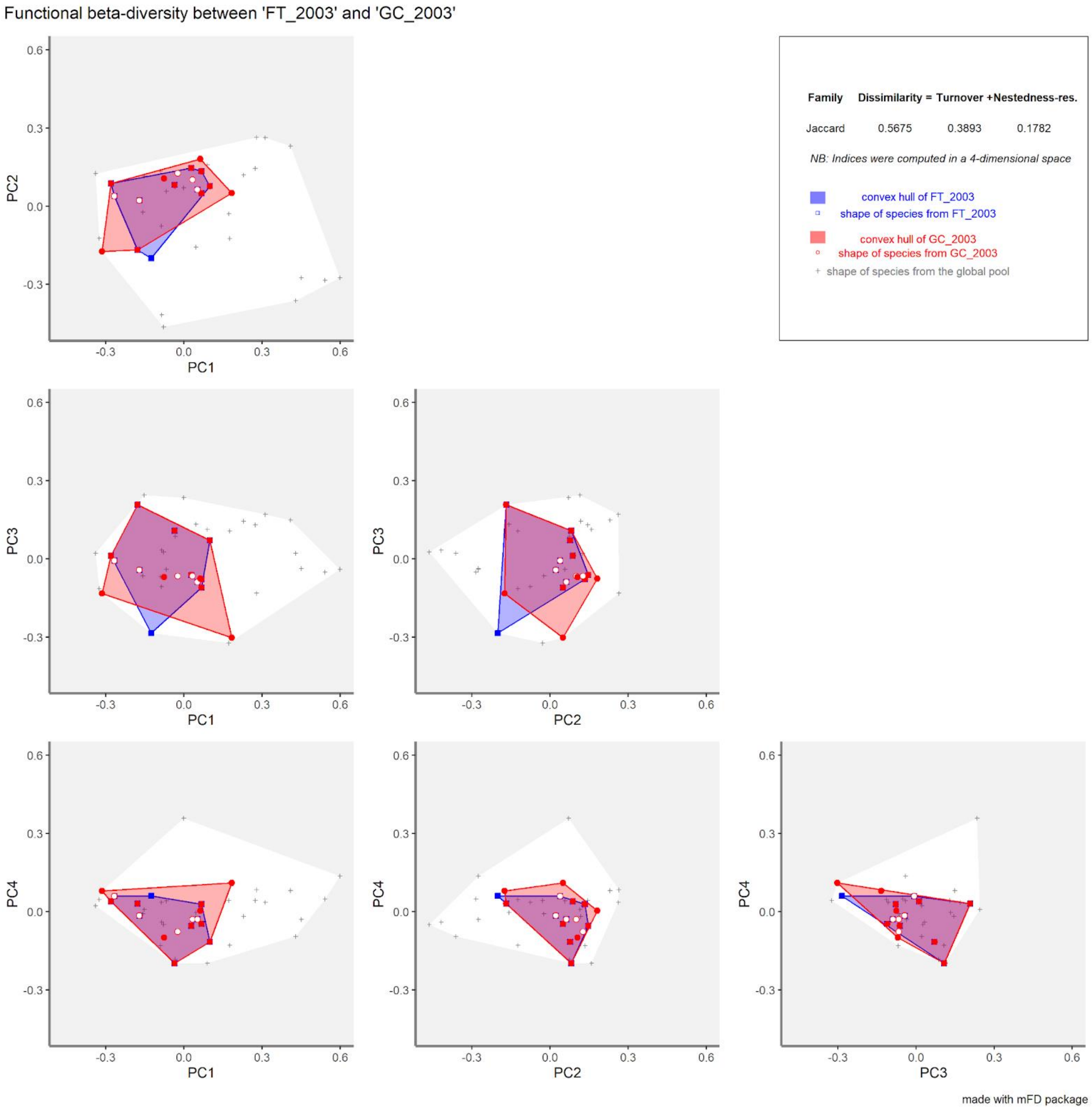


Figure 23. Seagrass beds functional beta-diversity between sites in the Canary Islands in 2003.

Looking more specifically into each assemblage, alpha functional diversity indices provide a different facet of the fish community. The functional richness from 2003 fish assemblages was low (Fuerteventura: 0.034 and Gran Canaria: 0.051 – Fig. 24) compared to all Canary Islands combined (FRic: 0.361), and less than Gran Canaria in 2011 (0.815) (Table 6). The reason behind could be in the sampling methods, or the number of sites per island which would greatly increase the probability of collecting additional species.

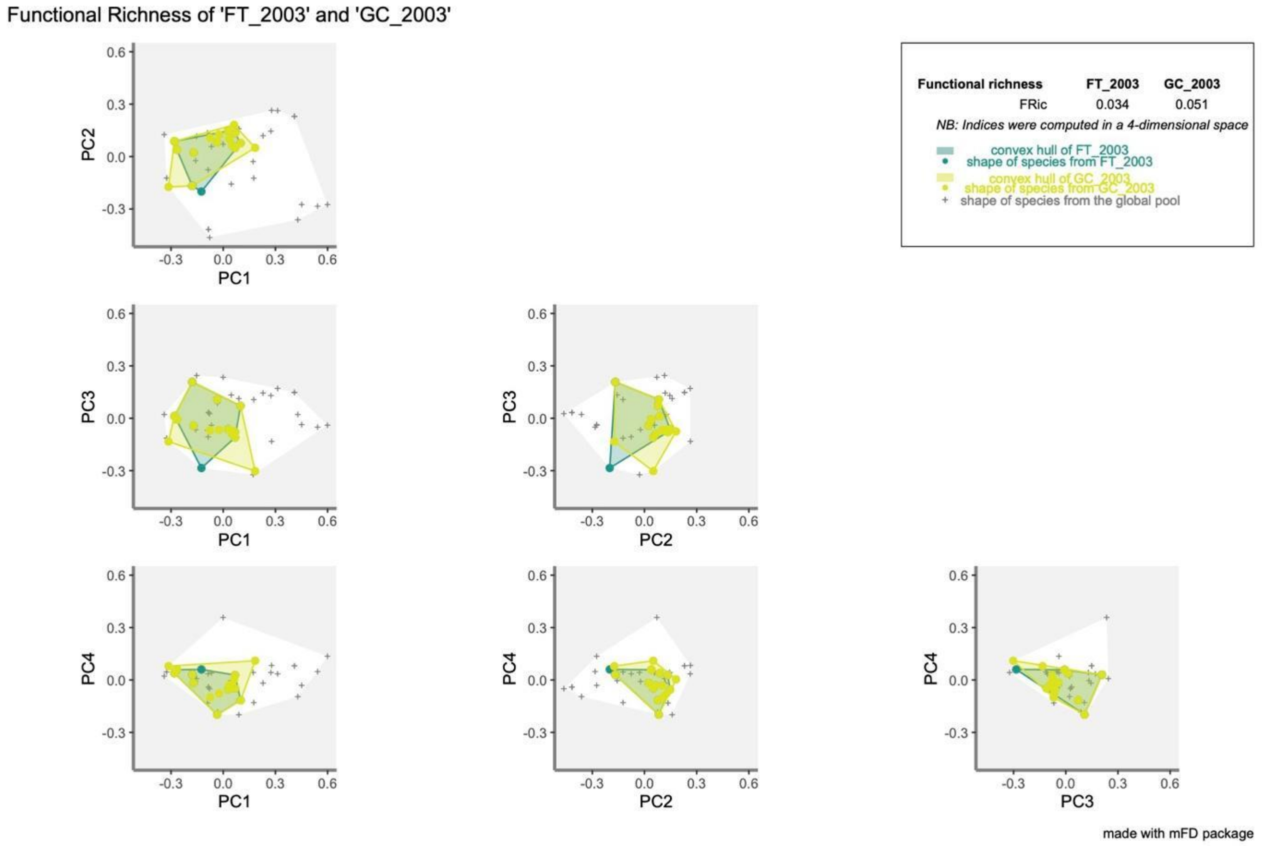


Figure 24. Functional richness of fish assemblages in Canary Islands, between Fuerteventura (FT) and Gran Canaria (GC), 2003.

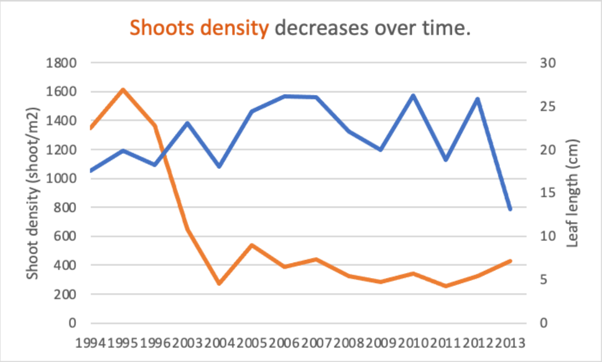
Table 6. Functional indices for fish assemblages in Cymodocea nodosa seagrass meadows of Canary Islands.

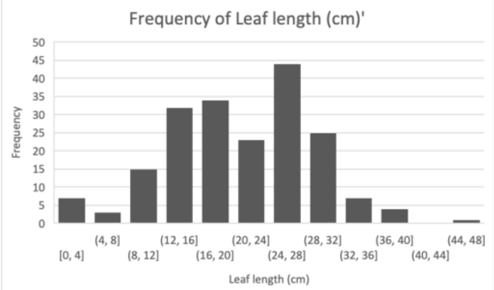
Site_year	sp_richn	FDis	FEve	FRic	FDiv	FOri	FSpe
Lanzarote_2003	17	0,138	0,263	0,036	0,717	0,022	0,245
Fuerteventura_2003	17	0,359	0,331	0,034	0,738	0,013	0,336
Gran_Canaria_2003	22	0,245	0,253	0,051	0,553	0,04	0,263
Islas_Canarias_2003	42	0,191	0,267	0,301	0,495	0,036	0,253
Gran_Canaria_2011	43	0,342	0,255	0,815	0,759	0,139	0,521

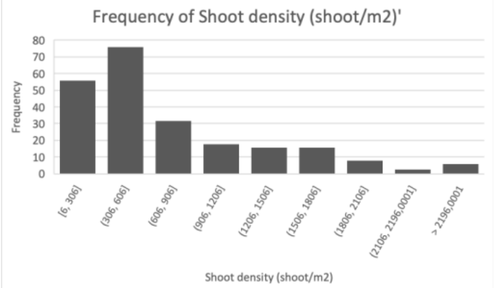
d. Community ecology

Seagrass indicators were retrieved from Fabbri *et al.* (2015). We used the longest time series to extract data on leaf length and shoots density. Values are reported in Fig. 25, which represents changes in shoots density over time, Leaf length and Shoots density frequencies. In all temporal series, there is a decrease in the average shoot density over time, and the maximum average is selected as a reference value for reference condition assessment.

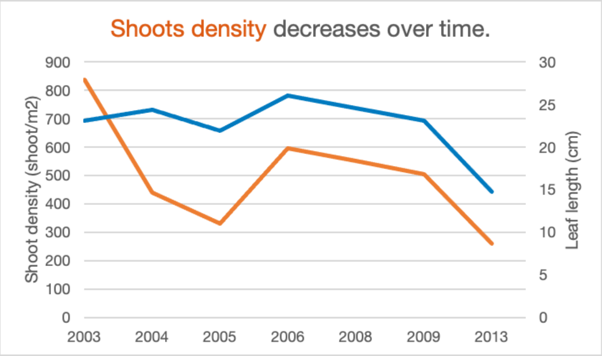
Gran Canaria

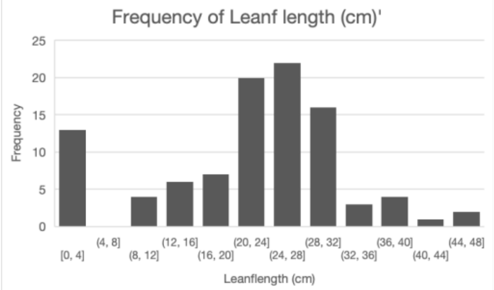


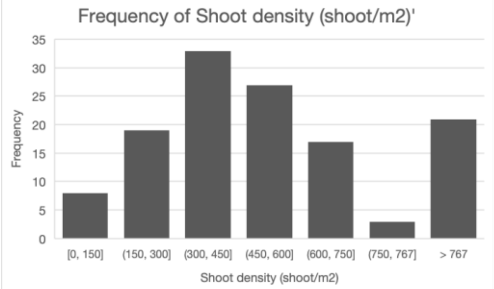




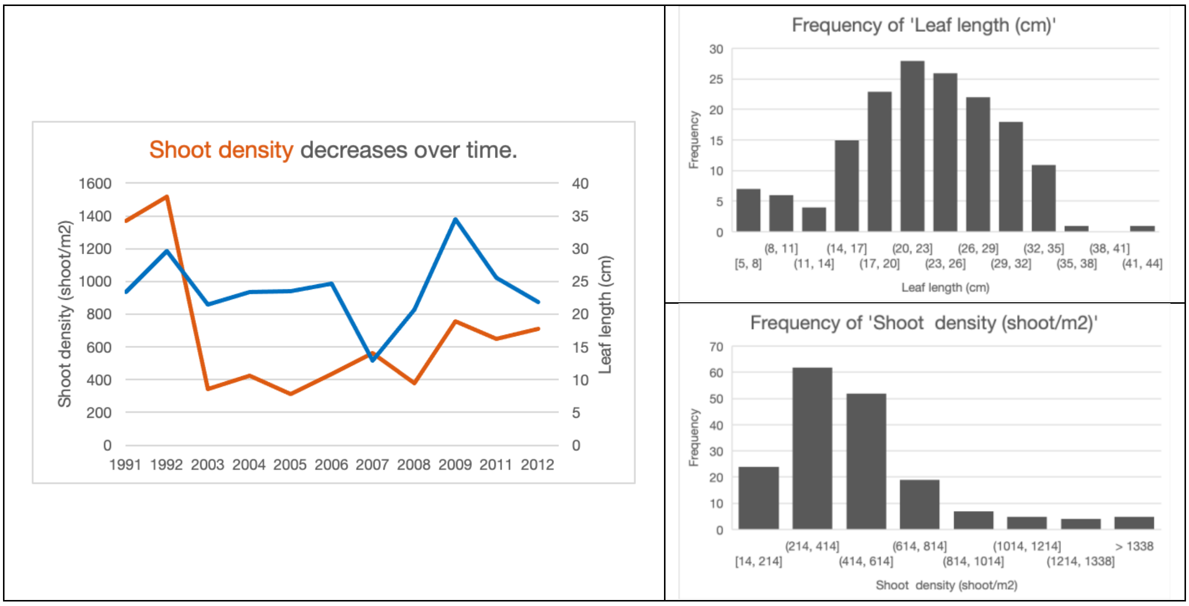
Fuerteventura



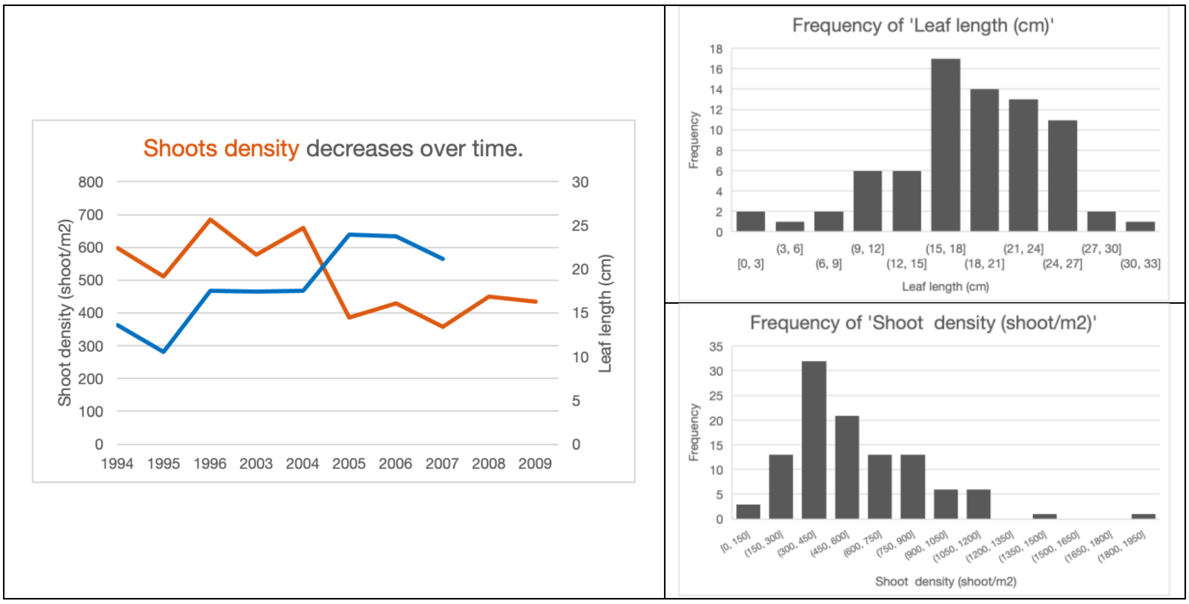




Tenerife



Lanzarote



	Gran Canaria	**Tenerife**	**Fuerteventura**	**Lanzarote**
Shoot density (avg±se)	1610±104	1371±193	837±127	684±52

Figure 25. Shoots density over time, Leaf length and Shoots density frequencies at four locations in the Canary Islands: Gran Canaria, Lanzarote, Tenerife and Fuerteventura and maximum average shoot density table for each location (Data source: Fabbri *et al.* (2015)).

e. Summary table.

	HABITAT			
Variable	**Spatial distribution** **1.4 Habitat distribution: 1.4.1**	**Fragmentation** **1.4 Habitat distribution: 1.4.2**	**Habitat loss/increase** **1.5 Habitat extent: 1.5.1 – 1.5.2**	**Connectivity** **1.4 Habitat distribution: 1.4.2**
Index	**Surface extent**	**Patch size**	**Loss/Increase**	**Connectivity**
Values	106 km^2	Small and Large	50%	Patch connectivity/isolation

	BIODIVERSITY				
Variable	**Functional space** **6.2 Condition of benthic community: 6.2.1 – 6.2.2**	**Functional diversity** **6.2 Condition of benthic community: 6.2.2** **4.3 Abundance/distribution of key trophic groups/species: 4.3.1** **3.3 Population age/size distribution: 3.3.1 – 3.3.4**			
Index	**Functional space**	**FRic**	**FEve**	**FDiv**	**FOri**
Values	58 species of fish and associated traits % change of functional space Beta diversity between territories, sites or years	GC_2011: 0.815	GC_2011: 0.255	GC_2011: 0.759	GC_2011:0.139

	COMMUNITY
Variable	**Community assemblages** **1.1 Species distribution: 1.1.3** **1.2 Population size: 1.2.1** **2.1 Invasive species: 2.1.1**
Index	**Community assemblages**
Values	106 km^2 → 2000 53 km^2 → 2018 7% of high density = 3,7 km^2 Avg max shoot density Gran Canaria 1610±104 Tenerife 1371±193 Fuerteventura 837±127 Lanzarote 684±52

5. FALKLAND ISLANDS - KELP FORESTS

a. Decision tree.

Following the decision tree elaborated in MOVE-ON D.2.1.a, we analysed the context of the Falklands Islands and selected the most appropriate method for defining reference conditions.

Site available to collect data for reference condition and natural variability

Data available for reference condition

Use existing reference site to define reference condition

Existing reference condition

b. Habitat mapping.

Detailed habitat data are available for the Falkland Islands kelp forests to produce maps at the regional scales, compatible with ecosystem condition assessment (Fig. 26).

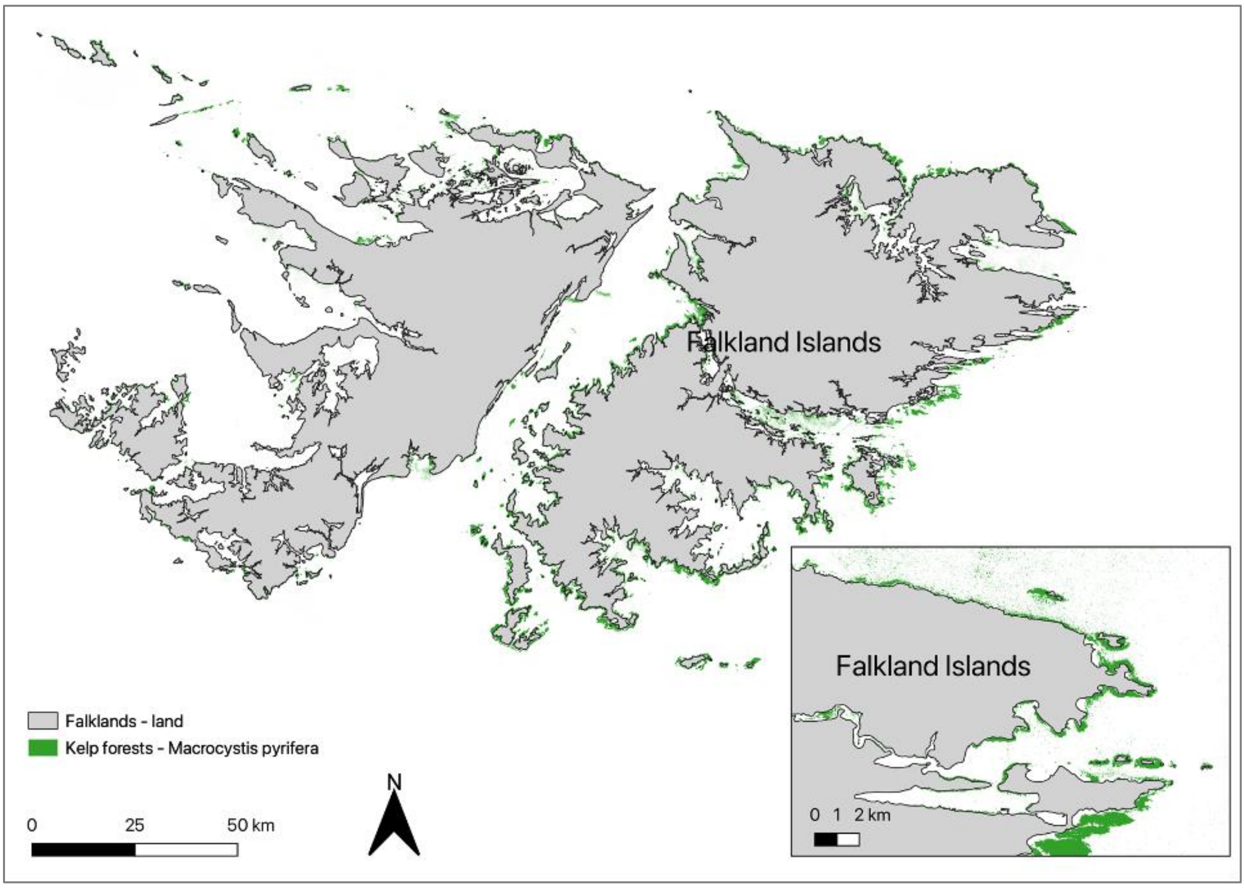


Figure 26. Extent of kelp forests in the Falkland Islands (adapted from Golding *et al.* 2019).

Current kelp distribution was mapped using image classifications based on Sentinel 1 (band 1) and Sentinel 2 (all 10 m bands) satellite imagery; Shuttle Radar Topography Mission (SRTM) data; and Landsat 8 (band 1) inputs within Google Earth Engine (Golding *et al*. 2019). The estimated total coverage of kelp forest surrounding the Falkland Islands was 830.1 km^2 in 2019.

c. Functional biodiversity

In the absence of monitoring data, the list of marine species found in the Falkland Islands was retrieved from Neely and Brickle (2013) identification guide to the intertidal and subtidal life of the Falkland Islands. A few taxa were targeted from this list, such as the Arthropods, Molluscs and Echinoderms for a total of 113 species.

All species were classified according to three different life-history traits using functional properties (1) Species maximum body size: <5 cm (small), 5–10 cm (medium), >10 (large); (2) Mobility: sessile, sedentary, crawler, swimmer; and (3) Morphology: articulate, bivalved, cushion, flat, globose, stellate, turbinate, vermiform. Traits values were retrieved from Neely and Brickle (2013) and completed using SeaLife-Base and expert knowledge. From this, we computed the functional space – for graphical reasons, we can only plot in 4-dimensional space (Fig. 27).

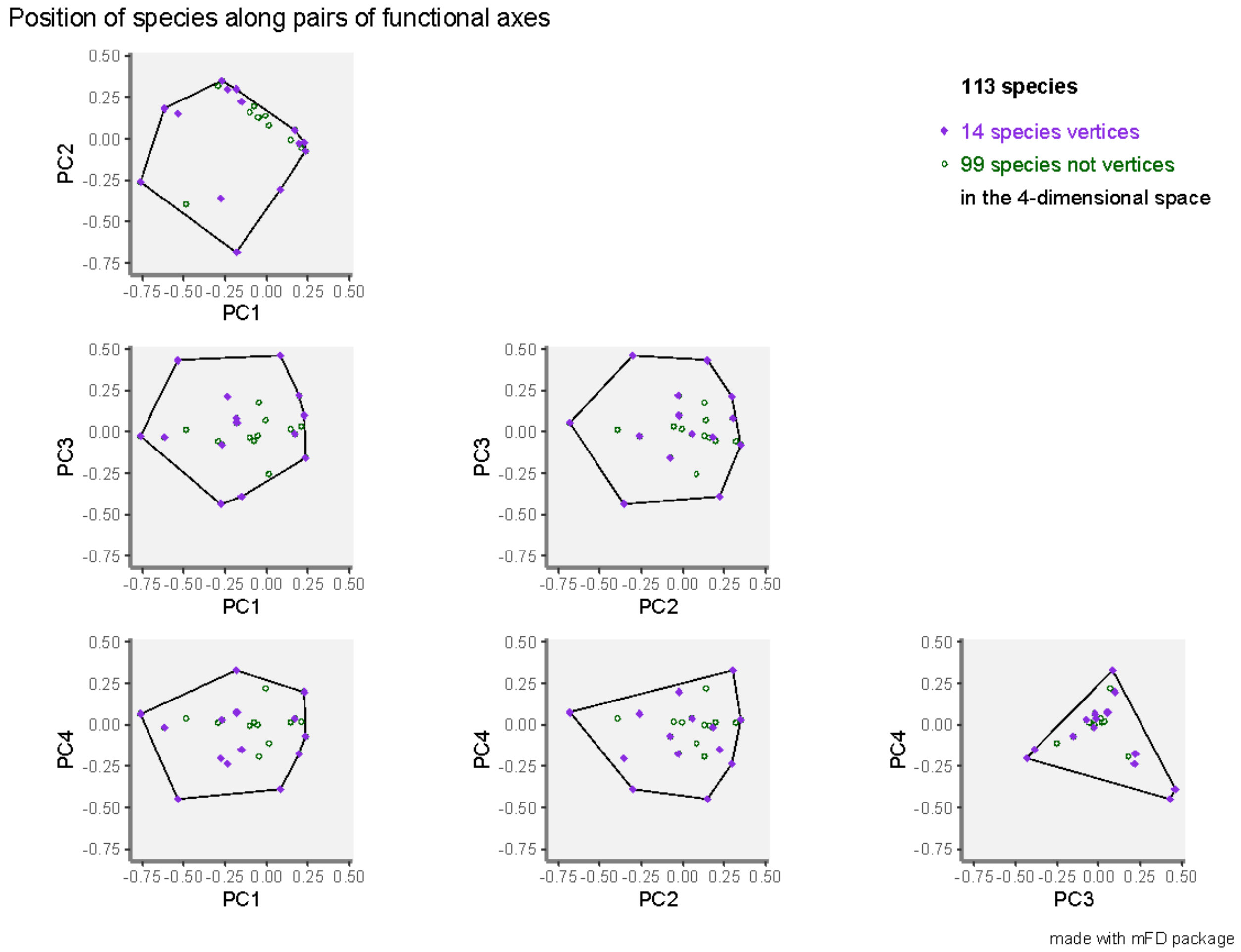


Figure 27. Functional space of invertebrate assemblages in kelp forest of the Falkland Islands. Position of species along pairs of functional axes.

The mFD package allows computing many alpha Functional Diversity indices. However, without invertebrates data made available, we couldn't compute those indices, each providing a different facet of invertebrate communities. Presence/absence data or abundance data (if available) from different sites or different years could then be compared.

d. Community ecology

Kelp density was calculated based on field survey data collected from across the Falkland Islands between 2008 and 2016 (Shallow Marine Surveys Group, unpublished data), with a total of 315 surveys conducted between 2008 and 2016. Density values for *Macrocystis pyrifera* and *Lessonia spp.* were based on the number of individual giant kelp thali observed in-situ one metre on either side along a 20 m transect (i.e. 40 m total sample area), placed randomly on the seabed within the kelp forest rocky habitat. Density (thali/m^2) for each species was averaged for autumn (March – May) and spring (September-November) surveys to account for any seasonal changes in density as the forest grows and senesces. Kelp health values are assumed to be homogenous throughout the extent of mapped kelp.

Due to the remoteness of these islands (Jones *et al*. 2018), any such differences would be solely biophysically driven (i.e. through wave exposure).

Overall values of *Macrocystis pyrifera* density were highly variable, ranging between ~ 0.02 and 2.75 thalli/m across all surveys, with a mean spring value of 0.293 ± 0.051 thalli/m^2 (mean ± s.e) and a mean autumn density value of 0.249 ± 0.039 thalli/m^2, both averaged across all years. Overall density values of Lessonia spp. were again highly variable, ranging between 0.025 and 4.4 thali (whole plants)/m^2 across all surveys, with a mean value of 0.642 ± 0.069 thalli/m^2 in spring, and 0.716 ± 0.082 thalli/m^2 in autumn (see Bayley *et al*. 2021).

e. Summary table

	HABITAT			
Variable	**Spatial distribution** **1.4 Habitat distribution: 1.4.1**	**Fragmentation** **1.4 Habitat distribution: 1.4.2**	**Habitat loss/increase** **1.5 Habitat extent: 1.5.1 – 1.5.2**	**Connectivity** **1.4 Habitat distribution: 1.4.2**
Index	**Surface extent**	**Patch size**	**Loss/Increase**	**Connectivity**
Values	830 km^2	Small and Large	Natural fluctuations	Highly connected

	BIODIVERSITY				
Variable	**Functional space** **6.2 Condition of benthic community: 6.2.1 – 6.2.2**	**Functional diversity** **6.2 Condition of benthic community: 6.2.2** **4.3 Abundance/distribution of key trophic groups/species: 4.3.1** **3.3 Population age/size distribution: 3.3.1 – 3.3.4**			
Index	**Functional space**	**FR**	**Fev**	**Fdiv**	**For**
Values	113 species of invertebrates (Arthropods, Molluscs and Echinoderms)	na	na	na	na

	COMMUNITY
Variable	**Community assemblages** **1.1 Species distribution: 1.1.3** **1.2 Population size: 1.2.1** **2.1 Invasive species: 2.1.1**
Index	**Community assemblages**
Values	Macrocystis pyrifera autumn: 0.249 ± 0.039 thalli/m^2 spring: 0.293 ± 0.051 thalli/m^2 Lessonia spp. automn: 0.716 ± 0.082 thalli/m^2 spring: 0.642 ± 0.069 thalli/m^2

CONCLUSION

Defining ecosystem condition reference is crucial for understanding the extent of changes and impacts on ecosystems over time. Having access to historical data not only helps us to understand how far an ecosystem is from its referent condition, it also prevents the shifting baseline syndrome (SBS) which "describes a gradual change in the accepted norms for the condition of the natural environment due to a lack of human experience, memory and/or knowledge of its past condition" (Soga & Gaston, 2018). However, we acknowledge that the data used in this report do not necessarily reflect the pristine conditions, but do represent a reference point for the future.

Having an appropriate baseline to refer to is key for nature conservation, restoration and management, through, for instance, the assessment of ecosystem services provided by Nature. Knowing the referent condition, we can assess changes in the bundle of ecosystem services due to the degradation of ecosystems from climate change-related stressors or anthropogenic pressures, and understand their potential contribution following ecosystems restoration, protection and management.

The presence of near pristine ecosystems in the European and British ORs and OCTs is worth putting forward as an example for nature conservation and a reference point for nearby territories that do not have historical data.

In the context of data-limited environments in the overseas territories of the European Union (EU), the assessment of ecosystem condition poses significant challenges. The limited availability of monitoring data and sparse historical records impede our ability to comprehensively evaluate the health and status of marine ecosystems. Traditional indicator-based approaches may not be sufficient to capture the full complexity and functioning of these unique ecosystems.

Drawing attention to the limitations of existing indicators, it is essential to acknowledge that they might not fully capture the intricate dynamics and resilience of marine ecosystems in EU overseas territories. To overcome these limitations, there is a crucial need to explore novel approaches, such as the integration of functional traits of species into ecosystem assessment frameworks. Functional traits provide valuable insights into the ecological roles and responses of species to environmental changes, allowing us to infer ecosystem condition even with limited monitoring data.

Developing indicators based on functional traits can help fill data gaps and provide a more holistic understanding of ecosystem health. By considering traits such as feeding behaviour, reproductive strategies, and habitat associations, we can gain valuable information on the functional diversity and ecological roles of species within these marine ecosystems. Integrating functional traits can also offer insights into ecosystem functioning and the potential impacts of disturbances, even in the absence of comprehensive monitoring datasets.

Furthermore, the collaboration between scientists, policymakers, and local communities becomes crucial to foster a more comprehensive ecosystem assessment in EU overseas territories. Engaging local knowledge and traditional ecological knowledge can supplement scientific data and contribute to a more holistic understanding of ecosystem condition.

The assessment of ecosystem condition in EU overseas territories requires innovative approaches in the face of data limitations. Emphasizing the need to move beyond traditional indicators, the integration of functional traits can offer valuable insights into marine ecosystem health and functioning. Developing and implementing new indicators, alongside interdisciplinary collaborations, will be pivotal in fostering sustainable management and conservation efforts in these unique and valuable marine ecosystems.

REFERENCES

Alongi, D. M. (2015). The impact of climate change on mangrove forests. Current Climate Change Reports, 1, 30-39.

Alvarez-Filip, L., Estrada-Saldívar, N., Pérez-Cervantes, E., Molina-Hernández, A. and González-Barrios, F.J. (2019). A rapid spread of the stony coral tissue loss disease outbreak in the Mexican Caribbean. PeerJ, 7

Anderson, M.J., Crist, T.O., Chase, J.M., Vellend, M., Inouye, B.D., Freestone, A.L., Sanders, N.J., Cornell, H.V., Comita, L.S., Davies, K.F. and Harrison, S.P. (2011). Navigating the multiple meanings of β diversity: a roadmap for the practicing ecologist. Ecology letters, 14(1), pp.19-28.

Baselga, A. (2010). Partitioning the turnover and nestedness components of beta diversity. Global Ecology and Biogeography, 19(1), 134–143. doi:10.1111/j.1466-8238.2009.00490.x.

Baselga, A. (2017). Partitioning abundance-based multiple-site dissimilarity into components: Balanced variation in abundance and abundance gradients. Methods in Ecology and Evolution, 8 (7), 799–808. doi:10.1111/2041-210X.12693.

Bayley, D., Brickle, P., Brewin, P., Golding, N. and Pelembe, T. (2021). Valuation of kelp forest ecosystem services in the Falkland Islands: A case study integrating blue carbon sequestration potential. One Ecosystem, 6, e62811.

Beaton, E. C., Küpper, F. C., van West, P., Brewin, P. E. and Brickle, P. (2020). The influence of depth and season on the benthic communities of a Macrocystis pyrifera forest in the Falkland Islands. Polar Biology, 43, 573-586.

Bigot L. (2008). Evolution spatio-temporelle de la biodiversité et de la structure des communautés benthiques entre 1998 et 2008 sur les stations sentinelles GCRMN de La Réunion. Rapport ECOMAR pour le compte de APMR 32 p. + annexes

Blackwood, J.C., Hastings, A. and Mumby, P.J. (2011). A model-based approach to determine the long-term effects of multiple interacting stressors on coral reefs. Ecological Applications, 21(7), pp.2722-2733.

Bourmaud, C.A.F., Abouïdane, A., Boissier, P., Leclère, L., Mirault, E. and Pennober, G., 2005. Coastal and marine biodiversity of La Reunion.

Casas, E., Martín-García, L., Otero-Ferrer, F., Tuya, F., Haroun, R. and Arbelo, M. (2021). Economic mapping and assessment of Cymodocea nodosa meadows as nursery grounds for commercially important fish species. A case study in the Canary Islands. One Ecosystem, 6, e70919.

Chabanet (1994). Etude des relations entre les peuplements benthiques et les peuplements ichtyologiques sur le complexe récifal de Saint-Gilles/La Saline (Ile de la Reunion). Thèse de doctorat. Université de Perpignan/Université de la Reunion. 266p.

Cillauren E and David G (2019). Report on State of the Art of MAES in the participating regions. Deliverable 2.2, MOVE project, European Commission Directorate General Environment Grant Agreement no. 07.027735/2018/776517/SUB/ENV.D2

Darling, E.S., Graham, N.A., Januchowski-Hartley, F.A., Nash, K.L., Pratchett, M.S. and Wilson, S.K. (2017). Relationships between structural complexity, coral traits, and reef fish assemblages. Coral Reefs, 36, pp.561-575.

Duarte CM, Losada IJ, Hendriks IE, et al. (2020). The role of coastal plant communities for climate change mitigation and adaptation. Nature Reviews Earth & Environment, 1(6), 315-328.

Duke, N., Ball, M. and Ellison, J. (1998), Factors influencing biodiversity and distributional gradients in mangroves. Global Ecology & Biogeography Letters, 7: 27-47

Espino Rodríguez, F. (2020). Ictiofauna asociada a praderas de Cymodocea Nodosa en las Islas Canarias (Océano Atlántico Noreste) (Doctoral dissertation).

Figuerola B., Barnes D. K. A., Brickle P. and Brewin P. E. (2017). Bryozoan Diversity Around the Falkland and South Georgia Islands: Overcoming Antarctic Barriers. Mar. Environ. Res. 126, 81–94. doi: 10.1016/j.marenvres.2017.02.005

Friedlander, A.M., Ballesteros, E., Bell, T.W., Caselle, J.E., Campagna, C., Goodell, W., Hüne, M., Muñoz, A., Salinas-de-León, P., Sala, E. and Dayton, P.K. (2020). Kelp forests at the end of the earth: 45 years later. Plos one, 15(3)

Fromard, F., Vega, C. and Proisy, C. (2004). Half a century of dynamic coastal change affecting mangrove shorelines of French Guiana. A case study based on remote sensing data analyses and field surveys. Marine Geology, 208(2-4), pp.265-280.

Golding N, Black B, Blake D, Brewin P, Harte M, Havercroft H, James R, Jones G (2019). Long-term coastal habitat mapping & monitoring handbook. Examples based on work undertaken in the Falkland Islands & South Georgia. DPLUS065 Coastal Habitat Mapping project.

Hughes, T.P., Kerry, J.T., Baird, A.H., Connolly, S.R., Dietzel, A., Eakin, C.M., Heron, S.F., Hoey, A.S., Hoogenboom, M.O., Liu, G. and McWilliam, M.J. (2018). Global warming transforms coral reef assemblages. *Nature*, *556*(7702), pp.492-496.

Hughes, T.P., Anderson, K.D., Connolly, S.R., Heron, S.F., Kerry, J.T., Lough, J.M., Baird, A.H., Baum, J.K., Berumen, M.L., Bridge, T.C. and Claar, D.C. (2018). Spatial and temporal patterns of mass bleaching of corals in the Anthropocene. Science, 359(6371), pp.80-83.

Krumhansl, K.A., Okamoto, D.K., Rassweiler, A., Novak, M., Bolton, J.J., Cavanaugh, K.C., Connell, S.D., Johnson, C.R., Konar, B., Ling, S.D. and Micheli, F. (2016). Global patterns of kelp forest change over the past half-century. Proceedings of the National Academy of Sciences, 113(48), pp.13785-13790.

Letourneur, Y., Chabanet, P., DurviLLe, P., Taquet, M., Teissier, E., Parmentier, M., Quero, J.C. and Pothin, K. (2004). An updated checklist of the marine fish fauna of Reunion Island, south-western Indian Ocean. Cybium, 28(3), pp.199-216.

Magneville, C., Loiseau, N., Albouy, C., Casajus, N., Claverie, T., Escalas, A., Leprieur, F., Maire, E., Mouillot, D. and Villéger, S. (2022). mFD: an R package to compute and illustrate the multiple facets of functional diversity. Ecography, 2022(1).

Mirault, É. (2007). Les fonctions et enjeux socio-économiques des écosystèmes récifaux: une approche géographique des valeurs de l'environnement appliquée à l'île de la Reunion(Doctoral dissertation, Paris 10).

Montero-Hidalgo, M., Tuya, F., Otero-Ferrer, F., Haroun, R., and Santos-Martín, F. (2023). Mapping and assessing seagrass meadows changes and blue carbon under past, current, and future scenarios. Science of The Total Environment, 872.

Mora-Soto, A., Capsey, A., Friedlander, A.M., Palacios, M., Brewin, P.E., Golding, N., Dayton, P., Van Tussenbroek, B., Montiel, A., Goodell, W. and Velasco-Charpentier, C. (2021). One of the least disturbed marine coastal ecosystems on Earth: Spatial and temporal persistence of Darwin's sub-Antarctic giant kelp forests. Journal of Biogeography, 48(10), pp.2562-2577.

Mouillot, D., Villéger, S., Parravicini, V., Kulbicki, M., Arias-González, J.E., Bender, M., Chabanet, P., Floeter, S.R., Friedlander, A., Vigliola, L. and Bellwood, D.R. (2014). Functional over-redundancy and high functional vulnerability in global fish faunas on tropical reefs. Proceedings of the National Academy of Sciences, 111(38), pp.13757-13762.

Neely, K.L. and Brickle, P. (2013). Marine life of the Falkland Islands.

Penin, L., Adjeroud, M., Schrimm, M., and Lenihan, H. S. (2007). High spatial variability in coral bleaching around Moorea (French Polynesia): patterns across locations and water depths. Comptes rendus biologies, 330(2), 171-181.

Quimbayo, J.P., Silva, F.C.D., Mendes, T.C., Ferrari, D.S., Danielski, S.L., Bender, M.G., Parravicini, V., Kulbicki, M. and Floeter, S.R. (2021). Life-history traits, geographical range, and conservation aspects of reef fishes from the Atlantic and Eastern Pacific. Ecology.

Quod, J. P. (1999). Consequences of the 1998 coral bleaching event for the islands of the western Indian Ocean.

R Core Team 2023R Core Team (2023). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Soga, M., & Gaston, K. J. (2018). Shifting baseline syndrome: causes, consequences, and implications. Frontiers in Ecology and the Environment, 16(4), 222-230.

Solar, R. R. D. C., Barlow, J., Ferreira, J., Berenguer, E., Lees, A. C., et al. (2015). How pervasive is biotic homogenization in human-modified tropical forest landscapes? Ecology Letters, 18(10), 1108–1118. doi:10.1111/ele.12494.

Thomassin, A., White, C.S., Stead, S.S. and David, G., 2010. Social acceptability of a marine protected area: the case of Reunion Island. Ocean & Coastal Management, 53(4), pp.169-179.

Trégarot, E., Caillaud, A., Cornet, C.C., Taureau, F., Catry, T., Cragg, S.M. and Failler, P. (2021). Mangrove ecological services at the forefront of coastal change in the French overseas territories. Science of the Total Environment, 763

Tuya, F., Png-Gonzalez, L., Riera, R., Haroun, R. and Espino, F. (2014). Ecological structure and function differs between habitats dominated by seagrasses and green seaweeds. Marine environmental research, 98, pp.1-13.

Van Tussenbroek, B. I. (1993). Plant and frond dynamics of the giant kelp, Macrocystis pyrifera, forming a fringing zone in the Falkland Islands. European Journal of Phycology, 28(3), 161-165.

Vercelloni, J., Kayal, M., Chancerelle, Y. and Planes, S. (2019). Exposure, vulnerability, and resiliency of French Polynesian coral reefs to environmental disturbances. Scientific Reports, 9(1), p.1027.

Villéger, S., Grenouillet, G., and Brosse, S. (2013). Decomposing functional β-diversity reveals that low functional β-diversity is driven by low functional turnover in E uropean fish assemblages. Global Ecology and Biogeography, 22(6), 671-681.

Worthington, T.A., Zu Ermgassen, P.S., Friess, D.A., Krauss, K.W., Lovelock, C.E., Thorley, J., Tingey, R., Woodroffe, C.D., Bunting, P., Cormier, N. and Lagomasino, D. (2020). A global biophysical typology of mangroves and its relevance for ecosystem structure and deforestation. Scientific reports, 10(1), p.14652.

Annexe 1: Computation of functional biodiversity indices

The computation of functional space and functional biodiversity indices follows the General Workflow of the mFD package (Magneville *et al.* 2022), using R version 4.2.3 (R Core Team, 2023).

The use of the mFD package is based on three datasets:

- A dataframe summarising traits values for each species.
- A dataframe summarising the type of each trait (Nominal, Ordinal, Quantitative).
- A matrix summarising species assemblages: could be presence-absence, abundance or biomass data.

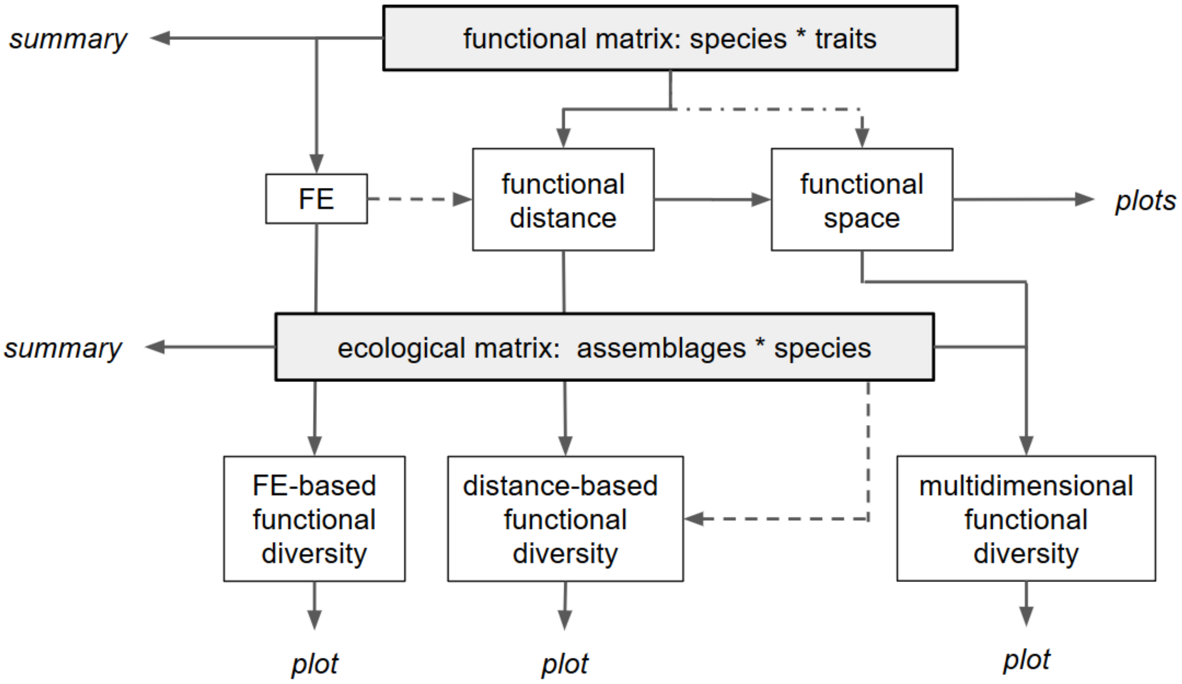


Figure 28. General workflow of the mFD package (Magneville *et al*. 2022)

The computation of functional diversity indices uses functional trait-based distance between species to build the functional space in which indices are computed. We use the Gower Distance metric to compute distances as we have non-continuous traits. In order to generate a multidimensional space in which diversity indices are computed, we perform a PCoA using the trait-based distances and evaluate the quality of PCoA-based multidimensional spaces according to the deviation between trait-based distances and distances in the functional spaces.

Once the dimensionality of the functional space is defined, the mFD package allows us to plot the given multidimentional space and the position of species in all 2-dimensions spaces made by pairs of axes.

Based on the position of each species on each assemblage, functional alpha diversity indices and functional beta diversity indices are computed.

Alpha diversity characterises each assemblage through multiple indices, such as:

- FDis **Functional Dispersion**: the biomass weighted deviation of species traits values from the center of the functional space filled by the assemblage *i.e.* the biomass-weighted mean distance to the biomass-weighted mean trait values of the assemblage.
- FRic **Functional Richness**: the proportion of functional space filled by species of the studied assemblage, *i.e.* the volume inside the convex-hull shaping species. To compute FRic the number of species must be at least higher than the number of functional axis + 1.
- FDiv **Functional Divergence**: the proportion of the biomass supported by the species with the most extreme functional traits *i.e.* the ones located close to the edge of the convex hull filled by the assemblage.
- FEve **Functional Evenness**: the regularity of biomass distribution in the functional space using the Minimum Spanning Tree linking all species present in the assemblage.
- FSpe **Functional Specialization**: the biomass weighted mean distance to the mean position of species from the global pool (present in all assemblages).
- FIde **Functional Identity**: the mean traits values for the assemblage. FIde is always computed when FDis is computed.
- FOri **Functional Originality**: the weighted mean distance to the nearest species from the global species pool.

mFD package allows us to compute beta diversity indices for each assemblage pairs following Villeger *et al.* (2013).

Beta-diversity is an interesting descriptor of diversity for assessing the effects of habitat and landscape change because it measures the difference in species composition between communities (Anderson *et al.* 2011). Beta-diversity is the result of two distinct processes — species replacement and species loss between communities — which can be expressed by two beta-diversity components, turnover and nestedness, respectively (Baselga, 2010, 2017). Thus, partitioning beta-diversity into these two components can identify whether changes in community composition due to anthropic disturbances result in species replacement (turnover) or species loss (nestedness) (Solar *et al.* 2015).

Annexe 2: Functional space quality and dimension

Reunion Island - reef fish assemblages functional space quality metrics

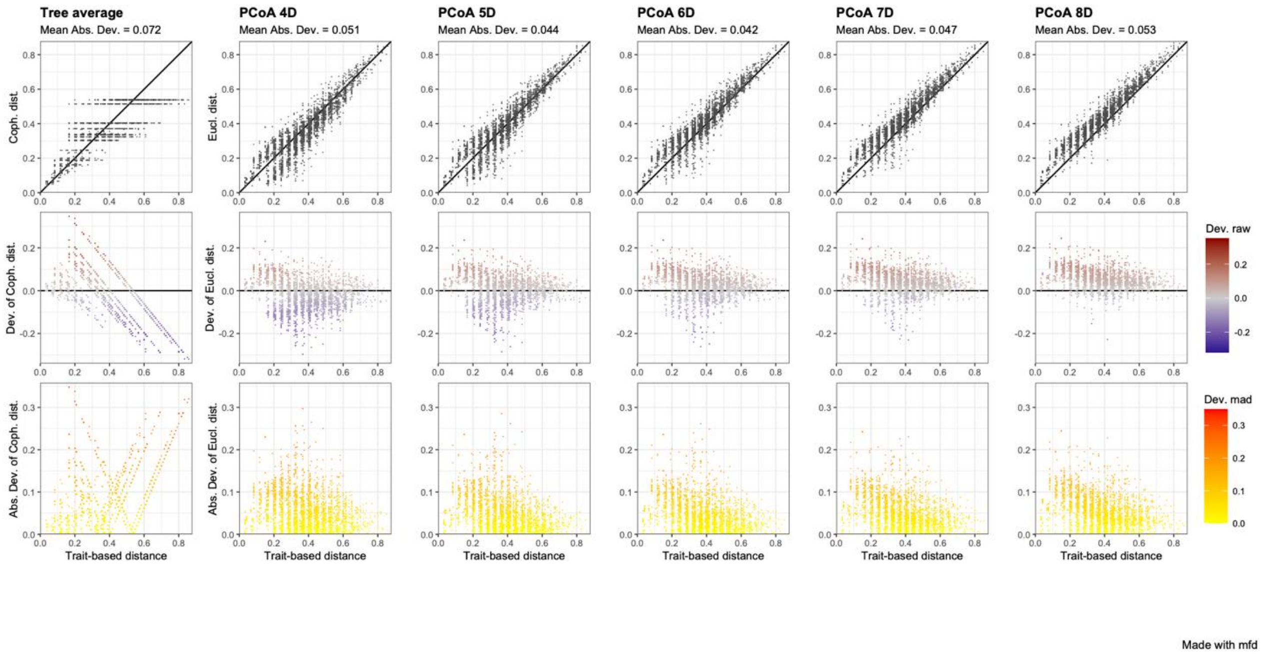


Figure 29. Reunion Island - reef fish assemblages quality metrics of functional spaces. Each column represents a functional space, the value of the quality metric is written on the top of each column.

Tikehau - reef fish assemblages functional space quality metrics

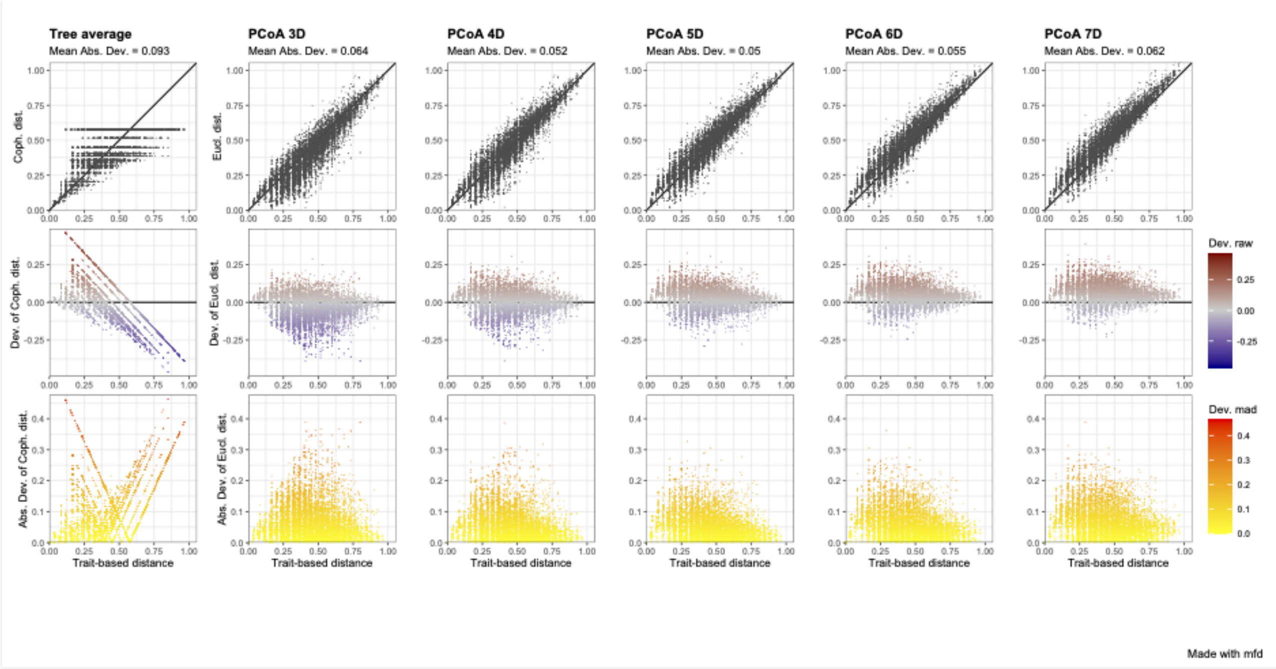


Figure 30. Tikehau - reef fish assemblages quality metrics of functional spaces. Each column represents a functional space, the value of the quality metric is written on the top of each column.

Canary Islands - Seagrass fish assemblages functional space quality metrics

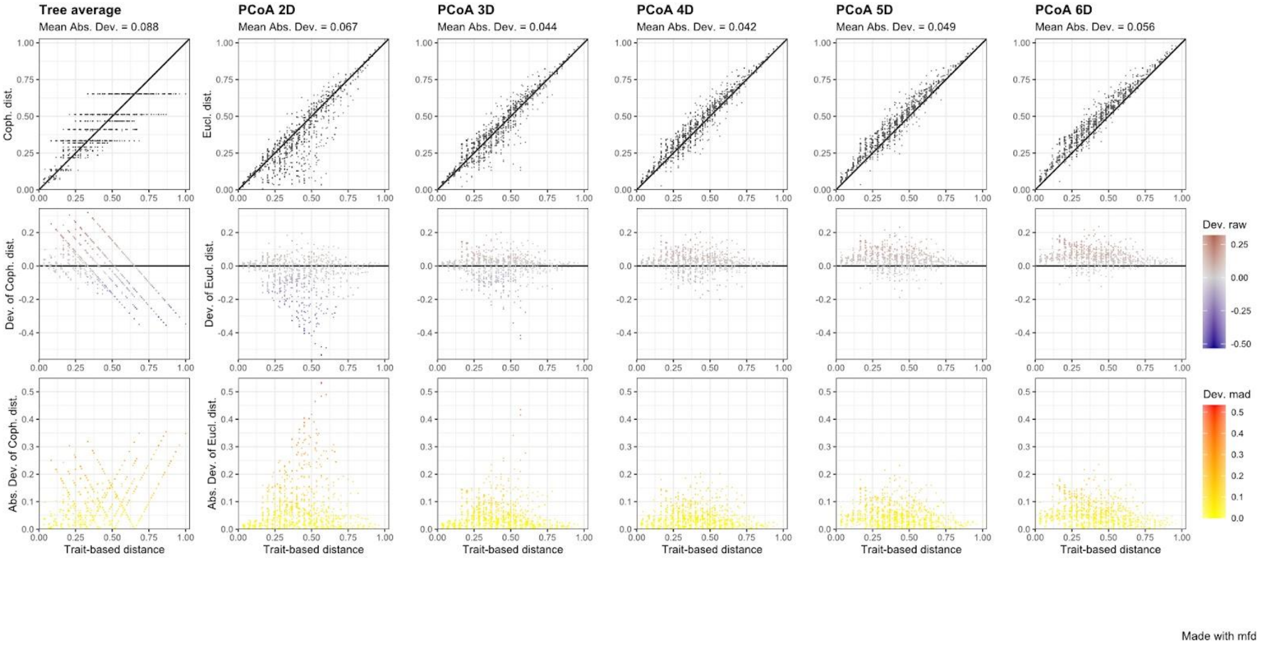


Figure 31. Canary Island - Seagrass fish assemblages quality metrics of functional spaces. Each column represents a functional space, the value of the quality metric is written on the top of each column.

Caribbean - Mangroves trees assemblages functional space quality metrics

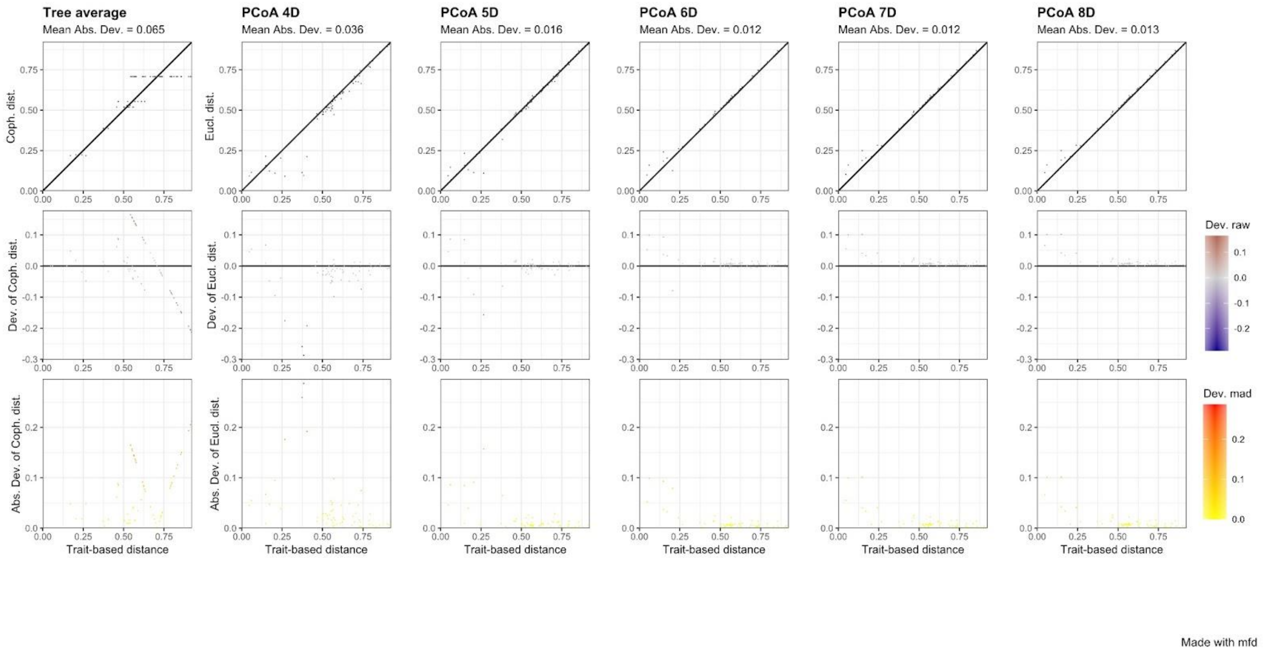


Figure 32. Caribbean - Mangrove trees assemblages quality metrics of functional spaces. Each column represents a functional space, the value of the quality metric is written on the top of each column.

Falkland Islands - kelp forests invertebrates assemblages functional space quality metrics

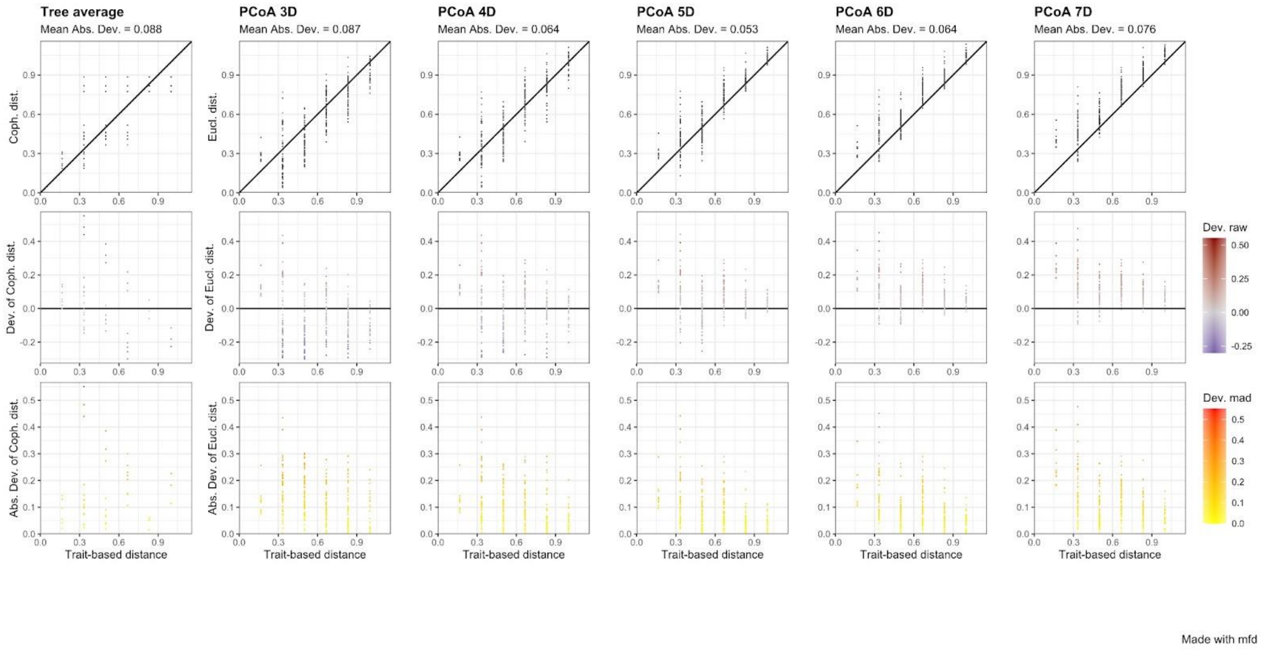


Figure 33. Falkland Islands - Kelp forests invertebrates assemblages quality metrics of functional spaces. Each column represents a functional space, the value of the quality metric is written on the top of each column.