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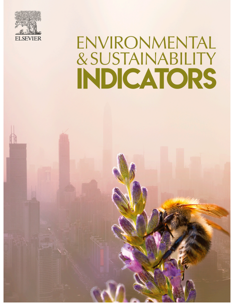

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ABSTRACT

Coral reefs are among the most diverse and productive ecosystems on the planet, providing goods and services to millions of people. Yet, they are threatened by a combination of natural and anthropogenic disturbances, which are increasing in both frequency and intensity. With 9% of the world's coral reef area, France is the 4th largest coral reef country, which confers a significant responsibility in terms of conservation. This synthesis evaluates the major coral reef monitoring programs in French overseas territories, aiming to identify gaps and proposing strategies to fulfill national (the French Coral Reef Initiative – IFRECOR, and the National Strategy for Protected Areas – SNAP) and European (the Water Framework Directive – WFD) requirements. We show that, despite a disparity in terms of objectives, implementation periods, spatial extent, and biological variables, these programs still represent a critical tool to inform stakeholders, managers and scientists about ecological changes. We emphasize that while these monitoring programs can be improved to meet the expectations of national public policy, such as through the use of common sampling strategies and multivariate indices that incorporate key ecological and functional processes, our analysis clearly demonstrates that several major requirements of the WFD are incompatible with coral reef monitoring programs. Establishing a reference point, determining alert thresholds, distinguishing natural and anthropogenic sources of disturbance, and the contrast between the allocated remediation time and the recovery cycles of benthic communities are fundamental obstacles to the application of the WFD for ecosystems as dynamic, diverse and complex as coral ecosystems.

Keywords: Coral reefs; Coral reef health; Ecological conditions; Monitoring; Indicators; IFRECOR; SNAP; WFD

1. Introduction

1.1. Scientific and political contexts

Coral reefs are among the most diverse, complex and productive ecosystems on Earth (Reaka-Kudla et al. 1996, Fisher et al. 2015, Williams et al. 2019). They offer coastal protection and a broad spectrum of economic, cultural, social, and aesthetic benefits to ~850 millions of people across more than 100 countries, predominantly within small-island states (Moberg & Folke 1999, Kittinger et al. 2012, Wodhead et al. 2019). Across an evolutionary time scale, coral reefs have experienced large-scale disturbances, such as thermal stress-induced bleaching events, cyclones and *Acanthaster* spp. outbreaks (De'ath et al. 2012, Hughes et al. 2017b, Adjeroud et al. 2018, Emslie et al. 2024). In recent decades, several anthropogenic impacts such as overfishing, pollution and sedimentation have added to the backdrop of global climatic stressors (Hoegh-Guldberg & Bruno 2010, Hoegh-Guldberg 2012, Reverter et al. 2024). Consequently, coral reef ecosystem functioning can be fundamentally altered, and with it the capacity of reefs to provide their crucial services to humanity (Brandl et al. 2024).

The recent evaluation by the Global Assessment of Biodiversity and Ecosystem Services (IPBES 2019) estimates that around half of the living coral surface of coral reefs has been lost over the last 150 years, and this decline has accelerated over the last three decades. Furthermore, more than 60% of coral reefs worldwide are under immediate and direct threat from local stressors (IPBES 2019, IPCC 2019), which increases to ~75% when local stressors are combined with thermal stress (Burke et al. 2011, Souter et al. 2021). Some projections suggest that mass mortalities of corals following bleaching events will become ever more frequent, with a decline of coral populations between 70% and 90% (Pandolfi et al. 2011, Mellin et al. 2024). In the context of this global 'coral reef crisis', evaluating the vulnerability, adaptability and resilience of reef communities and associated coastal human societies is urgently needed (Cinner et al. 2016, Hughes et al. 2017a, Mcleod et al. 2019).

In response, the scientific community alongside conservation managers and policymakers have initiated measures to protect coral reefs. These include the establishment of the Global Coral

Reef Monitoring Network (GCRMN) under the auspices of the International Coral Reef Initiative (ICRI), citizen-science initiatives such as Reef Check, and the ever-increasing establishment of Marine Protected Areas (MPAs). The primary goals of these initiatives are to produce comprehensive monitoring systems to evaluate the status of coral reefs, to track their trajectory after disturbances, and to implement management strategies that support resilience and the sustainable use of reef resources (Kleypas et al. 2001).

1.2. France: a significant responsibility

As the world's fourth-largest coral country, accounting for approximately 9% of the world's coral reef extent, and with coral reefs spread across the Indian, Pacific, and Atlantic Oceans (Fig. 1), France holds a significant responsibility in terms of conservation and management of coral reef ecosystems (Bambridge et al. 2019, Claudet et al. 2021, Pellerin et al. 2025).

In a decisive move in 1999, the French government initiated the French Coral Reef Initiative (IFRECOR), with the aim of promoting the conservation and sustainable management of coral reefs, seagrass beds and mangroves across French overseas territories (Fig. 2). This initiative underscores France's dedication to the ICRI and its GCRMN, a commitment that dates back to the inception of these programs in 1997. Tasked with evaluating coral reef health and resilience on a quinquennial basis, IFRECOR must also help ensure that 100% of reefs under effective protection by 2025. Furthermore in 2007, France embarked on a national strategy to establish and manage marine protected areas (MPAs), which was subsequently revised in 2012 to include French overseas territories. This strategy, known as the National Strategy for Protected Areas (hereafter SNAP), seeks to increase the conservation value of marine regions characterized by significant biodiversity and conservation needs, while also ensuring the sustainable development of associated human activities (Fig. 2).

At the international level, France actively implements the Water Framework Directive (WFD) of the European Union (adopted in 2000), which seeks to maintain or restore a good ecological and chemical status of groundwater and surface water bodies by 2027 (European Commission Directive 2000) (Fig. 2). Uniquely, France has voluntarily extended the scope of the WFD to include coastal waters surrounding its reefs, seagrass beds, and mangroves in certain overseas territories, distinguishing itself as the only European nation to take this step. The French coral reefs covered by the WFD include the outermost regions (OMR) under European jurisdiction, namely La Réunion, Mayotte, Guadeloupe, Martinique, and Saint Martin, while the overseas countries and territories (OCT) – French Polynesia, New Caledonia, Wallis and Futuna, the Scattered Islands, Clipperton, and Saint Barthélemy – do not fall under European jurisdiction and have their own political status with regard to environmental matters. It is important to note that other European initiatives, such as the Habitats Directive and the Marine Strategy Framework Directive, have been established to protect and conserve marine species and habitats. However, the WFD remains the most relevant for tropical benthic communities in OMR, and our review therefore focuses on this initiative.

In the context of the WFD, the bioindication tool must reflect the state of health of an environment, based on the characteristics of the living communities, as well as the causes responsible for the alteration of the environment (Hering et al. 2010, Voulvoulis et al. 2017). For coastal temperate and tropical marine habitats, the biological quality elements (BQE) relate to phytoplankton, macroalgae and angiosperms, benthic invertebrates and fishes. The status of these communities must be analyzed in terms of deviation from a reference state, which is expressed as the Ecological Quality Ratio (EQR): the state of an ecosystem compared to an equivalent environment that is free from anthropogenic pressures or subject to pressures of very low intensity (European Commission Directive 2000). The main stages in the development of a WFD bioindication tool are: (i) setting up a standardized sampling network (including an environmental impact gradient), (ii) defining a reference state, (iii) collecting biological data (which must include species occurrence, abundance and/or biomass), abiotic data and environmental impact data, (iv) examining the relationships between anthropogenic pressures and ecological status (to identify the metrics most sensitive to a given pressure), (v) aggregation of metrics (to finalize the indicators by constructing aggregation rules for the selected metrics, and to ensure that the resulting indicator also responds satisfactorily to impacts). Optional, but recommended, steps include an indicator validation phase, which must be carried out on a different database from the one used to create the indicator (European Commission Directive 2000).

The implementation of the WFD in certain ecosystems, such as inland waters, has led to significant progress in monitoring, conservation, and restoration. However, its implementation in other ecosystems, such as mangroves and coastal benthic communities, has highlighted a significant gap between the theoretical objectives of the WFD and the practical realities of monitoring these ecosystems (Moss 2008, Hering et al. 2010, Voulvoulis et al. 2017, Dirberg et al. 2020).

1.3. Objectives of this synthesis

Several years to decades following their inception, French stakeholders and managers have recognized the need to improve and reinforce existing coral reef monitoring networks. This includes increasing their comparability both within and among overseas territories. Furthermore, there is a call for the development of robust indicators that holistically report on the ecosystem's health status (Monnier et al. 2021), which includes the integration of data from benthic and fish communities to provide a better understanding of ecological dynamics and interactions. Other demands include the development of integrative indicators for each territory, tailored to their unique characteristics, and the quantification of impacts of both large-scale and localized disturbances, including bleaching events, cyclones, diseases outbreaks, and invasive alien species more effectively (Adjeroud et al. 2005). Indeed, French stakeholders and managers have demonstrated a keen interest in the critical evaluation of these coral reef monitoring endeavors, particularly concerning their efficacy and capacity to meet challenging national and international policy objectives. In this context, the present synthesis is designed to achieve three key goals: (i) to conduct a comprehensive evaluation of the current state of major coral reef health monitoring programs across French overseas territories, showcasing recent advancements in national reporting and alignment with public policy objectives;

(ii) to pinpoint deficiencies within these monitoring programs that hinder their ability to meet both national (e.g., IFRECOR, SNAP) and European (e.g., WFD) requirements; and (iii) to propose actionable strategies to refine and augment these monitoring and conservation efforts in the context of adaptive co-management. This is not only aimed at ensuring compliance with the aforementioned standards, but also at enhancing the uniformity and interoperability of monitoring systems to bolster and streamline national reporting efforts. These objectives underscore a strategic push towards refining coral reef conservation efforts, aligning them more closely with both existing and emergent environmental policy frameworks.

2. Status of coral reef monitoring in French overseas territories

2.1. Contrasting characteristics of monitoring programs

Coral reef monitoring programs have not been set up in a coordinated way across the French overseas territories and consequently differ in many aspects (Table 1). This diversity is evident in their varied origins, objectives, stakeholder engagements, and impacts on reef conservation and management strategies. Indeed, monitoring programs have either been established by research organizations in response to scientific questions, such as the Tiahura Radial at Moorea, French Polynesia (Galzin 1987), or by political decisions, such as the UNESCO World Heritage in New Caledonia and the GCRMN network in the SWIO and French Caribbean. Consequently, these programs not only pursue different goals but also engage with stakeholders and contribute to reef management in diverse ways. Beyond the different intentions, monitoring programs were also established at different time-periods within the disturbance-recovery cycles of the reefs, which has consequences for the assignment of reference state conditions to which subsequent data points can be compared as well as for the relevance of the analysis of temporal trajectories. The oldest monitoring programs, such as the French Polynesian Tiahura Radial in 1979 and Polynesia Mana in 1991 (Salvat et al. 2008), were established before the onset of the first reported mass coral mortalities associated with global coral bleaching events in 1998 (Skirving et al. 2019). In contrast, other programs, such as the one established for the WFD at La Réunion and Martinique, started in 2015 and 2017, respectively, after the occurrence of large-scale disturbances. This disparity in the length of time series, and in particular the limited historical perspective of some of them, poses challenges for analyzing trends in national reports and for distinguishing the impact of certain anthropogenic pressures, as required by the WFD.

The monitoring programs are carried out by either public research institutions, eco-guards from public sector, engineers from private consulting companies, or sometimes citizen scientists, resulting in a wide range of expertise levels. This variation significantly affects the choice of variables and the precision of taxonomic identifications among the monitoring programs (Table 1). There is particularly strong variability in the temporal and spatial coverage of monitoring efforts. Frequencies range from annual to occasional opportunistic data collection (especially on remote reefs such as the Iles Eparses), and there is considerable variability in the number of monitoring stations relative to the geographic and ecological diversity of the focal reefs. For example, the ~16,200 km^2 of coral reefs across 124 islands in French Polynesia and the relatively small ~18 km^2 of reefs in La Réunion are monitored using the same number of stations (Table 1). The selection of reef habitats for monitoring also varies, with some programs covering all major habitats to ensure a comprehensive evaluation of reef condition (i.e., Tiahura Radial at Moorea, World Heritage in New Caledonia), while others, often constrained by resource limitations (human, financial, or logistical), focus on specific habitats. For SNAP, the monitoring networks are far from optimal for assessing the effect of protected areas in most French overseas territories, as coverage often does not adequately cover habitats within and outside of marine reserves or exhibits temporal inconsistencies (such as in Saint-Martin and Saint-Barthélemy). Importantly, except for monitoring programs associated with localized impact studies, such as those on nickel mining in New Caledonia (Adjeroud et al. 2016), none of the existing monitoring networks in French overseas territories were designed to systematically address human-induced impacts through stratified sampling across gradients of anthropogenic disturbances.

The methods used across French monitoring programs to estimate the abundance, percent cover and diversity of targeted reef organisms vary considerably. Commonly used techniques include Line Intercept Transects (LIT), Point Intercept Transects (PIT) and photo-quadrat methods to estimate the diversity, percent cover, and abundance of benthic sessile organisms. Conversely, belt-transects are generally used for fish surveys. However, the size and the replication of these sampling units across each monitoring site is not consistent, which limits statistical approaches and the attainment of reliable generalizable results and interpretations.

Another source of variability among French monitoring programs concerns the ecological variables and indicators recorded and used to assess coral reef status. Overall, 57 biological variables/indicators related to reef communities have been selected in French monitoring programs, including data on abundance, biomass, substrate cover, diversity, and life forms, not only for major taxa such as scleractinian corals, fishes, algae, but also for sponges, soft corals, gorgonian, zoanthids, echinoderms, and mollusks (Fig. 3). Despite this large diversity of indicators, only 19 descriptors are consistently employed across all monitoring efforts, with foundational descriptors like coral and algal cover (inclusive of macroalgae, turf, and crustose coralline algae), along with fish herbivore biomass being systematically recorded in each region or site. The presence of diverse ecological variables across monitoring programs enables the development of comprehensive multivariate indicators for reefs and facilitates the exploration of reef community structure and dynamics, as exemplified by studies conducted in French Polynesia and La Réunion (Lamy et al. 2016, Adjeroud et al. 2018, Vercelloni et al. 2019, Jouval et al. 2023). However, the use of only a small fraction of indicators for reporting and analyses, raises the question of how useful or necessary some of the most granular variables are, since much information regarding the overall reef status can be gleaned from broader indicators (Brandl et al. 2024). Furthermore, some variables that capture important ecological processes and functions, are not – or only insufficiently – considered. For example, coral recruitment – which can be estimated through the abundance of juvenile coral colonies (< 5 cm in diameter) using transects or quadrats designed for adult colonies – is rarely monitored. This complicates the assessment of recovery and resilience capacities of coral reefs, which remains a central objective of initiatives such as IFRECOR.

The significant heterogeneity among French monitoring programs poses a real challenge for data interoperability, analyses and reporting at the national level, and decision-making to meet public policy requirements. Furthermore, the fact that none of these monitoring programs were designed specifically to assess anthropogenic pressures is a major obstacle to the WFD.

2.2. Initiatives to meet the national and international requirements

Following the creation of IFRECOR, France has devoted great effort to enhance the monitoring of coral reef health. This includes efforts to unify reporting standards at the national level, evidenced by progressive improvements in reports from the inaugural publication in 1999 (Gabrié et al. 1999) to the most recent in 2021 (IFRECOR 2021). In the latest report, data on reef status are systematically organized by regions, incorporating analyses of coral and macroalgal coverage, as well as herbivore biomass or abundance. Furthermore, this report methodically addresses the major stressors affecting these ecosystems. This standardized presentation markedly improves the synthesis of historical and current reef health data. Indeed, IFRECOR now provides a rigorous long-term annual dataset that significantly enhances the scale and strength of inference that can be made about coral reef health. Although further refinements are necessary, and despite the inherent heterogeneity of the monitoring efforts, these programs largely meet the expectations and objectives set by IFRECOR and SNAP.

In contrast, current monitoring efforts do not successfully meet the stringent targets set by the WFD. In La Réunion and Martinique, this shortfall primarily stems from a fundamental mismatch: the existing overseas monitoring programs were established prior to the implementation of the WFD and were not tailored to its strict criteria. Consequently, these monitoring programs lack strategically placed stations along gradients of anthropogenic pressure, and lack reference stations – both critical, for correlating pressures with ecological responses. This underscores the intrinsic difficulties in retrofitting existing monitoring systems to satisfy the precise demands of European environmental policy (Voulvoulis et al. 2017).

The implementation of the WFD on coral reefs in La Réunion has been a gradual process spanning several years. The French Regional Environment Directorate (DEAL), originally in charge of the WFD implementation for La Réunion, initiated various projects in 2000 to compile existing data and acquire new data, with the aim of establishing regular monitoring within the WFD's Monitoring and Control Network (RCS). Since 2008, the DEAL has relied on the French Research Institute for Exploitation of the Sea (IFREMER), which has taken on the role of project manager, creating and coordinating thematic working groups involving local experts. Between 2010 and 2012, the working group on coral reef benthic communities aimed to identify relevant ecological variables and indicators for assessing the state of water bodies. The current WFD monitoring at La Réunion, started in 2015, includes 7 stations located on the outer reef slope with a sampling frequency of three years. Seven GCRMN monitoring stations on the outer slope, monitored annually, are then added to the calculation of an integrated indicator of coral reef health, developed and tested during several years (Monnier et al. 2021). This indicator includes six variables: 1) cover of living coral colonies (relative to cover of hard substrate; CLC), 2) cover of *Acropora* spp. colonies (CA), 3) cover of branching and tabular *Acropora* spp. (CABT), 4) cover of macroalgae (CMA), 5) cover of crustose coralline algae (CCCA), and 6) cover of soft corals (ACSC). The final calculation of this indicator (I) includes varying weight given by expert opinion of the six variables, following the equation:

$$I = (10CLC+5CA+1CABT+2CMA+1CCCA+1CSC)/20$$

Although this index has greatly improved the characterization of reef benthic communities, notably by integrating several complementary variables on corals, algae and soft corals, it nevertheless remains far from WFD standards and therefore cannot be considered a WFD bioindicator. In addition to the subjectivity of the expert weighting, the main limitation of this indicator in responding to the WFD context is the absence of a consistent and rigorous pressure-impact relationship (Monnier et al. 2021). As discussed below (*3.4 Identification and discrimination* *of disturbance sources*), this limitation is mainly linked to the inherent difficulty of discriminating among the various natural and anthropogenic threats acting at various spatial and temporal scales and often in synergy (Adjeroud 1997, Reverter et al. 2024). This is further exacerbated by the choice of the outer oceanic slope as the habitat of reference, as it is less exposed to direct anthropogenic pressures than the reef flat directly adjacent to the coast (Cuet et al. 2023).

The implementation of the WFD in the Antilles was initiated in 2006 with a comprehensive assessment of the hydrographic district of Martinique, and delineation of Coastal Water Bodies. The monitoring program for the WFD 2016-2021 cycle in Martinique was initially approved by a prefectural order in December 2015, for effective implementation from January 2017. Initially, sites for monitoring were selected based on a literature review, existing monitoring efforts in Martinique (RNO, GCRMN/IFRECOR), and expert advice. A total of 16 sites in coastal water bodies and one site in a transitional water body were monitored annually for physical-chemical and biological variables. Four sites were added in 2017 to cover all water bodies, creating temporal disparities among stations. Due to limited knowledge of the marine environment in Martinique, these sites and their placements required adjustment over the years, based on *in situ* observations and the acquisition of new information through specific surveys. Simultaneously, efforts were made locally to develop 'WFD-compatible' methodologies tailored to the unique island context of the Antilles, focusing on: (i) the selection of biological and physical-chemical variables and monitoring protocols (in collaboration with Guadeloupe), (ii) the choice of indicators (data processing and aggregation methodology), and (iii) the development of quality grids and reference values used for the water bodies' status assessment. The evaluation of coral reef health is based on the aggregation of two indicators, the cover of living coral colonies, and the cover of macroalgae, both expressed as relative to cover of hard substrate (Monnier et al. 2021). As on La Réunion, this indicator for the Martinique reefs has not been validated to meet WFD requirements. One of the main limitations is the lack of long-term historical data that would provide a reference state for Martinique's coral reefs. Moreover, as in the case of La Réunion, a major concern is the clear identification of the environmental drivers of changes in reef communities (drivers-pressures-state-impacts-responses, DPSIR, relationships), a major element of WFD policy which has also posed issues for other ecosystems (Hering et al. 2010, Birk et al. 2012, Voulvoulis et al. 2017).

3. Major constraints for fulfilling national and European requirements

3.1. Comprehensive assessment with a unified integrative indicator

Despite recent efforts to standardize reporting at a national level, the diverse nature of monitoring efforts across French overseas territories presents significant challenges for consistency and data interoperability. This hampers the formulation of a comprehensive assessment, thereby diminishing the utility of monitoring data for shaping policy and guiding management strategies at the national level.

A major obstacle lies in the development of a unified index capable of encapsulating the current ecological health and potential for recovery and resilience of French coral reefs (Adjeroud et al. 2005, Chabanet et al. 2005). The complexity of ecological dynamics, including abrupt ecosystem transitions (phase-shifts) and changes in ecological trajectories (Brandl et al. 2024), complicates their integration into monitoring frameworks (Mellin et al. 2020, Cresswell et al. 2024, Reverter et al. 2024). These frameworks are typically annual and localized and are largely limited to the composition and abundance of a few taxonomic groups. The absence of comprehensive, integrative indices also limits the establishment of coherent public policies and decision-making processes at a national level (Monnier et al. 2021). Furthermore, it restricts the potential for these indicators to inform the public about environmental transformations or to enhance scientific research (Cinner & Barnes 2019, Castro-Cadenas et al. 2022, Santavy et al. 2022, Gudka et al. 2023).

3.2. The challenge of defining a reference state

A significant hurdle in adhering to WFD criteria lies in establishing a reference state for the benthic community (Hering et al. 2010, Josefsson & Baaner 2011, Bouleau et al. 2015, Voulvoulis et al. 2017). According to the WFD, this reference condition corresponds to the quality of water or composition of the benthos in an “undisturbed state”, which presents a challenge for many ecosystems including mangroves (Dirberg et al. 2020) and rivers (Dufour & Piégay 2009), but is even greater for frequently disturbed and highly dynamic ecosystems such as coral reefs, which are characterized by the occurrence of multiple disturbances, such as cyclones, marine heatwaves, and demographic outbreaks of predators.

Two approaches (temporal or spatial) are possible for determining the reference state. The spatial approach is often used when long-term monitoring series are not available. For a reef habitat, it consists in selecting the area with the best ecological conditions and the healthiest reef communities as a reference condition for other equivalent habitats (effectively a space-for-time substitution). For example, the remote inhabited island of Europa is often considered as a reference state for the Southwest Indian Ocean (Jouval et al. 2023). However, the strong heterogeneity of reef communities from local to regional scales, one of the general characteristics of coral reefs (Adjeroud et al. 2005, Reverter et al. 2024), strongly limits this approach when focusing on other reefs in the region.

The temporal approach consists in selecting the oldest values of a specific ecological variable recorded in a time series as the reference state, under the assumption that historical data represent largely undisturbed systems. However, most contemporary reefs show marked interannual variability in their biological communities as short-term responses to acute and chronic stressors, and decadal-scale cycles of decline and recovery (Lamy et al. 2016, Vercelloni et al. 2019, Cresswell et al. 2024) that are characterized by annual recruitment phases and slow growth rates (Adjeroud et al. 2017). In this context, rigorously defining a reference state is challenging, and to address this issue, the reference state has often been determined by expert opinion, typically based on the highest record value from the considered time series (Salvat et al. 2008). However, this approach, while occasionally beneficial, lacks the rigor necessary to meet WFD standards fully. In Moorea, French Polynesia, where interannual monitoring has been in place since the 1980s, the reference state of the outer reef slope habitat has been estimated as ~50% of living coral cover (Salvat et al. 2008, Lamy et al. 2016). Yet, many would argue that coral cover of 50% in the 1980s is not comparable with equivalent cover in the 2020s. Indeed, it has been widely documented that the disturbances of recent decades have significantly transformed coral communities, with species disappearing from reefs, such as *Acropora* spp., and increased dominance of species, such as *Porites* spp. and *Pocillopora* spp., adapted to recent changes (Berumen & Pratchett 2006, Adjeroud et al. 2018, McClanahan et al. 2020, Carlot et al. 2022, Reverter et al. 2024). This example underscores once again the complexities involved in identifying ecological variables and indicators to precisely define a reference state for coral reef ecosystems.

3.3. Challenges in establishing alert thresholds

Another major difficulty in meeting the requirements of the WFD is the establishment of a 'threshold' for state changes, which is also very useful for interpreting IFRECOR and SNAP monitoring results. As mentioned previously, coral reef ecosystems are characterized by strong spatio-temporal heterogeneity, which is governed by multiple interacting physical and biological processes that vary in intensity, frequency, and spatial scale (Mumby et al. 2013, Donovan et al. 2021, Brandl et al. 2024), as well as intrinsic factors inherent to the biology of coral reef foundation species. For instance, fluctuations in coral abundance and cover across years can be attributed to the natural cycles of reproduction, recruitment and post-settlement survival, which occur independently of external disturbances (Penin et al. 2010, Adjeroud et al. 2017, Smith et al. 2023). Therefore, it is often difficult to attribute a change in reef communities to a particular environmental stressor (Adjeroud et al. 2005, Guzman et al. 2020), which limits our capacity to define ecologically relevant thresholds that are critical for compliance with national and international policy frameworks.

When long time series are available and knowledge about disturbance regimes is detailed, thresholds can be estimated statistically using regular time-series analysis (Van Wynsberge et al.

2013). However, data to conduct such tests are rare and we often rely on expert opinion. In this last scenario, an arbitrary fluctuation of ±10% live coral cover between consecutive annual records is often used to categorize trends as stable, recovering or degrading (Adjeroud & Lasne 2022, Kayal et al. 2022). Although they do not comply with the strict framework of the WFD, these thresholds based on expert opinion may still prove useful in the context of less restrictive management and conservation strategies such as IFRECOR and SNAP. Moreover, distinguishing between 'natural' changes, which do not require management actions, and those resulting from environmental disturbances, which require targeted intervention to mitigate threats, poses a recurring challenge for the WFD's applicability to coral reef environments (Hering et al. 2010).

3.4. Identification and discrimination of disturbance sources

Studies over the last decades have identified some of the major extrinsic factors that control spatial patterns and temporal variability in coral assemblages, such as the availability of adequate substrate, sediment characteristics, light, water quality, and hydrodynamic forces (Williams et al. 2013, Robinson et al. 2018, McClanahan et al. 2020). However, the co-variation of these environmental factors with episodic large-scale disturbances produces important confounding effects that severely limit our capacity to detect and establish cause-and-effect relationships (Côté et al. 2016, Donovan et al. 2021, Walker et al. 2024). This complexity becomes particularly evident when attempting to pinpoint the causes behind reef degradation, as virtually all causes of large-scale disturbances that are considered "natural" are intertwined with effects of climate change (i.e., "cocktail of pressures"; McClanahan et al. 2020, Donovan et al. 2021). This issue is pivotal for environmental policies, including national initiatives like IFRECOR and SNAP, but holds even greater significance under the WFD. The WFD monitoring, predicated on the principle of identifying and mitigating human-linked disturbances, needs a clear distinction between natural and anthropogenic effects (Hering et al. 2010, Birk et al. 2012, Voulvoulis et al. 2017). Identifying clear cause-and-effect links between disturbance and ecosystem status appears feasible in less complex ecosystems like continental rivers (the primary focus during the development of the WFD). However, the natural variability of coral reefs and the intertwined nature of dominant stressors require decades of continuous monitoring to rigorously identify these relationships. Consequently, beginning with the issue of identifying specific drivers of change within coral reef indicators, the causal chain of the DPSIR model is virtually impossible to apply to coral reef ecosystems.

3.5. Discrepancy between temporal scales of remediation measures and dynamics of reef communities

Most public policies that deal with coral reef ecosystem monitoring aim to identify the causes of declines to halt them and restore a healthy ecological status. However, within both the IFRECOR and SNAP framework, this status is not clearly identified. Conversely, the WFD mandates the restoration of ecological conditions as a critical objective, imposing obligatory actions, with penalties prescribed for failure to achieve this restoration within a stipulated timeframe (European Commission Directive 2000). Specifically, the WFD demands the reinstatement of good ecological status within six years from the initiation of remedial actions, with some possible extension for particular actions. Given their extreme temporal variability, coral reefs do not align with a six-year recovery mandate. This discrepancy is further exacerbated by the challenges in differentiating between natural variation and human-induced alterations in reef conditions. Moreover, the recovery of many reef populations, most importantly corals – the primary architects of reef ecosystems – can span decades (Lamy et al. 2016, Adjeroud et al. 2018, Cresswell et al. 2024), further illustrating the mismatch between the prescribed temporal framework for ecological restoration and the inherent recovery timelines of coral reef communities (Hering et al. 2010, Voulvoulis et al. 2017).

4. Enhancing monitoring strategies for coral reef conservation

Monitoring programs in French overseas territories have made great strides in recent years to identify critical points for improving the quality and relevance of monitoring, as well as reporting on a national and international scale (Monnier et al. 2021). Despite these advancements, the implementation of such improvements often encounters resistance, stemming partly from reluctance to alter long-established monitoring systems, deficiencies in technical expertise for integrating novel ecological variables, apprehensions about compromising historical data integrity, and, most importantly, by logistical and financial constraints. However, the dynamic nature of environmental challenges necessitates the periodic revision of monitoring programs to elevate their effectiveness and alignment with evolving conservation goals (Lindenmayer & Likens 2009, Obura et al. 2019).

Properly executed enhancements of monitoring protocols, undertaken with meticulous planning and adherence to scientific rigor are designed to complement, not detract from, the historical value of monitoring data. Incorporating additional monitoring stations or adding biological variables to enable the calculation of relevant new indicators (as in New Caledonia, where the Coral Reef Monitoring Network – RORC – increased from 24 stations in 2009 to 101 in 2023, and juvenile corals were recorded since 2020), enriches the dataset rather than negating its historical significance. Considering the identified strengths and limitations of coral reef monitoring programs in French overseas territories, we advocate for specific advancements to bolster the relevance and impact of these efforts in shaping public policy. These recommendations aim to refine monitoring practices to capture and respond to the complexities of coral reef ecosystems effectively.

4.1. Reducing the variability in monitoring approaches

Achieving full uniformity in monitoring programs across disparate overseas territories is impractical, primarily due to the distinct ecological and environmental characteristics of reefs, such as those differentiating Caribbean reefs from those in the Indo-Pacific. However, it is necessary to achieve a minimum degree of consistency across monitoring programs. In this context, increasing the interoperability of sampling strategies, ecological variables, and indicators appears as a priority for analytical and reporting purposes.

In terms of sampling strategy, monitoring multiple reef habitats rather than just one habitat (as is sometimes performed for logistical and financial reasons) appears important. Indeed, habitats such as the outer reef slope and fringing reefs do not host the same reef communities are often subject to fundamentally different ecological communities, dynamics, and environmental drivers (Adjeroud 1997, Lamy et al. 2016, Brandl et al. 2025). Thus, a comprehensive and informative monitoring requires the inclusion of these main habitats to accurately reflect the temporal dynamic of reef communities (Chabanet et al. 2005, Mellin et al. 2020). In Guadeloupe, Saint-Barthelemy, Saint-Martin, and Wallis and Futuna, enhancing monitoring coverage in areas where it is currently sparse or irregular is essential to ensure that regional outcomes are not skewed by limited station sampling, based on habitat-specific baselines. Another important point concerns the frequency of sampling. It is essential that the monitoring stations are, at least, sampled annually, at the same time of year. Not only because many key ecological processes, such as reproduction and recruitment of corals, follow an annual cycle, but also because this allows for the evaluation of disturbance impacts over appropriate temporal scales (Kayal et al. 2012, Reverter et al. 2024).

Finally, a rapid convergence towards a common suite of biological variables and indicators across monitoring programs is necessary. For French overseas monitoring programs, it is essential to collect data on major components such as the main coral, fish, and algal categories, as for other coral reef monitoring programs worldwide (Obura et al. 2019). For corals, the basic data – colony abundance and/or coral cover – must be collected at the genus level and include information on the growth forms, as data on higher level, such as the commonly used “total” coral cover, does not capture frequently reported changes in the taxonomic composition of assemblages (Adjeroud et al. 2018, Brito-Millan et al. 2019). We also find it necessary to distinguish juvenile corals (immature colony of < 5 cm in diameter) within transects and quadrats used to sample adult colonies. The quantification of juvenile corals, which provides a more accessible measure of recruitment than early-stage recruits requiring artificial settlement plates, is particularly valuable. These juveniles offer insights into recent settlement patterns (several successive cohorts) and early post-settlement survival (growth and mortality), contributing to health and resilience indicators (Adjeroud et al. 2017, Ford et al. 2018, Edmunds & Riegl 2020, Jouval et al. 2023). For fishes, taxonomic identification at the genus level is also the minimal requirement, although the species level is preferable to track changes due to non-surveyed components or new species (e.g., invasive or introduced species). Distinguishing major trophic groups, and in particular corallivores and herbivores, is also necessary as they play a key role in the dynamics and recolonization process following disturbances (Fisher et al. 2015, Rice et al. 2019). In ecosystems where herbivory is largely performed by other organisms such as sea urchins, these organisms must also be included in monitoring. Moreover, estimation of fish body size is essential to convert simple abundance into biomass, which can be a more ecologically relevant variable. For algae, identification to species or even genus is very difficult, due to a lack of expertise, but it is essential to distinguish crustose coralline algae, turf, and macroalgae, as these categories have different ecological roles (Smith et al. 2020). In addition to these biological variables, which we hold critical to include in monitoring programs, other non-essential variables, such as coral and fish species richness or coral growth forms, could also serve as useful complements to monitoring efforts.

Beyond biological data, the inclusion of environmental variables can greatly enrich interpretations of reef community changes (Fichez et al. 2005, Obura et al. 2019). The advent of affordable sensors for continuous measurement of parameters like temperature, light, and salinity represents an opportunity to augment monitoring with critical environmental data (Obura et al. 2019). Additionally, leveraging advancements in photogrammetry to assess reef structural complexity could significantly enhance habitat characterization within coral reef health monitoring programs (Urbina-Barreto et al. 2021a, b).

Finally, the significance of biological descriptors within monitoring efforts is contingent upon their meticulous collection, emphasizing the necessity of replicating sampling units at each monitoring station to facilitate robust statistical analyses (Montilla et al. 2020). However, the number of replicates differs greatly among monitoring programs both within and between regions (Table 1), and this has a major impact on the power of the models when looking for significant differences in annual trends. In fact, only a rigorous methodology will enhance the reliability of monitoring results and enable meaningful conclusions to be drawn about reef health and resilience.

4.2. Contribution of modeling to define a reference state

Examining the temporal trajectory of ecological indices is useful for management and conservation. However, ecological indices are hard to interpret in absolute terms as it is extremely challenging to understand if a certain value correspond to a 'good' or 'poor' ecological status. Comparing values of ecological indices to reference conditions can bestow conceptually and quantitatively clear ecological meaning on these indices. Due to the limited temporal scope of available data, the baseline conditions from which changes can be measured can hardly be considered 'pristine'.

Using modeling approaches may overcome several limitations of the temporal and spatial approaches mentioned earlier and represent a promising avenue for scientific investigation. If all the relevant variables are considered, modeling potentially allows to define 'pristine' conditions (Brandl et al. 2024), estimating what a certain coral reefs might have looked like in the absence of anthropogenic disturbances. Indeed, many indicators of interest (e.g., coral cover) are determined by biogeographical history, local environmental conditions, the past regime of natural disturbance and the current anthropogenic stressors. If all these variables are accounted, it is possible to estimate the conditional effect of anthropogenic stressors and predict what the indicator of interest would be in absence of human impact (Fig. 4). Bayesian modelling may be particularly interesting for this, as it allows for the integration of previous knowledge through the definition of priors and the assessment of uncertainty around model estimates through the analysis of posterior probabilities (Brandl et al. 2024). This feature may allow to compute indices of ecosystem status as the distance from reference conditions associated with the standard deviation (Fig. 4). However, it is crucial to exercise caution when interpreting the outputs of such models. Variability in model inputs and assumptions can significantly influence the results, potentially leading to erroneous conclusions. For example, uncertainty may be extremely large due to limited data used to estimate the reference conditions. In these cases, such an approach would be unable to identify a significant deviation from a reference condition even when this actually exists. Given this risk of overestimating the models' results, we nevertheless find it important to view the models as decision-making tools rather than definitive benchmarks.

Coral reef research often faces challenges due to sparse data, particularly in regions where monitoring efforts are limited in space and time. The use of models and recent advances in machine learning offer promising solutions to address these data gaps. For example, coral reef sites sharing similar history and biogeography might be used to hindcast previous conditions for data-poor systems. By integrating additional data sources, such as demographic parameters (e.g., recruitment, survival, growth; Kayal et al. 2018), into existing knowledge of nearby reefs, it may be possible to enhance the accuracy of these reconstructions. The machine learning modelling typically involves two stages. In the first stage, the model is trained using observed values of a target variable, such as hard coral cover, along with a set of predictor values (e.g., sea surface temperature, chlorophyll a concentration) extracted from the same sites and years as the observations. In the second stage, the trained model is applied to a new set of predictor values to estimate the target variable at unsampled locations or during unmonitored time periods. A wide array of predictors, derived from satellite data are available at high spatial and temporal resolutions (at scales of a few hundred meters to a few kilometers) making them particularly suitable for this type of modeling (Gorelick et al. 2017). Through such multidisciplinary approaches, we can improve our understanding of historical coral reef conditions and strengthen the foundation for assessing contemporary changes. Nevertheless, the uncertainties inherent in the use of these modeling approaches must be clearly communicated to the users of these models notably to managers and policymakers.

4.3. Leveraging integrative indicators to capture multifaceted coral reef health dynamics

Several indicators of coral reef health have been proposed (Ben-Tzvi et al. 2004, Kaufman et al. 2011, Lasagna et al. 2014, Lirman et al. 2014, Rowland et al. 2020), while those assessing resilience capacities are more recent (Flower et al. 2017, Lam et al. 2017, Ford et al. 2018, Bachtiar et al. 2019, Thompson et al. 2020, Gudka et al. 2024, Broudic et al. 2025). Ideally, ecological indices should integrate key ecological and functional processes, such as diversity, abundance and biomass of reef corals, macroalgae, and herbivores, coral diseases and recruitment, as well as environmental and human stressors such as temperature, fishing, sedimentation, and pollution (McClanahan et al. 2012, Fujita et al. 2013, Lam et al. 2017, Lyu et al. 2024). Depending on the variables included in its calculation, such indices provide not only an estimate of current status, but also of resilience/recovery capacities when data on coral recruitment are included (Maynard et al. 2015, Ford et al. 2018, Jouval et al. 2023, Randrianarivo et al. 2024).

A recent approach to estimate the health and recovery potential of reef communities is the Recovery Index (RI) calculated using the Technique for Order Preference by Similarity to an Ideal Solution method (TOPSIS) approach (Parravicini et al. 2012, 2014, Jouval et al. 2023). This multi-criteria decision-matrix framework was developed for coral reefs of the Southwestern Indian Ocean (SWIO), with an RI including coral species richness and abundance, juvenile coral density, hard coral cover, proportion of stress-tolerant coral species cover, algal cover, herbivorous fish biomass, and sea surface temperature anomalies (Jouval et al. 2023, Randrianarivo et al. 2024). The RI has been tested in two contrasting situations in the SWIO. First, RI was calculated at local and regional scales along a gradient of anthropogenic development, from populated islands such as La Réunion and Mayotte to uninhabited ones such as Glorieuses and Europa in the Iles Eparses (Jouval et al. 2023). Second, in Madagascar, the TOPSIS approach was used to examine the effects of MPAs on the RI of reef communities (Randrianarivo et al. 2024).

TOPSIS consists of comparing alternatives (in our case, study sites) to the best and worst possible alternatives (IS for Ideal Solutions; Hwang & Yoon 1981). The technique allows for the ranking of each site according to a score that represents a synthesis of the variables used to describe the recovery capacity. The objective of TOPSIS is to classify alternative solutions according to their relative distance to “ideal” positive (IPS) and negative (INS) solutions. The basic principle is that the chosen alternative must have the shortest distance to the IPS, and thus the largest distance to the INS. In our case, it enabled ranking study sites according to their RI, from 0 to 1 for stations with the lowest to the highest recovery potential, respectively (Jouval et al. 2023).

Although limited in time, the results of two recent studies in the SWIO using RI with the TOPSIS method (Jouval et al. 2023, Randrianarivo et al. 2024) suggest that this indicator is suited to highlighting a gradient of anthropogenic pressure and the effect of conservation measures (MPAs), and could be a useful tool for reporting on coral reef health and resilience capacities, notably in a resilience-based management context (Maynard et al. 2015, Lam et al. 2017). Other indicators, such as the Coral Reef Rapid Assessment Method (Broudic et al. 2025), the Coral Condition Index (Lasagna et al. 2014) and the Deterioration Index (Ben-Tzvi et al. 2004), are also highly relevant for monitoring the ecological health of coral reefs. However, the use of multi-criteria analyses such as TOPSIS is particularly interesting because it is compatible with the idea of reference conditions. In the case of Parravicini et al. (2012, 2014) and Jouval et al. (2023), ideal solutions are estimated according to the ‘best’ and the ‘worst’ data available in the dataset. However, ideal solutions might be also defined by ecological models or expert opinion to improve the ecological relevance of the ranking exercise. The employment of TOPSIS and multi-criteria analysis in general responds to the many expectations of political decision-makers and stakeholders, which is the development of indices of reef status that integrate several biological and environmental components to better respond to the national policy objectives, and which may facilitate reporting at the national level. Moreover, this RI does not preclude a more granular approach, with details of all variables integrated in its calculation to analyze specific results (Fig. 5). Therefore, we strongly recommend its use in the French overseas territories, particularly for IFRECOR and SNAP. In the current state, such an index of reef health could be implemented in some monitoring of French overseas territories, but for other reefs, it would be appropriate to include other variables that are currently not considered. If a 'resilience' dimension is to be added to the monitoring program, which we strongly recommend given the increase in the frequency and intensity of disturbances that reefs undergo, data on coral recruitment (via the abundance of juvenile colonies) can be included (Fig. 3, Table 1). Other adaptations are also possible. For example, in the Caribbean where coral dominance is lower than in the Indo-Pacific and the substrate is largely colonized by other invertebrates, the RI could give greater weight to variations in sponge and gorgonian abundance. Moreover, on reefs where herbivory is dominated by sea urchins, their abundance should be integrated in the calculation of the RI. Thus, consistent implementation of the RI may offer strong value in terms of the quality and relevance of coral reef health monitoring in French overseas.

5. Concluding remarks and perspectives

5.1. The value of French overseas monitoring in the context of national environmental policies

Despite the limitations identified in this synthesis, monitoring efforts in the French overseas territories remain crucial for the surveillance of ecosystem conditions and the knowledge of stakeholders, policymakers, and scientists. The extensive network of monitoring sites, spanning three oceans, alongside the longstanding expertise underlying some of these initiatives, significantly bolsters France's capacity to address both national and international conservation challenges as the 4th largest coral reef nation.

With strategic political commitment and enhanced financial support, it is conceivable to address and potentially rectify many of the outlined deficiencies within these programs. Recent efforts to report and share data reactively, and to consolidate and harmonize reporting mechanisms across the national level serve as a testimony to the potential for these monitoring programs to more effectively align with the objectives of national public policies such as IFRECOR and SNAP (Monnier et al. 2021). Advancing towards a more integrated approach, which encourages collaboration across monitoring sites and synchronizes methodologies and indicators, promises increased efficiency, consistency, and improved knowledge exchange (Teixeira et al. 2016, Obura et al. 2019, Di Camillo et al. 2023). This effort, covering French reefs, must be carried out in collaboration with monitoring initiatives in other regions and countries, such as Reef Check and AGRRA (Atlantic and Guld Rapid Reef Assessment), to further the harmonization of existing sampling protocols, indicators and analyses. In this context, it is also crucial to intensify fundamental research based on these monitoring programs, which would enable more detailed analyses of the drivers of change or the involvement of organismal adaptation in temporal trajectories (Flower et al. 2017, Edmunds 2024).

5.2. Incompatibility with the WFD requirements

On the other hand, our analysis highlights the incompatibility of coral reef monitoring programs and the strict requirements of the WFD, a limitation also noted for other ecosystems (Moss 2008, Hering et al. 2010, Josefsson et al. 2011, Bouleau & Pont 2015, Voulvoulis et al. 2017). Certain aspects of the WFD, such as a more rigorous sampling strategy and the inclusion of more relevant biological variables and indicators, have certainly helped improve monitoring programs in French overseas regions. However, the unique attributes of coral reef ecosystems present significant obstacles, including the establishment of reference points ("undisturbed state"), the delineation of state thresholds, and the differentiation between natural and anthropogenic stressors, juxtaposed against the prescribed remediation timelines mandated by the WFD in the context of recovery cycles of reef communities. The basis of the WFD, rooted in the management of continental water bodies, reveals a fundamental incompatibility with the dynamic, complex, and biodiverse nature of coral reefs. Addressing these discrepancies necessitates more than minor adjustments to current monitoring sampling strategies or indicators; it calls for a reconsideration of the WFD's applicability to coral reef environments. The energy and the human and financial resources devoted to implementing the WFD in this inappropriate ecosystem are interfering with and risk compromising recent efforts to improve and harmonize monitoring protocols and analyses within the framework of IFRECOR and SNAP.

Among the possible alternatives, indicators based on coral reef benthic communities could be used as a complementary status indicator to the other biological quality elements (BQE), without downgrading or direct implications in terms of remediation measures, as previously suggested (Le Moal et al. 2016). The development of specific policy adjustments or the creation of legal frameworks tailored to coral reefs could have a significant impact on policymaking. However, the fact that France is the only European country to have adopted the WFD for coral reefs, which makes the exercise even more difficult due to the absence of equivalent initiatives in the EU, also raises the question of its relevance on an international scale. These considerations extend deeply into the realm of policy and legal reform, which are beyond the scope and objectives of the present synthesis.

5.3. Embracing new technologies for future monitoring efforts

Amidst escalating environmental disturbances, the advent of cutting-edge technologies offers promising avenues for coral reef monitoring practices (Obura et al. 2019, Apprill et al. 2023, Cardenas et al. 2024, Sultan et al. 2025). Recent advancements in *in situ* and satellite-based remote sensing measurements (Muller-Karger et al. 2018, Teague et al. 2022, Mills et al. 2024), ecological acoustics (Elise et al. 2019), eDNA analysis (Hassan et al. 2024, Shen et al. 2024), reef water microorganisms (Apprill & Salerno 2025), photogrammetry (Carlot et al. 2020, Urbina-Barreto 2021a, b) and artificial intelligence (Beijbom et al. 2015, Gonzalez-Rivero et al. 2022, Ouassine et al. 2025) herald a new era for ecological monitoring and research. Moreover, the ecological importance and extent of mesophotic coral ecosystems (i.e., light-dependent corals and associated communities found at depths ranging from 30 to 150 m), which have recently been revealed, also argue for their inclusion in monitoring networks and conservation programs (Rocha et al. 2018, Hoarau et al. 2024). It would also be relevant to include a temporal analysis of land-use changes, as the decline of reefs may also reflect coastal pressures and transformations. Integrating these new technologies into monitoring programs will require financial investment and the development of advanced expertise, and will have implications for overall governance. As these technological innovations continue to develop and mature, there is a need for the establishment of interdisciplinary working groups dedicated to integrating these tools into existing monitoring frameworks for better conservation and management policy (Cardenas et al. 2024). In this context, the integration of socio-economic aspects into monitoring programs also needs to be developed, drawing in particular on citizen science and local knowledge (Obura et al. 2019). This effort will make it possible to develop indicators that better integrate not only the ecological characteristics and functions of reef ecosystems, but also the ecosystem services rendered to human populations (Maynard et al. 2010, Obura et al. 2019, Castro-Cadenas et al. 2022). Such proactive measures will not only enhance our understanding of the coral reef socio-ecological systems, but also equip conservation efforts to navigate the impending challenges of the twenty-first century, ensuring that monitoring practices remain robust, responsive, and relevant in the face of changing environmental conditions.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Data availability

The data underlying the results presented in the study are available from the corresponding author.

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Table 1. Major characteristics of coral reef health monitoring in French overseas territories.

Region[1]	Island	Name of the monitoring	Environmental policies	No. of stations	Habitats[3]	Starting date	Sampling frequency	Sampling method for benthos	Benthic categories	Sampling method for fishes	Fish categories	Sampling method for invertebrates	Invertebrate categories	Data collectors
SWIO	La Réunion	GCRMN	IFRECOR	16	FR, OS	1998	Regular/interannual	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Eco-guards, researchers, private consultants
SWIO	La Réunion	DCE	WFD	7	OS	2015	Regular/every 3 years	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species	3 belt transects (20×4 m)	Sea urchins	Private consultants
SWIO	La Réunion	MPA effect	SNAP	40	FR, OS	2007	Regular/every 5 years	40 photo-quadrats (0.70×0.35 m)	Genera/Species	3 belt transects (50×5 m)	Species			Eco-guards, researchers
SWIO	Mayotte	GCRMN	IFRECOR	21	FR, LR, OS	1999	Regular/interannual	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Private consultants, eco-guards
SWIO	Mayotte	MPA effect	SNAP	10	BR	1995	Occasional			3 belt transects (50×5 m)	Species			Private consultants
SWIO	Mayotte	Fringing Reef	IFRECOR	200	FR	1989	Occasional	1 MSA (20 m) and 10 photo-quadrats (2×2 m)	Targeted categories[4]					Private consultants
SWIO	Mayotte	MSA	IFRECOR	60	BR, LR	2005	Occasional	1 MSA (25 m) and 10 photo-quadrats (5×5 m)	Targeted categories	1 belt transect (25×5 m)	Herbivores and emblematic species			Private consultants
SWIO	Europa	GCRMN	IFRECOR	6	FR, OS	2011	Occasional	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Researchers
SWIO	Glorieuses	GCRMN	IFRECOR	6	FR, OS	2002	Occasional	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Researchers
SWIO	Bassas Da India	GCRMN	IFRECOR	4	OS, LR	2011	Occasional	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Researchers
SWIO	Juan de Nova	GCRMN	IFRECOR	7	FR, OS	2008	Occasional	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Researchers
SWIO	Tromelin	GCRMN	IFRECOR	3	OS	2011	Occasional	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Researchers
SWIO	Geyser/Zelée	GCRMN	IFRECOR	14	BR, OS	1996	Regular/interannual for some stations, occasional for others	3 LIT of 20 m	Genera/Species	3 belt transects (50×5 m)	Species			Eco-guards, researchers, private consultants
CP	Moorea	MPA monitoring	SNAP	42	FR, LR, OS	2004	Regular/semi-annual	3 PIT of 25 m with points every 0.5 m	Targeted categories	3 belt transects (25×2 m)	Species	3 belt transects (25×2 m)	Targeted categories	Researchers
CP	Moorea	Tiahura radial	IFRECOR	3	FR, LR, OS	1979	Regular/interannual	1 PIT of 50 m with points every 1 m, 1 PIT of 25 m with points every 0.25 m	Genera	4 belt transects (25×2 m)	Species			Researchers
CP	14 islands[2]	Polynesia Mana	IFRECOR	21	OS	1991	Regular/biannual since 1997, occasional before	20 photo-quadrats (1×1 m)	Genera	3 belt transects (50×5 m)	Species			Researchers
CP	Wallis and Futuna	Wallis & Futuna monitoring	IFRECOR	12	FR, LR, OS	2019	Occasional	4 PIT of 20 m with points every 0.5 m	Targeted categories	4 belt transects (20×5 m)	Targeted categories	4 belt transects (20×5 m)	Targeted categories	Eco-guards
CP	Wallis and Futuna	Polynesia Mana	IFRECOR	6	OS	1999	Regular/every 3 years	20 photo-quadrats (1×1 m)	Genera				Species level	Eco-guards
WP	New Caledonia	RORC	IFRECOR	101	FR, LR, BR	1997	Regular/interannual since 2003, occasional before	4 PIT of 20 m with points every 0.5 m	Targeted categories	4 belt transects (20×5 m)	Targeted categories	4 belt transects (20×5 m)	Targeted categories	Participatory science
WP	New Caledonia	World Heritage	IFRECOR	235	FR, LR, BR, OS	2006	Regular/every 6-10 years	1 LIT of 50 m	Targeted categories	Distance sampling (1×50 m)	Targeted categories	1 belt transect (50×5 m)	Species level	Private consultants, researchers

Table 1. Continued.

Region	**Island**	**Name of the monitoring**	**Environmental policies**	**No. of stations**	**Habitats**	**Starting date**	**Sampling frequency**	**Sampling method for benthos**	**Benthic categories**	**Sampling method for fishes**	**Fish categories**	**Sampling method for invertebrates**	**Invertebrate categories**	**Data collectors**
CA	Martinique	GCRMN	IFRECOR	5	FR, BCR	2001	Regular/interannual until 2013, occasional after	3×50 m belt transects / 50 photo-quadrats (1×1 m) Coral juveniles: 0.25×0.25 cm quadrats (10 per transect)	Genera/Species	3 belt transects (50×4 m for mobile species, 50×2 m for others (i.e., territorial)	Species	3 belt transects (50×1 m)	Diadematidae sea urchins	Private consultants
CA	Martinique	DCE	WFD	15	FR, BCR	2007	Regular/interannual	6 PIT of 10 m with point every 0.2 m, 60 quadrats (0.25×0.25 m)	Genera/Species			60 quadrats (1×1 m)	Sea urchins	Private consultants
CA	Martinique	Reef Check	IFRECOR	4	FR, BCR	2009	Occasional	4 PIT of 20 m with points every 0.5 m	Targeted categories	4 belt transects (20×5 m)	Targeted categories	4 belt transects (20×5 m)	Targeted categories	Participatory science
CA	Guadeloupe	GCRMN	IFRECOR	4	FR	2002	Regular/interannual	6 LIT 10 m transects along a 60 m transect, 6 belt transects 10×0.5 m for coral juveniles	Genera/Species	10 belt transects (30×2 m) along a 150 m transect	Targeted categories	6 belt transects (10×1 m)	Sea urchins	Researchers, private consultants
CA	Guadeloupe	MPA network	SNAP	2	FR	2017	Occasional	6 PIT of 10 m with point every 0.2 m or 3 PIT of 20 m with point every 0.5 m, 60 quadrats (1.0×0.5 m) for juvenile corals	Genera/Species	3 belt transects (50×2 m)	Targeted categories	60 quadrats (1×1 m)	Sea urchins	Eco-guards, private consultants
CA	Guadeloupe	DCE	WFD	15	FR, BCR	2008	Regular/interannual	6 PIT of 10 m with point every 0.2 m, 60 quadrats (0.25×0.25 m), 6×10 m linear transects +and 1×0.5 m quadrats for coral juveniles	Genera/Species			60 quadrats (1×1 m)	Sea urchins	Private consultants
CA	Guadeloupe	Reef Check	IFRECOR	8	BCR	2007	Interannual	4 PIT (20 m), with points every 0.5 m	Targeted categories	4 belt transects (20×5 m)	Targeted categories	4 belt transects (20×5 m)	Targeted categories	Participatory science
CA	Saint-Barthelemy	GCRMN	IFRECOR	2	BCR	2002	Bi-annual until 2006, interannual from 2007	6 LIT 10 m transects along a 60 m transect, 6 belt transects 10×0.5 m for coral juveniles	Genera/Species	10 belt transects (30×2 m) along a 150 m transect	Targeted categories	6 belt transects (10×1 m)	Sea urchins	Eco-guards, researchers
CA	Saint-Barthelemy	MPA network	SNAP	2	BCR	2007	Interannual	6 PIT of 10 m with point every 0.2 m or 3 PIT of 20 m with point every 0.5 m, 60 quadrats (1.0×0.5 m) for juvenile corals	Genera/Species	3 belt transects (50×2 m)	Targeted categories	60 quadrats (1×1 m)	Sea urchins	Eco-guards, private consultants
CA	Saint-Barthelemy	Reef Check	IFRECOR	2	BCR	2018	Interannual	4 PIT (20 m), with points every 0.5 m	Targeted categories	4 belt transects (20×5 m)	Targeted categories	4 belt transects (20×5 m)	Targeted categories	Participatory science

CA	Petite Terre	MPA network	SNAP	2	BCR	2007	Interannual	6 PIT of 10 m with point every 0.2 m or 3 PIT of 20 m with point every 0.5 m, 60 quadrats (1.0×0.5 m) for juvenile corals	Genera/Species	3 belt transects (50×2 m)	Targeted categories	60 quadrats (1×1 m)	Sea urchins	Eco-guards, private consultants
CA	Saint-Martin	MPA network	SNAP	8	BCR	2007	Interannual	6 PIT of 10 m with point every 0.2 m or 3 PIT of 20 m with point every 0.5 m, 60 quadrats (1.0×0.5 m) for juvenile corals	Genera/Species	3 belt transects (50×2 m)	Targeted categories	60 quadrats (1×1 m)	Sea urchins	Eco-guards, private consultants
CA	Saint-Martin	Reef Check	IFRECOR	4	BCR	2007	Interannual	4 PIT (20 m), with points every 0.5 m	Targeted categories	4 belt transects (20×5 m)	Targeted categories	4 belt transects (20×5 m)	Targeted categories	Eco-guards, participatory science
CA	Saint-Martin	DCE	WFD	1	BCR	2018	Regular/interannual	6 PIT of 10 m with point every 0.2 m, 60 quadrats (0.25×0.25 m)	Genera/Species			60 quadrats (1×1 m)	Sea urchins	Private consultants

1: SWIO: South-Western Indian Ocean; CP: Central Pacific; WP: West Pacific; CA: Caribbean; 2: Moorea, Tahiti, Bora Bora, Tetiaora, Aratika, Marutea sud, Mataiva, Nengo-Nengo, Rangiroa, Takapoto, Tikehau, Ua Uka, Nuku Hiva, Tubuai ; 3: OS: FR: fringing reef; LR: lagoonal/mid-shelf reefs; BR: barrier reef; OS: outer reef slope; BCR: volcanic rocky bottom; 4: These categories include various taxa that vary by region and monitoring program

Figure captions

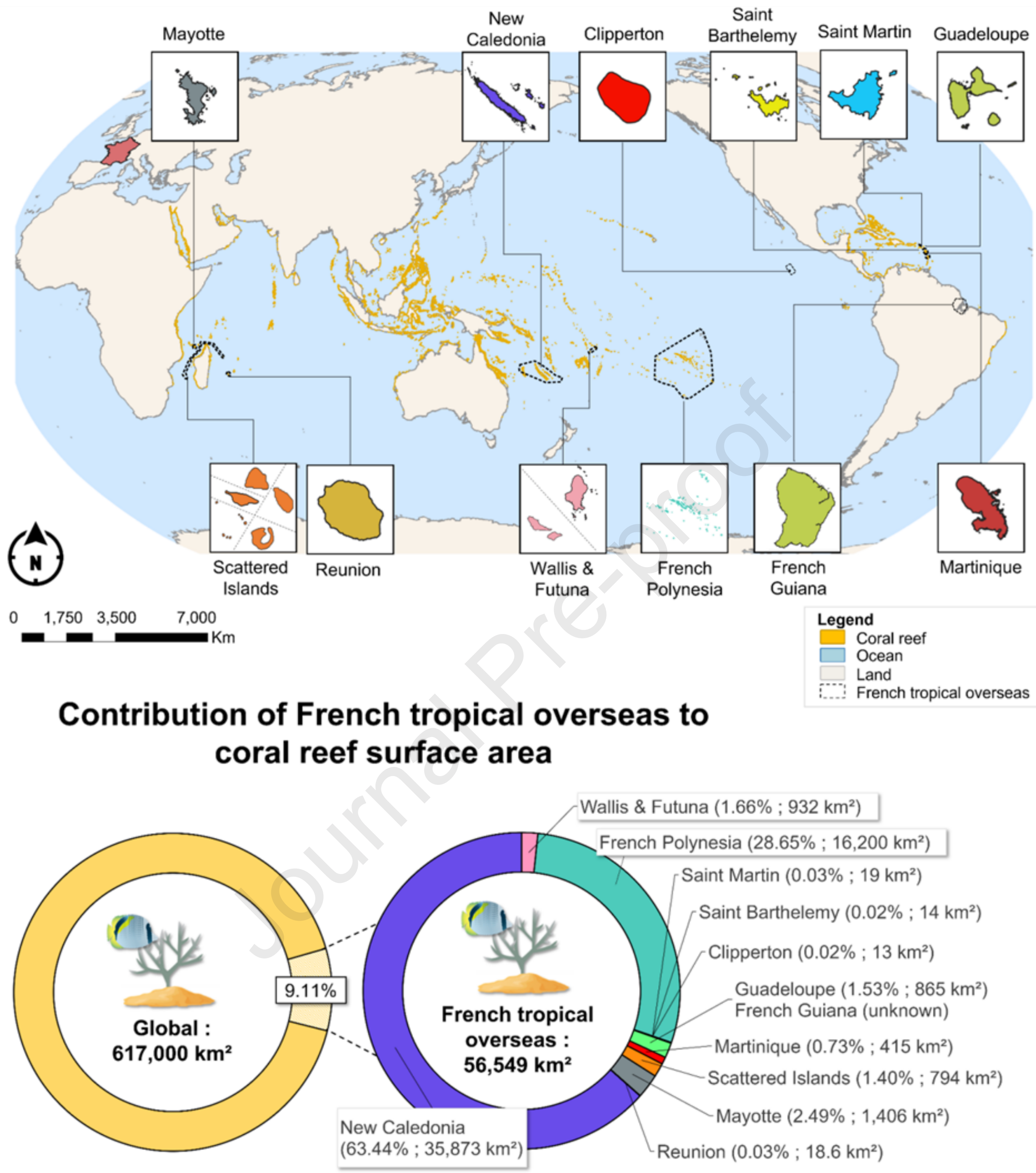
Fig 1. Contribution of French coral reefs to coral reef area at a global scale, and their distribution among overseas territories. From Pellerin et al. (2025).

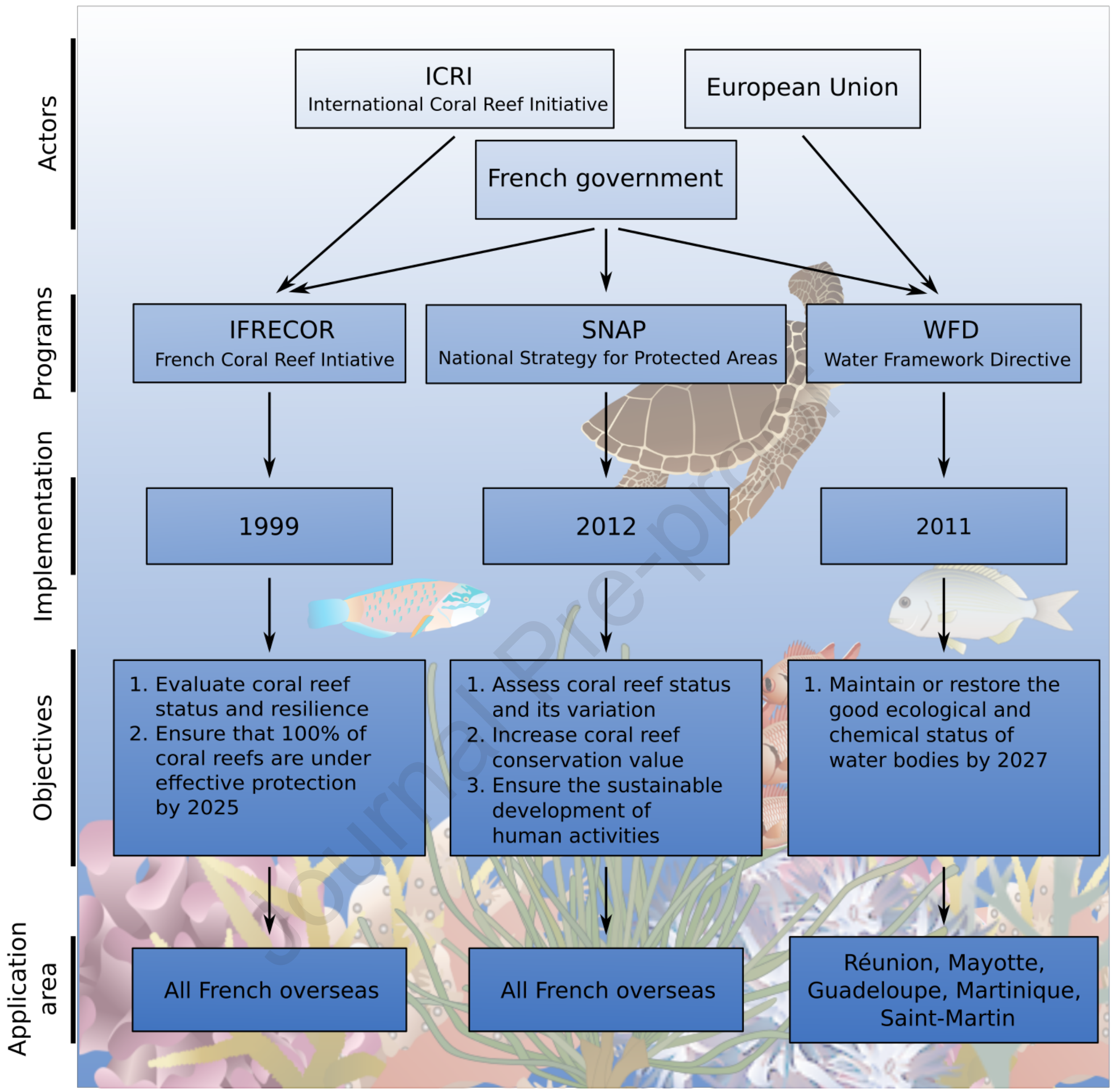
Fig. 2. Schematic representation of the objectives, protagonists and targets of the main national (IFRECOR, SNAP) and European (WFD) environmental policies.

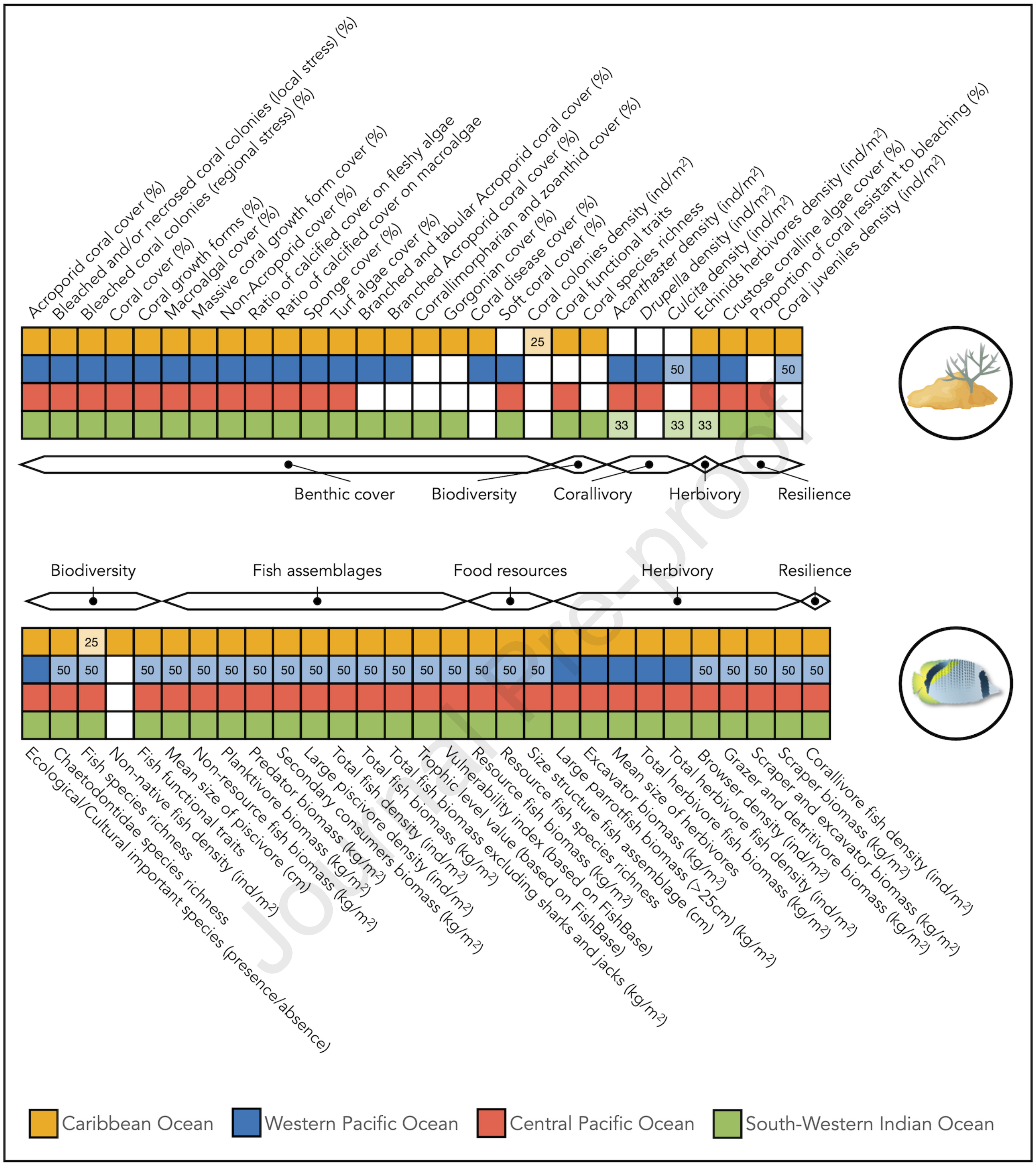
Fig. 3. Ecological variables and indicators used in French overseas reef monitoring across major regions. Caribbean Ocean: Guadeloupe, Martinique, Saint-Martin, Saint-Barthelemy; West Pacific Ocean: New Caledonia, Wallis and Futuna; Central Pacific Ocean: French Polynesia; Southwest Indian Ocean: La Réunion, Mayotte and Iles Eparses).

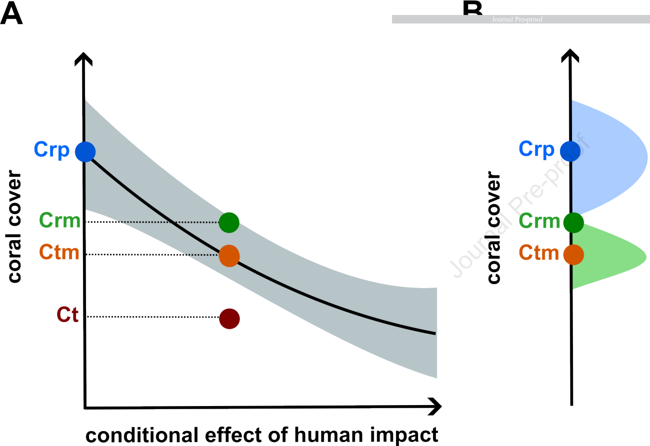
Fig. 4. Concept of a modelling approach to define coral reef status based on unbiased reference conditions. The approach is illustrated using coral cover as the response variable for reef status and a generic variable for human impact (A). *Ct* is the target reef to be evaluated. The intercept of the coral cover–impact relationship (*Crp*) represents an idealized "pristine" condition (i.e., the modelled coral cover in the absence of human pressure). *Crm* represents the maximum attainable coral cover under a level of human impact equivalent to that of *Ct*, while *Ctm* is the modelled coral cover under that same impact. A Bayesian framework is used to estimate posterior distributions and thus the uncertainty of each point (B). Based on this approach, we propose two ecological indices (C) that quantify the deviation of *Ct* from (i) pristine conditions and (ii) the maximum attainable coral cover under the prevailing level of human impact, when the latter cannot be mitigated by conservation measures.

Fig. 5. Illustration of the use of the Recovery Index (RI) calculated with the TOPSIS approach to examine the effects of Marine Protected Areas (MPAs) on reef communities. Three regions (Masoala in the northeast, Nosy-Be in the northwest, and Salary Nord in the southwest) and 18 sampling stations were located around Madagascar. The normalized values of the eight recovery metrics at each station are presented in pie radar plots and the RIs are in bold in middle of each plot. Stations coded NTZ ("No Take Zone") are located in unfished areas. From Randrianarivo et al. (2024).


Mayotte
New Caledonia
Clipperton
Saint Barthelemy
Saint Martin
Guadeloupe
Scattered Islands
Reunion
Wallis & Futuna
French Polynesia
French Guiana
Martinique
N
0 1,750 3,500 7,000 Km
Legend
Coral reef
Ocean
Land
French tropical overseas
Contribution of French tropical overseas to coral reef surface area
Global : 617,000 km²
9.11%
French tropical overseas : 56,549 km²
Wallis & Futuna (1.66% ; 932 km²)
French Polynesia (28.65% ; 16,200 km²)
Saint Martin (0.03% ; 19 km²)
Saint Barthelemy (0.02% ; 14 km²)
Clipperton (0.02% ; 13 km²)
Guadeloupe (1.53% ; 865 km²)
French Guiana (unknown)
Martinique (0.73% ; 415 km²)
Scattered Islands (1.40% ; 794 km²)
Mayotte (2.49% ; 1,406 km²)
Reunion (0.03% ; 18.6 km²)
New Caledonia (63.44% ; 35,873 km²)


Actors
Programs
Implementation
Objectives
Application area
ICRI
International Coral Reef Initiative
European Union
French government
IFRECOR
French Coral Reef Intiative
SNAP
National Strategy for Protected Areas
WFD
Water Framework Directive
1999
2012
2011
1. Evaluate coral reef status and resilience
2. Ensure that 100% of coral reefs are under effective protection by 2025
1. Assess coral reef status and its variation
2. Increase coral reef conservation value
3. Ensure the sustainable development of human activities
1. Maintain or restore the good ecological and chemical status of water bodies by 2027
All French overseas
All French overseas
Réunion, Mayotte, Guadeloupe, Martinique, Saint-Martin


Acroporid coral cover (%)
Bleached and/or necrosed coral colonies (local stress) (%)
Bleached coral colonies (regional stress) (%)
Coral cover (%)
Coral growth forms (%)
Macroalgal cover (%)
Massive coral growth form cover (%)
Non-Acroporid coral cover (%)
Ratio of calcified cover on fleshy algae
Ratio of calcified cover on macroalgae
Sponge cover (%)
Turf algae cover (%)
Branched and tabular Acroporid coral cover (%)
Branched Acroporid coral cover (%)
Corallimorpharian and zoanthid cover (%)
Gorgonian cover (%)
Coral disease cover (%)
Soft coral cover (%)
Coral colonies density (ind/m²)
Coral functional traits
Coral species richness
Acanthaster density (ind/m²)
Drupella density (ind/m²)
Culcita density (ind/m²)
Echinids herbivores density (ind/m²)
Crustose coralline algae cover (%)
Proportion of coral resistant to bleaching (%)
Coral juveniles density (ind/m²)
25
50
50
33
33
33
Benthic cover
Biodiversity
Corallivory
Herbivory
Resilience
Biodiversity
Fish assemblages
Food resources
Herbivory
Resilience
25
50 50 50 50 50 50 50 50 50 50 50 50 50 50 50 50 50
50 50 50 50 50
Ecological/Cultural important species (presence/absence)
Chaetodontidae species richness
Fish species richness
Non-native fish density (ind/m²)
Fish functional traits
Mean size of piscivore (cm)
Non-resource fish biomass (kg/m²)
Planktivore biomass (kg/m²)
Predator biomass (kg/m²)
Secondary consumers biomass (kg/m²)
Large piscivore density (ind/m²)
Total fish density (ind/m²)
Total fish biomass (kg/m²)
Total fish biomass excluding sharks and jacks (kg/m²)
Trophic level value (based on FishBase)
Vulnerability index (based on FishBase)
Resource fish biomass (kg/m²)
Resource fish species richness
Size structure fish assemblage (cm)
Large parrotfish biomass (>25cm) (kg/m²)
Excavator biomass (kg/m²)
Mean size of herbivores
Total herbivore fish biomass (kg/m²)
Total herbivore fish density (ind/m²)
Browser density (ind/m²)
Grazer and detritivore biomass (kg/m²)
Scraper and excavator biomass (kg/m²)
Scraper biomass (kg/m²)
Corallivore fish density (ind/m²)
Caribbean Ocean
Western Pacific Ocean
Central Pacific Ocean
South-Western Indian Ocean



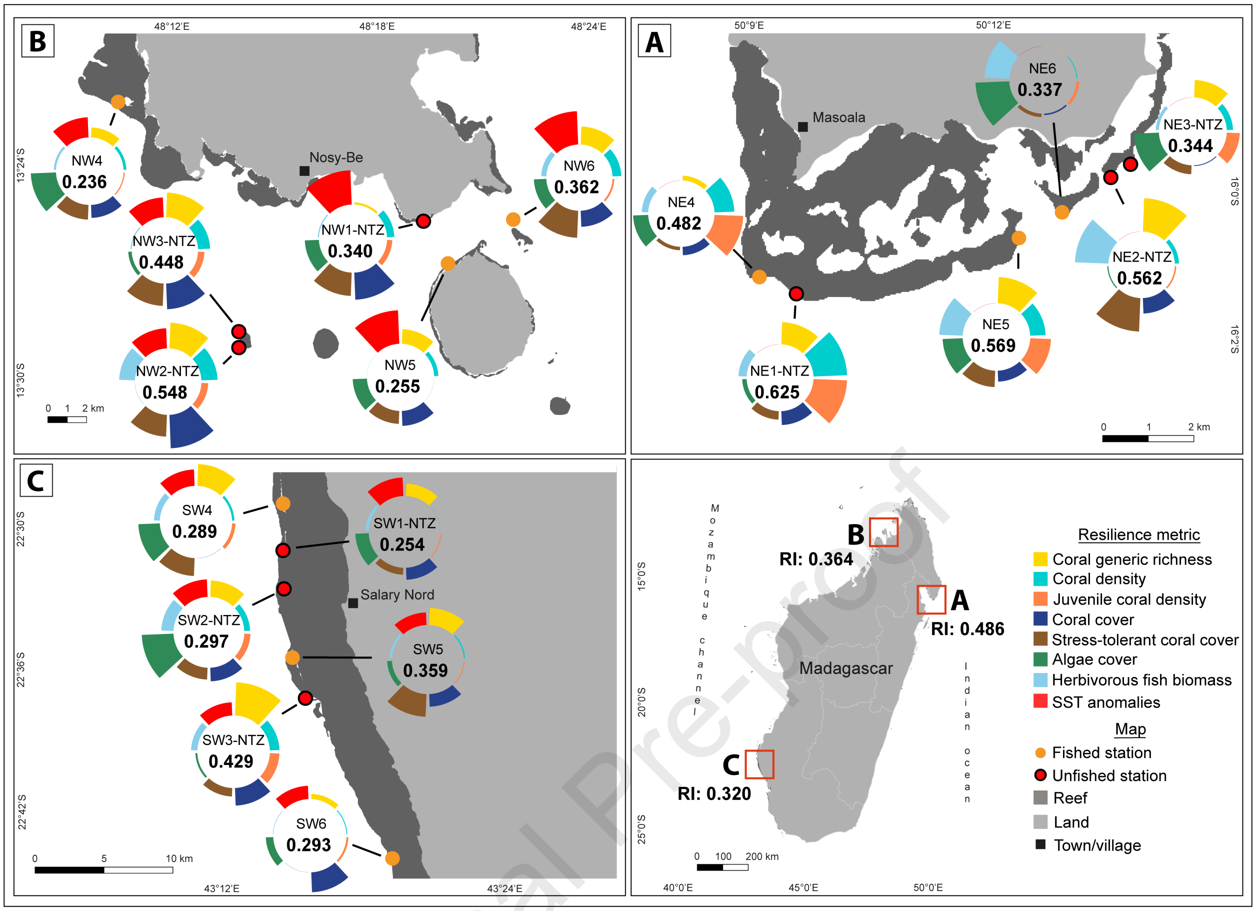
C

Deviation from the modelled pristine condition

$$\frac{\text{Ct-Crp}}{\text{sd(Crp posterior)}}$$

Deviation from the maximum attainable condition given human impact

$$\frac{\text{Ct-Crm}}{\text{sd(Ctm posterior)}}$$


B
48°12'E
48°18'E
48°24'E
13°24'S
13°30'S
Nosy-Be
NW4
0.236
NW6
0.362
NW3-NTZ
0.448
NW1-NTZ
0.340
NW2-NTZ
0.548
NW5
0.255
0 1 2 km
A
50°9'E
50°12'E
16°0'S
16°2'S
Masoala
NE6
0.337
NE3-NTZ
0.344
NE4
0.482
NE2-NTZ
0.562
NE5
0.569
NE1-NTZ
0.625
0 1 2 km
C
22°30'S
22°36'S
22°42'S
SW4
0.289
SW1-NTZ
0.254
Salary Nord
SW2-NTZ
0.297
SW5
0.359
SW3-NTZ
0.429
SW6
0.293
0 5 10 km
43°12'E
43°24'E
Mozambique channel
Indian ocean
B
RI: 0.364
A
RI: 0.486
Madagascar
C
RI: 0.320
15°0'S
20°0'S
25°0'S
0 100 200 km
40°0'E
45°0'E
50°0'E
Resilience metric
Coral generic richness
Coral density
Juvenile coral density
Coral cover
Stress-tolerant coral cover
Algae cover
Herbivorous fish biomass
SST anomalies
Map
Fished station
Unfished station
Reef
Land
Town/village

- French coral reef monitoring programs represent a critical tool to inform about ecological changes
- The use of indicators that integrate key ecological and functional processes is recommended
- These monitoring programs may meet the expectations of national public policies
- The requirements of the Water Framework Directive are incompatible with coral reef ecosystems

Declaration of Interest Statement

☒ The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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☐ The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: