Environmental and Sustainability Indicators 27 (2025) 100811

Contents lists available at ScienceDirect

Environmental and Sustainability Indicators

journal homepage: www.sciencedirect.com/journal/environmental-and-sustainability-indicators

Toward a spatialized index of the "benthic ecological state" using a new Coral Reef Rapid Assessment Method (CORRAM): case study at Reunion Island

L. Broudic [a,b,c,*,1], M. Pinault [d,g,1], R. Claud [b], J.B. Nicet [d], J. Wickel [d], T. Bajjouk [e], T. Rungassamy [f], L. Bigot [g], N. Nikolic [b,h,i], E. Crochelet [b], M. Thibault [g,j], C. Maze [c], B. Bergerot [a]

[a] UMR 6553 University of Rennes CNRS - ECOBIO, Rennes, France
[b] ARBRE (Agence de Recherche pour la Biodiversité à la Réunion - Research Agency for Biodiversity in La Réunion Island), La Réunion, France
[c] UMR 7048 CNRS Sciences Po - CEVIPOF - IRN APOLIMER Chaire Outre-mer, 75007, Paris, France
[d] MAREX Ltd- Marine Expertise Company. Saint-Leu, Reunion Island, France
[e] IFREMER Centre de Brest, DYNECO/LEBCO, BP 70, 29280, Plouzané, France
[f] Réserve Naturelle Marine de La Réunion, Saint-Paul, France
[g] UMR Entropie, University of Saint-Denis, Reunion Island, France
[h] INRAE AQUA ECOBIOP, Saint Pée sur Nivelle, France
[i] Centre de Recherche sur la Biodiversité et l'Environnement (CRBE), Université de Toulouse, CNRS, IRD, Toulouse INP, Université Toulouse 3 – Paul Sabatier (UT3), Toulouse, France
[j] Association BESTRUN, 97490, Saint-Denis, La Réunion, France

ARTICLE INFO

Keywords:
Coral reef health
Circular plot
RAM
Coral reef index
Western Indian ocean island states

ABSTRACT

The development of a spatial index that can be used to represent the benthic ecological state of coral reefs is essential for guiding public conservation policies. We propose the spatialization of a new index of "benthic ecological state" using the Coral Reef Rapid Assessment Method (CORRAM). Eight benthic biodiversity indicators selected from the literature that reflect key resistance and resilience mechanisms were assessed in 786 circular plots on the Saint-Pierre reef flat of Reunion Island. The "benthic ecological state" index was constructed from the eight benthic biodiversity indicators and spatialized using ordinary kriging. To better assess functional mechanisms with respect to environmental pressures, the "benthic ecological state" index can be sub-divided into two indexes. By establishing the associations between these indexes and abiotic environmental variables, the "coral community structure" and "benthic community vitality" showed a differential response sensibility including interaction with abiotic environmental variables studied. Our large dataset enabled us to propose a "benthic ecological state" of reference based on habitat type for the Saint-Pierre reef flat in 2021. These indexes highlight areas of high socio-ecological issues and contribute to quantifying the deviation between the "reference ecological state" and the "benthic ecological state" at a given spatio-temporal point. This proof of concept provides a methodological framework that can be replicated at multiple scales (from local to global) in other reefs. Using a field-based method with spatial indexing and pressure data, it is now possible to accurately locate and quantify areas of coral reef are vulnerability and of major concern.

1. Introduction

In the "Status of Coral Reefs of the World: 2020" report, Souter et al. (2021) unequivocally state that coral reefs are in decline. Between the 1990s and 2020, 50 % of the world's coral cover was lost, and this decline still occurs. However, assessing sessile benthic communities in coral reefs still poses considerable challenges for the scientific community. Indeed, changes in ecosystem characteristics can be relatively

* Corresponding author. Léo Broudic. Affiliation University of Rennes, CNRS, ECOBIO - UMR 6553, Rennes, France – IRN APOLIMER – Science Po CNRS CEVIPOF Chaire Outre-mer Paris, France – ARBRE (Agence de Recherche pour la Biodiversité à la Réunion - Research Agency for Biodiversity in La Réunion Island), La Réunion, France.

E-mail addresses: leobroudic@proton.me, leo.broudic@univ-rennes.fr (L. Broudic).

[1] Co first authors.

https://doi.org/10.1016/j.indic.2025.100811
Received 20 May 2025; Received in revised form 10 July 2025; Accepted 21 July 2025
Available online 30 July 2025

significant and occur at differing speeds, particularly in response to local (McLean et al., 2016) and global (Castro-Sanguino et al., 2021; Obura and Grimsditch, 2009) anthropogenic pressures. Accordingly, studying the link between ecological characteristics and environmental pressures requires the selection of assessment indicators based on their responses to these pressures (Castro-Sanguino et al., 2021; Reverter et al., 2024). Finally, reef areas can be very large and heterogeneous, thereby requiring a spatial approach based on sampling methodologies that facilitate representative assessments of the surface area and diversity of habitats (Andréfouët, 2008; Bajjouk et al., 2019).

Assess the status of reefs at global scale is generally based on key indicators, such as live coral cover, that can provide a reasonably good indication of ecosystem health (Miloslavich et al., 2018). However, the live coral cover is not positively or linearly related to coral biodiversity (Richards, 2013) and the exclusive use of benthic organisms cover for the assessment of coral reefs does not meet the recommendations of international programs in terms of ecological indicators. The GeoBON Essential Biodiversity Variables (EBVs) and the International Coral Reef Initiative (ICRI) Five A's (Accessible, Accountable, Assessment, Actionable, and Aligned) recommend the use of ecological indicators that reflect ecosystem persistence and self-organizing functions over time (McClanahan et al., 2002; McManus and Polsenberg, 2004; Pereira et al., 2013). From the perspective of assessing the resilience of coral reefs in the context of climate change, the most relevant ecological indicators for managers of marine protected areas have been identified and prioritized (McClanahan et al., 2012). Thanks to the abundant literature, the structuring and regulatory mechanisms of coral reefs are well understood (Bellwood et al., 2004; Roff and Mumby, 2012). Several proxies facilitate the collection of data in the field, such as coral biodiversity which is reflected by its structural complexity (Pratchett et al., 2015; Richards, 2013).

Although useful to a certain extent, when used to assess complex ecosystems, one limitation of individual indicators is that they can mask certain effects or draw attention to specific pressures (Heink and Kowarik, 2010). Accordingly, a number of authors have assessed the utility of combinations of these indicators to provide a globally more comprehensive index (Alfsen and Sæbø, 1993; Maynard et al., 2015). These indexes, with or without weighting, provide synthetic information on the status of an ecosystem as a whole (holistic ecological indicator: Jameson et al., 2001) and can be used to quantify an ecological processes (Bajjouk et al., 2019; Brandl et al., 2024; Castro-Sanguino et al., 2021; Jouval et al., 2023; Maynard et al., 2015; Reverter et al., 2024). These indexes must be relevant for the evaluation of ecosystem resilience [i.e., the capacity of an ecosystem to return, at least temporarily, to a stable state following a disturbance (Pimm, 1984)] and resistance [(i.e., the capacity of an ecosystem to withstand a disturbance without significant alteration of ecological functions (Holling, 1973)].

To ensure full operability, these indexes can be associated with reference thresholds to facilitate the interpretation of their spatio-temporal variability and assess their sensitivity (Jameson et al., 1998, 2001). For coral reefs, the establishment of spatio-temporal thresholds would enable the comparison of results against the median ecological state for one area (Obura et al., 2017). At the local scale, the reference ecological state represents the optimal ecological state at a given time and for a given reef complex (e.g. landscape unit such as a habitat) (Clewell and Aronson, 2012; Jameson et al., 1998). Accordingly, the difference between the regional standard and the reference ecological state provides an estimate of the "reef performance" of a local ecosystem (Castro-Sanguino et al., 2021).

Regardless of the indicator or index targeted, one of the main problems in assessing the health of coral reefs is acquiring data on a large spatio-temporal scale. In this regard, two main methods are currently used, namely *in situ* "quantitative" methods (e.g. English et al., 1997; Hill and Wilkinson, 2004) and remote sensing *via* aerial imagery (Obura et al., 2019; Teague et al., 2022). *In situ* methods, such as the Linear Intercept Transect, the Point Intercept Transect or the Photo Quadrat, can be used to facilitate the monitoring of reef benthic communities with the highest level of taxonomic accuracy (Souter et al., 2021). Remote sensing facilitates the spatial and temporal monitoring of ecological indicators, such as coral bleaching (Xu et al., 2021) or algal inflorescences (Brisset et al., 2021). However, *in situ* methods and remote sensing techniques each present specific constraints and limitations. While *in situ* methods are time-consuming and limited to a small number of stations, remote sensing involves high acquisition and processing costs (Bajjouk et al., 2019; Nguyen et al., 2021; Teague et al., 2022).

In situ "semi-quantitative" Rapid Assessment Methods (RAMs) use standardized estimators (scoring) based on ecological processes. It contributes to reduce the sampling time per station (Ervin, 2003; Fennessy et al., 2007; Quétier et al., 2014; Reiss and Hernandez, 2018; Sayre et al., 1999). RAMs appear to meet the measurability and representativeness objectives required for the spatialization of ecological data. Compared with quantitative methods, the use of RAMs requires less time per station and a lower level of taxonomic expertise. Moreover, they are considerably less expensive than classical methods or remote sensing and enable the establishment of larger sampling plans (Fennessy et al., 2007). The quality of the ecological information collected will depends on the quality of the standardization of the method (Quétier et al., 2014) to limit the subjective effect of the operator during the estimates (Meyer et al., 2015). In this regard, appropriate short-term training and inter-calibration sessions have been highlighted as effective measures for significantly reducing the subjectivity of operators (Herlihy et al., 2009; McInnes and Everard, 2017).

In this context, in this study, we develop an index for assessing the ecological status of coral reefs (i.e., a "benthic ecological state" index) that is compatible with spatialization needs and based on ecological indicators acquired *in situ* using the Coral Reef Rapid Assessment Method (CORRAM – Broudic et al., 2025; Pinault et al., 2025). To this end, (*i*) on the basis of a scientific literature review, we selected eight descriptive benthic indicators (i.e. Benthic Biodiversity Indicators; BBIs) of the composition, structure and vitality of coral reefs that respond to environmental pressures and (*ii*) obtained estimates of these eight BBIs for the Saint-Pierre reef flat (Reunion Island) using the CORRAM method. In addition, (*iii*) we developed a standardized and normalized index that provides spatialized information on the ecological state and quantifies the "reef performance" (Castro-Sanguino et al., 2021), defined as the deviation between the "reference ecological state" and the "benthic ecological state" of the sampled reef community. Furthermore, (*iv*) we assessed the sensitivity of this index to 10 abiotic environmental variables known to influence the "benthic ecological state" index. Moreover, we propose splitting the "benthic ecological state" index into two sub-indexes (i.e. the "coral community structure" and "benthic community vitality" indexes). These three indexes are intended to be used as tools to assess the ecological state of a coral reef, while identifying areas of high socio-ecological issues. We are proposing a robust and replicable proof of concept on a specific site (Saint-Pierre reef flat) to enable our approach to be replicated on other reefs.

2. Methodology

2.1. Study area

Reunion Island (−21.14°, 55.53°), a French overseas territory located 680 km east of Madagascar in the southwestern Indian Ocean. It is a young volcanic island, approximately 2.1 million years old (Cadet, 1980), the coral reefs of which began developing along the west and south-west coasts approximately 8000 years ago (Battistini et al., 1975). The Saint-Pierre reef flat comprises a reef zone between the high-water mark and reef front (Fig. 1a–c). The reef encompasses 12 habitat type (adapted from Nicet et al., 2016), which are characterized by their geomorphology (substrate type, topographic complexity and bathymetry) and dominant benthic communities (dominant coral and

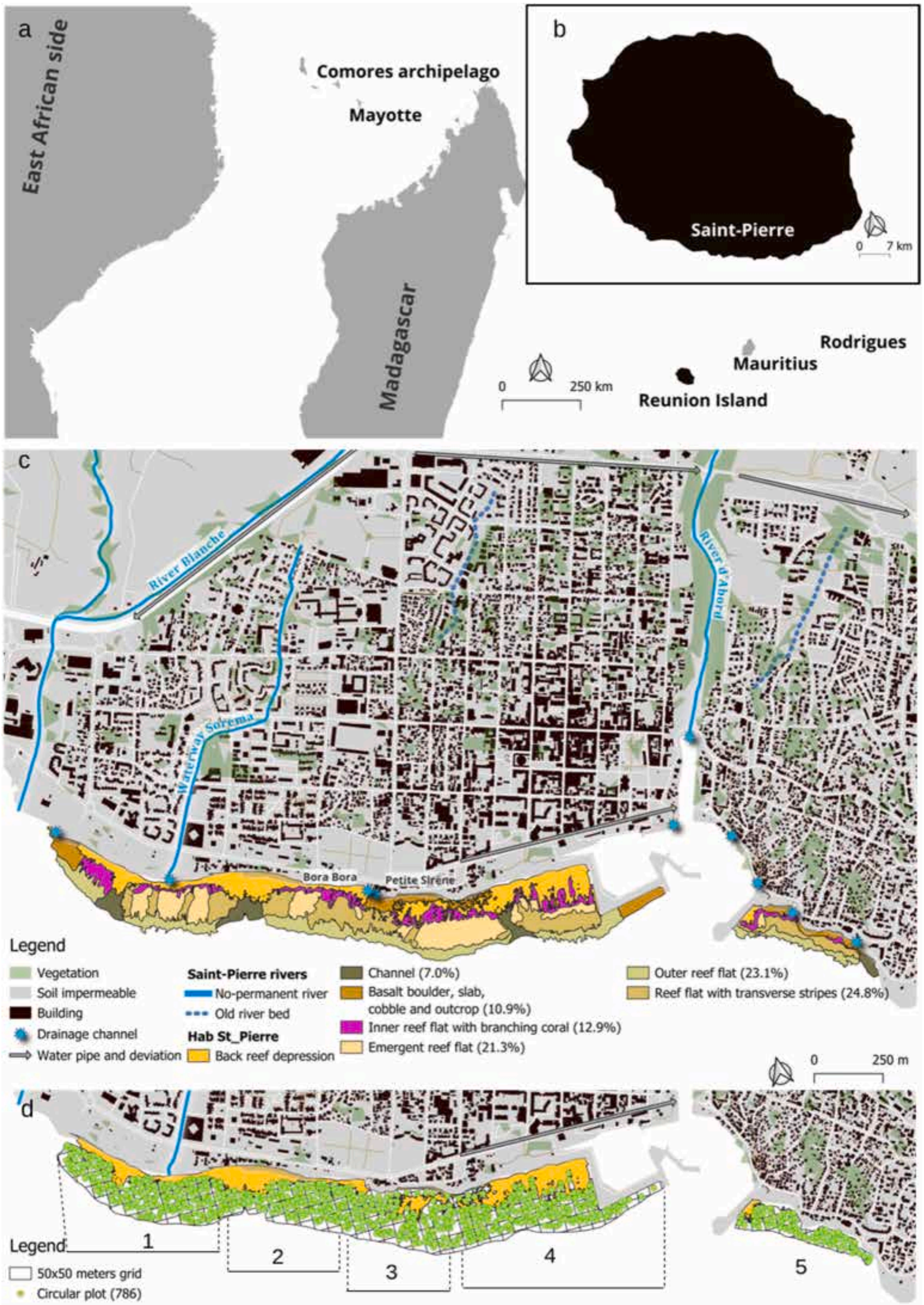

Fig. 1. Location of the commune of Saint-Pierre in Reunion Island and schematic zoning of the urban stormwater network (Saint-Pierre town, com. pers.). a: Reunion Island within Western Indian Ocean, b: Reunion island, c: mapping of the Saint-Pierre - Terre-Sainte reef flat with the relative surface areas of the reef habitat between parenthesis (back reef depression is excluded from the calculation, adapted from Nicet et al., 2016). d: Sampling plan for hard substrate benthic communities: 5 circular plots of 100 m2 per 50 × 50 m grid (786 replicates). 1 to 5: areas.

non-coralline genera). The Saint-Pierre reef is adjacent to the town of the same name, which has been heavily developed over the last 50 years, rendering the soil impermeable and disrupting rainwater runoff, some of which is channeled through pipes into rivers (Fontaine, 2007).

2.2. Benthic biodiversity indicators (BBIs)

2.2.1. Selection of BBIs

Our initial step, based on the scientific literature, was to compile a list of descriptive ecological indicators, selected with respect to their differential responses to environmental pressures. Then, indicators had to meet four criteria: (*i*) the time of *in situ* estimation according to Dahl's method (1981) with inter-observer calibration using RAM, (*ii*) the absence of redundancy in the ecological information provided by the assessed indicators, (*iii*) their documented sensitivity to different environmental pressures (i.e. climate change, eutrophication, pH anomalies, physical degradation, and siltation) and (*iv*) the adaptability of the regional standards established in this study for coral reefs of the southwest Indian Ocean.

Eight BBIs were selected (Table 1): (1) live coral cover, (2)

Table 1
List of Coral Reef Rapid Assessment Method (CORRAM) benthic biodiversity indicators (BBIs). Functional roles, *in situ* assessment methods, and sensitivity to different BBI pressures. N = number of circular plots.

Ecology function assessed by the BBI	BBI	Quantitative estimation associated score	Mean values ± standard error for the Saint-Pierre reef flat (n = 786)	Pressures with known adverse effects on BBI
SCLERACTINIA				
1. Live Cover coral				
Coral polyps synthesize their calcareous skeleton in the form of calyces, which accumulate on top of each other to form, strengthen and grow the reef. They are its founding organisms (Graham et al., 2011; Risk, 1972; Roberts and Ormand, 1987). Coral reef monitoring is therefore based primarily on the percentage of hard substrate covered by living coral colonies. The higher a habitat's coral cover, the greater its bio-constructive activity (Clements et al., 2018; Done, 1991; Hughes et al., 2007; McClanahan et al., 2012).	What percentage of hard substrate is covered by living coral communities? **Estimate the percentage of cover in 100m^2**	Scoring – Estimation 0. – 0 % 0.5 – 1 % 1. – 2 %–5 % 1. – 6 %–15 % 2. – 16 %–33 % 2.5. – 34 %–61 % 3. – 62 %–100 %	20.5 % ± 17.7 %	<u>Watershed influence</u> (Araujo et al., 2015; Carlson et al., 2019; McClanahan and Obura, 1997; Pastorok and Bilyard, 1985; Reopanichkul et al., 2009; Risk, 2014; Stoddart, 1969; Tuttle and Donahue, 2022; Victor et al., 2006; Wear and Thurber, 2015) Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> (Barkley et al., 2015; Gagliano et al., 2010; Pelejero et al., 2005; Stoddart, 1969; Turquet et al., 2001) Temperature pH Salinity Dissolved oxygen <u>Geomorphological infuence</u> (Bajjouk et al., 2019; Kench and Brander, 2006; Knutson et al., 1997; Montaggioni and Faure, 1980; Scopélitis et al., 2009) Bathymetry Proximity to river Proximity to outer reef flat <u>Natural impact</u> (Cane, 1997; Conand et al., 2000; Ginsburg et al., 2018; Lenihan et al., 2015; Reidenbach et al., 2009, 2021; Rosenberg and Ben-Haim, 2002) *Acanthaster* invasion Global changes and recurring climatic phenomena Hydrodynamics
2. Percentage of *Acropora* genus within the coral community				
Under favorable abiotic conditions, the genera *Acropora* and *Isopora* (Acroporiae) grow rapidly, becoming dominant genera in the coral stand (Darling et al., 2012; Wallace, 1978). In the Indo-Pacific, high cover by these genera reflects a high stage of ecological succession (theoretical climax stage – Pratchett et al., 2015). Given their low tolerance to variations in abiotic conditions (notably rising water temperatures), their low proportion generally reflects the chronic and/or acute action of natural and/or anthropogenic pressures (Edinger and Risk, 2000; Naim et al., 2000; Patton, 1994; Young et al., 2012).	What percentage of live coral cover is represented by the genera *Acropora* and *Isopora*? **Estimate the percentage of cover in 100m^2**	Scoring – Estimation 0. – 0 % 0.5. – 1 %–2 % 1. – 3 %–7 % 1.5. – 8 %–19 % 2. – 20 %–40 % 2.5. – 41 %–67 % 3. – 68 %–100 %	34.5 % ± 29.1 %	<u>Watershed influence</u> Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen <u>Geomorphological infuence</u> Bathymetry Proximity to river Proximity to outer reef flat <u>Natural impact</u> *Acanthaster* invasion Global changes and recurring climatic phenomena Hydrodynamics
3. Structural complexity				
As they grow, corals can adopt a range of spatial organizations. Diversified forms provide a high level of complexity, enabling many associated species to become established themselves (Chabanet et al., 1997; Veron, 2000; Wahab et al., 2018). There is a link between coral architecture and the ecological structure of the ecosystem. Coral reefs can be classified according to a succession of morphotypes of increasing complexity. These morphological facies are reliable predictors of several aspects of reef conservation value, including coral species richness and the presence of rare coral species (Edinger and Risk, 2000; English et al., 1997; Pratchett et al., 2015).	Which growth forms (morphotypes) are most common in coral reefs? **Estimate a quantitative value in 100m^2**	0. Where present, coral stands are dominated by massive and sub-massive, encrusting forms with low relief (nb shapes <2). 1. In addition to the above morphotypes, coral stands feature foliose, digitate or corymbose forms, with more pronounced relief (nb shape [3 or 4]). 2. Coral morphotypes are diversified and tend to grow vertically: columnar and thick-branched forms appear	4.5 ± 2.8 nb shapes	<u>Watershed influence</u> Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen <u>Geomorphological infuence</u> Bathymetry Proximity to river Proximity to outer reef flat <u>Natural impact</u> Hydrodynamics

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Table 1 (*continued*)

Ecology function assessed by the BBI	BBI	Quantitative estimation associated score	Mean values ± standard error for the Saint-Pierre reef flat (n = 786)	Pressures with known adverse effects on BBI
		(nb shape [5 or 6]). 3. Coral morphotypes are highly diversified, with branching and/or tabular forms dominating the population and providing a wide range of habitats (nb shape [7 or 8]).		
4. Mean diameter of coral colonies Although growth rates vary widely between species, it is recognized that the larger a coral colony, the older, more resistant and more fertile it is (Shinn, 1966; Counsell et al., 2019). The size classes of living colonies therefore provide information, based on the growth rates of the species concerned, on the time elapsed since the last major disturbance of the ecosystem, resulting in potential or proven mortality (dead colonies) of the largest colonies (Harvell et al., 1999; Miller et al., 2000; Naim et al., 2000). They also provide information on the ability of living colonies to survive future pressures (Stoddart, 1969).	How are the size classes of living colonies distributed within the coral population? **Estimate a quantitative value in 100m^2**	0. Where present, living colonies are homogeneous in size, with diameters mostly under 5 cm and no colony larger than 40 cm. 1. Living coral colonies are homogeneous in size, with diameters mostly less than 15 cm, and no colony larger than 40 cm. 2. Living coral colonies are heterogeneous in size, with small, medium and large colonies over 40 cm. 3. Living coral colonies are heterogeneous in size, with small, medium and large colonies over 100 cm.	39 ± 13 cm	<u>Watershed influence</u> Siltation Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen <u>Geomorphological infuence</u> Bathymetry Proximity to river Proximity to outer reef flat <u>Natural impact</u> *Acanthaster* invasion Global changes and recurring climatic phenomena Hydrodynamics
5. Coral state of health Exposure of a living coral colony to a stress or pathogen can cause multiple physiological responses of increasing severity, ranging from reduced fertility and growth, to depigmentation (fluorescence, bleaching) or the appearance of tissue necrosis (Fuess et al., 2017; Green and Bruckner, 2000; Séré et al., 2015). This general state of health can be revealed by external, visible and recognizable characteristics. The presence of dead colonies, the final stage in physiological responses, indicates a general disturbance in the abiotic characteristics of the environment (Ben-Tzvi et al., 2004; Hughes, 1994; Wallace, 1978).	What is the general state of health of coral colonies (necrosis, debris, mortality) and what is the prevalence of disease within the population? **Estimate a semi-quantitative value in 100m^2**	0. Coral colonies show numerous signs of necrosis and disease. Some may be bleached. Many colonies are already dead, with an accumulation of debris. 1. Coral colonies show frequent necrosis and disease symptoms, but few colonies are dead and debris is scarce. 2. Coral colonies show rare necrosis and/or disease symptoms, with very few dead colonies (i.e. < 5) and debris are observed. 3. No colonies are dead, necrotic, bleached, broken or infected. Coral stands show maximum vitality.	1.6 ± 0.8/3	<u>Watershed influence</u> Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen
6. Juvenile coral density The density of juvenile corals (1–5 cm in diameter – Jouval et al., 2023) is an indicator of the capacity for settlement (colonization) or population renewal (resilience). A high density is indicative of strong demographic dynamics (Ben-Tzvi et al., 2004; Jouval et al., 2019). By settling on hard substrates not occupied by adult coral colonies (limiting density regulation), recruits enter into competition with other organisms in the	What is the density of juvenile corals (1–2 cm) observed on hard substrates not occupied by adult coral colonies? **Estimate the number of juveniles in four 50×50 cm quadrats**	Scoring / Estimation 0. 0 juvenile 1. 1 to 2 juveniles 2. 3 to 4 juveniles 3. >4 juveniles	1.5 ± 1.7 ind.m^{-2}	<u>Watershed influence</u> Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen <u>Geomorphological infuence</u> Bathymetry Proximity to river

(*continued on next page*)

Table 1 (*continued*)

Ecology function assessed by the BBI	BBI	Quantitative estimation associated score	Mean values ± standard error for the Saint-Pierre reef flat (n = 786)	Pressures with known adverse effects on BBI
benthic community (notably algae) and may be consumed by excavating species (Trapon et al., 2013). This ecological balance can be disturbed by natural and/or anthropogenic pressures, resulting in lower density values (increased mortality – Counsell et al., 2019; Hughes et al., 2019; Meesters et al., 1996; Shinn, 1966).				Proximity to outer reef flat <u>Natural impact</u> *Acanthaster* invasion Global changes and recurring climatic phenomena Hydrodynamics
ALGAE				
7. Fleshy algae cover (>2 cm)				
Algae are primary producers, feeding on inorganic nutrient salts (nitrates, phosphates). They colonize the hard substrates of euphotic zones and compete with other species in the benthic community (Fichez et al., 2005). The balance of this competition may be tilted in their favor following the disappearance of herbivores or an excessive supply of nutrient salts (Graham et al., 2014; Rasher and Hay, 2010; Zubia et al., 2018). They then invade hard substrates and can smother and poison coral colonies, substantially reducing their vitality and rate of recovery. In addition to their proven role as competitors of coral populations, macroalgae are also monitored as part of water quality bio-monitoring networks (fleshy macroalgal index). A list of species has been compiled for the reefs of Reunion Island (Zubia et al., 2018).	What percentage of hard substrate is covered by erect algae (>2 cm in height)? **Estimate the percentage of cover in 100m^2** Species list for Reunion Island: **Phaeophycae** *Dictyota* spp *Lobophora variegata* *Turbinaria ornata* **Chlorophytae** *Bryopsis pennata* *Chaetomorpha vieillardii* *Cladophorpsis sundanesis* *Derbesia* sp1 *Ulva* spp *Valonia* spp *Caulerpa* spp *Cauterpa* spp *Dictyosphaeria cavernosa* *Boergenesia forbesii* **Rhodophytae** *Gracilaria* spp *Hypnea* spp *Peyssonnelia* spp **Cyanobacteriotae** *Anabeana* sp1 *Hydrocoleum* spp *Leptolyngbya* spp *Lyngbya* spp *Phormidium hendersonii* *Symploca* spp	Scoring / Estimation 0. / 85 %–100 % 0.5. / 22 %–84 % 1. / 10 %–21 % 1.5. / 5 %–9 % 2. / 3 %–4 % 2.5. / 1 %–2 % 3. / 0 %	9.7 % ± 14.2 %	<u>Watershed influence</u> Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen <u>Natural impact</u> Global changes and recurring climatic phenomena Hydrodynamics
NON-CORAL SESSILE FAUNA (e.g. SPONGES, ZOANTHARIAE, ALCYONARIAE, GORGONARIAE AND ANTIPATHARIAE)				
8. Opportunistic species cover				
Non-coral sessile benthic fauna (e.g. sponges, Zoantharia, Alcyonaria, Gorgonaria, Antipatharia, ascidians) compete with other species in the benthic community (Bellwood et al., 2004; Chadwick and Morrow, 2011; Wulff, 2001). However, these heterotrophic organisms, feeding on organic particles suspended in seawater, can withstand high turbidities, beyond the tolerance thresholds of coral and algal species. Their presence on coral reefs is therefore linked to the attenuation of light intensity with depth (Biggerstaff et al., 2017), or as a result of chronic degradation of water quality (turbidity, suspended organic matter – Bell et al., 2021; Fong and Paul, 2011).	What percentage of hard substrates is covered by non-coral sessile fauna? **Estimate the percentage of cover in 100m^2**	Scoring / Estimation 0. / 85 %–100 % 0.5. / 22 %–84 % 1. / 10 %–21 % 1.5. / 5 %–9 % 2. / 3 %–4 % 2.5. / 1 %–2 % 3. / 0 %	4.3 % ± 6.2 %	<u>Watershed influence</u> Siltation terrigenous inputs Proximity to wastewater Turbidity <u>Physicochemical parameters</u> Temperature pH Salinity Dissolved oxygen <u>Natural impact</u> Hydrodynamics

percentage of *Acropora* genus within the coral community, (3) juvenile corals density estimated per square meter, (4) structural complexity (i.e. number of coral growth forms), (5) mean diameter of coral colonies, (6) coral state of health, (7) opportunistic species cover, and (8) fleshy algae cover. The final two BBIs are considered information-enhancing, in that a high percentage cover of these organisms reflects ecosystem degradation, whereas the other BBIs indicate positive ecosystem attributes. Details of the definitions and relevance of these BBIs, as well as their responses to different pressures, are outlined in Table 1.

2.2.2. *Coral reef rapid assessment method (CORRAM)*

The fieldwork was conducted from September 7th to December 17th, 2021, at the Saint-Pierre reef flat during the intermediate season period, which is characterized by more stable environmental conditions compared with the dry and wet seasons (e.g., a lower risk of heat peaks, strong swells, or heavy rain, Fig. S1). To estimate the eight BBIs, an experienced operator used the CORRAM method, based on the use of a circular plots (Edwards et al., 2017; Ortiz and Tissot, 2008) and a visual quadrat approach (Hill and Wilkinson, 2004). Each circular plot served as a 100 m^2 station, the center of which was marked by a weighted rope, with a 5.6 m radius being using a fiberglass tape measure. The topology of the surface makes it possible to capture the spatial heterogeneity of a reef, particularly on reef flats (Duvall et al., 2019), whilst providing an entire circular plot in the field of view. The coordinates of the center of each circular plot were recorded using a Garmin 76© GPS system.

Two operators collected the data. Before sampling, they made inter-calibration of Benthic Biotic Indicators (BBIs) visual estimation on one circular plot. Each operator first produced its own estimates, which were then compared to assess their consistency.

A preliminary tour of each station was conducted to gain an overall perspective. Initially, we obtained estimation for the cover of benthic organism indicators. Given the three-dimensional structure of the reef, the sum of these percentages can exceed 100 %. The structural complexity of the reef was estimated based on coral form (Dahl, 1981) and the mean diameter of the coral colonies. Given its qualitative nature (score of between 0 and 3), coral state of health was estimated separately. Juvenile coral density was estimated using four randomly placed quadrats (50 × 50 cm). Unidentified organisms were photographed for subsequent confirmation of identity. For each station, estimates of the eight BBIs required approximately 3–5 min.

The sampling plan was based on dividing the reef flat according to a grid, in which each 50 × 50 m square mesh measured 2500 m^2 (Fig. 1d). Within each mesh, five randomly located circular plots were assessed. Given that at the edges of the reef, the meshes were unable cover a 2500 m^2 area, only one to four circular plots were assessed. In addition, some meshes, particularly those close to the reef front, could not be assessed, owing to the risk of accidents (strong swelling, very shallow depths). Prior to commencing data collection, the operators performed self-calibration on two to five circular plots, thereby contributing to a reduction in observational heterogeneity and enhanced comparability.

The timing of field sessions was dependent on the strength of the swell (<1.5 m) and water depth (>0.7 m) to ensure that the reef flat was accessible for 2 h before and after the high-tide slack. In total, 10 days (40 h) under water were necessary to ensure sufficient sampling.

2.3. *BBIs and indexes*

2.3.1. *Calibration, normalization and standardization of BBIs*

Prior to index construction, BBIs were calibrated using experimentally determined minimum and maximum values. Subsequently, they were normalized to reduce positive skewness and approximate a Gaussian distribution, and then standardized by converting raw values into a uniform scoring range from 0 to 3.

Normalization and standardization required the development of transformation formulas, which were parameterized based on the ecological implication of each BBI, whether it reflected a beneficial (i.e., increasing values indicated improved ecological state) or detrimental (i.e., increasing values indicated worsening ecological state) attribute.

For beneficial BBIs (live coral cover, proportion of *Acropora* within the coral community, juvenile coral density, structural complexity, mean diameter of coral colonies, and coral state of health), the transformation function is detailed in Equation (1):

$$a^*\ln(x+1)+b^*x \tag{1}$$

where "a" is the normalization constant adjusting the convexity of the logarithmic curve,

And "b" is the standardization constant ensuring that the normalized scores remain within the 0–3 range.

For detrimental BBIs (opportunistic species cover and fleshy algae cover), the transformation function is detailed in Equation (2):

$$3-(a^*\ln(x+1))+b^*x \tag{2}$$

Where the subtraction by 3 reverses the score, thus resulting in a low score when the coverage of these two BBIs is high.

2.3.2. *Calculation of indexes*

The "benthic ecological state" index corresponds to the average of the eight BBIs, previously calibrated, normalized, and standardized, then converted to a 0 to 10 scale. This score is calculated without weighting. The conversion of raw values (ranging from 0 to 3 per BBI) to a 0 to 10 scale is based on a division by 24 (8 BBIs × 3), followed by a multiplication by 10, according to Equation (3):

$$\sum\frac{BBIs^*10}{24} \tag{3}$$

To gain a more nuanced understanding of the ecological mechanisms at play, particularly those related to resilience and degradation, the "benthic ecological state" index can be divided into two complementary sub-indexes. These components specifically describe the "coral community structure" and the "benthic community vitality" (Fig. 2).

The "coral community structure" sub-index characterizes the processes by which reef habitat is built in three-dimensional space, estimating both the habitable volume and the diversity of ecological niches provided by corals to associated organisms (fish, macroinvertebrates, etc.). It offers an assessment of the ecosystem's habitability potential by evaluating the degree of development and diversity of the ecosystem's "architect species."

This sub-index is based on the average of four BBIs "structure": live coral cover, structural complexity, mean colony diameter, and the percentage of *Acropora* within the coral community. These BBIs, once calibrated, normalized, and standardized, respond slowly to environmental disturbances and are reliable descriptors of the ecosystem's maturity and robustness (in terms of age, complexity and diversity). The conversion to a 0–10 scale involves dividing the sum of BBIs "structure" by 12 (4 BBIs × 3) and multiplying by 10, using Equation (4):

$$\sum BBIs\frac{\text{“}structure\text{”}^*10}{12} \tag{4}$$

The "benthic community vitality" sub-index brings together BBIs that assess the ecosystem's current health and its potential for resilience in the face of natural or anthropogenic pressures. It provides a snapshot of the reef's current dynamic trajectory, whether it is regenerating, stable, or declining.

This sub-index is calculated as the average of four BBIs "vitality": coral state of health, juvenile coral density, and the coverage of both fleshy algae and opportunistic species. Once calibrated, normalized, and standardized, these "vitality" indicators respond more rapidly to environmental changes than the structure BBIs. They capture ongoing ecological processes that may ultimately result in positive (conservation/progression) or negative (regression) transformations of the ecosystem's structural components. The raw scores are converted to a 0–10 scale using the same method, with a division by 12 (4 BBIs × 3) followed by a multiplication by 10 (Equation (5)):

$$\sum BBIs\frac{\text{“}vitality\text{”}^*10}{12} \tag{5}$$

In accordance with the classification proposed by Andersen et al. (2004), and many other indice - EFI+: Solana et al. (2009), I2M2: Mondy et al. (2012), IBML: Tison-Rosebery et al. (2023) -, the index is divided into five categories (very bad, bad, moderate, good and very good). As the use of terms such as "bad" or "good" could involves value judgments (Keeney and Gregory, 2005) these categories are described in the Supplementary Material (Table S3). The data transformation process refers specifically to descriptions relevant to the reef flat of Saint-Pierre.

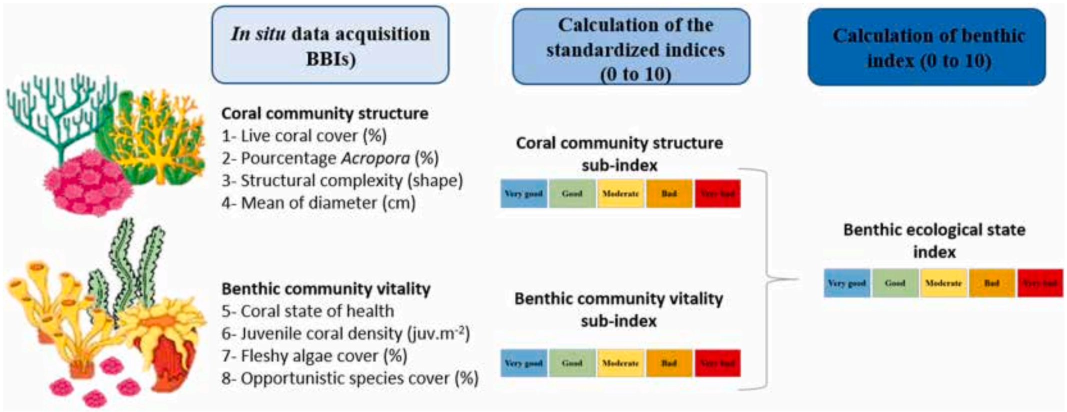

Fig. 2. Calculation, standardization and classification of scores for the index ("Benthic ecological state") and the 2 sub-indexes ("Coral community structure" and "Benthic community vitality" based on the 8 BBIs (benthic biodiversity indicators) assessed *in situ*. "Very bad": score from [0–2], "Bad":]2–4], "Moderate":]4–6], "Good":]6–8], "Very good":]8–10].

Therefore, a "very good" state does not necessarily correspond to an undisturbed area, but rather to the highest possible scores for this area.

2.4. Abiotic environmental variables

In the second step, we obtained *in situ* measurements for 10 abiotic environmental variables, selected for their anticipated influence on BBIs (Table 1) and the feasibility of data collection (cost, measurement time, and instrumentation). The selection process grouped the variables into three categories. Among these, geomorphology (bathymetry and proximity to river, and reef crest) takes into consideration the life history of a reef and its influence on the potential colonization, establishment, and development of Scleractinia's. Watershed variables (proximity to sewage pollution inputs, stress linked to siltation of terrigenous input, and stress linked to turbidity) influence water quality and its impact on benthic communities whereas physico-chemical parameters (temperature, salinity, pH, and dissolved oxygen) reflect the general physical and chemical conditions of reef habitats. The rationale for the selection of environmental variables, as well as the data acquisition methods, is provided in detail in the Supplementary Information (Table S4).

The 10 variables considered provide part of the explanation for the distribution of "benthic ecological status". Due to the lack of spatial data, we have not included certain potentially explanatory variables, such as nutrient concentration or tourism activities.

2.5. Definition of the "reference ecological state" and the "DeltaRef" of habitats

The "reference ecological state" is defined as the highest level of ecological condition reached, based on the "benthic ecological state" index, for a given area and time period. The spatial unit considered is the reef habitat. For a given habitat, the reference ecological state is estimated by calculating the average of the top 5 % of stations with the highest scores within that habitat.

Based on the definitions proposed by the European Union Water Framework Directive (Andersen et al., 2004), the "DeltaRef" is defined here as the difference, for each habitat, between the reference ecological state and the mean benthic ecological state index. The "DeltaRef" was calculated from all the stations within a habitat. This value represents the relative deviation (present as a percentage) of the average condition of a habitat from its "reference ecological state", as observed at the relevant geographic scale. This concept is closely related to the notion of "reef performance" proposed by Castro-Sanguino et al. (2021).

2.6. Statistical analysis

All statistical tests and spatial analyses were performed using RStudio software (Posit team, 2023), and maps were generated using QGIS software 3.34.0 (QGIS Development Team, 2023).

2.6.1. Relationships between BBIs and abiotic environmental variables

The 10 assessed abiotic environmental variables were normalized using a Box-Cox transformation (Box and Cox, 1964), and there associations with the eight BBIs were analyzed using principal component analysis (PCA; FactomineR packages, Husson et al. (2016) and Factoextra, (Kassambara, 2016). The relative contributions of the 8 principal component axes are represented graphically. On the basis of their correlations with the BBIs, the 10 abiotic environmental variables and the sub-indexes and the index were subsequently projected onto a factorial plan.

Pearon's correlation models were used to analyze the relationships between BBIs, sub-indexes and index, and abiotic environmental variables.

Given that habitat is a qualitative variable, its influence on the distribution of BBIs, sub-indexes and the index was assessed using an analysis of variance (ANOVA), followed by a post-hoc Student-Newman-Keuls test. The normality of the residuals (Shapiro and Wilk, 1965) and the homogeneity of variances (Bartlett, 1937) were also assessed, with the same approach being used to compare the "DeltaRef" among habitats.

2.6.2. Spatialization of sub-indexes, index and abiotic environmental variables

Prior to performing spatial interpolation analyses, Moran's I index (Moran, 1950) was calculated to highlight the significant spatial autocorrelation for each sub-index, index, and abiotic environmental variable. The spatial interpolation method was determined using the dichotomous decision tree (Li and Heap, 2008). Maps were produced *via* spatial interpolation using ordinary kriging (Matheron, 1963), and semi-variogram parameters were defined manually using the "variogram" function (gstat library). For each BBI, the number of even-numbered neighbors and the semi-variogram parameters (i.e., sill, exponential model, range, and nugget) were specified based on a complete dataset. Ordinary kriging was generated from the model defined by the semi-variogram using the "gstat" function (gstat library), with each interpolation generating both a prediction and variance maps. Following cross-validation, the mean absolute error and the standard absolute error of the residuals were calculated after cross-validation to verify the mean residuals magnitude between the observations and predictions. The effect of anisotropy was evaluated along the four cardinal points. Each prediction was compared using an ANCOVA analysis with interpolations generated under isotropic conditions. The spatial resolution was calculated according to the station density for each site and zone (Hengl, 2006) and compared following the recommendations of Bajjouk et al. (2019), who have suggested that a resolution of 10 m or finer is required to capture coral spatial heterogeneity.

To examine relationships with abiotic environmental variables, we chose to represent the index graphically using continuous values. This approach allows for a detailed interpretation of spatial distribution, particularly useful for cartographic analysis of pressure-affected areas. In contrast, for management purposes, we used discrete values (categorized as "Very bad", "Bad", "Moderate", "Good", "Very good") to break down the sub-indexes. This representation improves readability and visual distinction between value classes, facilitating the identification of homogeneous index areas.

Having spatialized the sub-indexes and index, were calculated area ratio, which represent the proportion of pixels within a given ecological score interval to the total number of pixels.

For cases in which ecological values were estimated if the field, the spatialization of abiotic environmental variables was used to extract the corresponding values for each circular plot and for each of the 786 circular plots, we obtained a database of ecological index scores and abiotic environmental variable.

3. Results

3.1. Description of the ecological state of the Saint-Pierre reef flat

A total of 786 circular plots with an average spacing of 13 m were surveyed, equivalent to 22.5 stations. ha^{-1}. The Saint-Pierre reef flat has an average live coral cover of 20.5 % ± 17.7 % (Table 1), with 34.5 % ± 29.1 % driving by the genus *Acropora*, characterized by a wide variety of morphologies (4.5 ± 2.8 shapes/100 m^2) and a mean colony diameter of 39 ± 13 cm. Notably, the coral colonies generally show signs of stress, such as disease and necrosis, particularly those of *Acropora* species (e.g. *Acropora muricata*, in inner reef flat). In contrast, other genera and *Acropora* species, such as *Acropora abrotanoïdes*, in the outer reef flat showed less evidence of stress and disease. Juvenile corals had an average density of 1.5 ± 1.7 ind. m^{-2}, with taxa competing for space with Scleractinia covering 9.7 % ± 14.2 % for fleshy algae and 4.3 % ± 6.2 % for opportunistic benthic fauna. High standard deviation values indicated significant spatial disparity.

The "benthic ecological state" index varied between habitats (Fig. 3). Habitats classified as "basalt, boulder, slab, cobble and outcrop" and "emergent reef flat" with means of 3.7/10 ± 1.3 and 4.2/10 ± 1.6, respectively, were found to have the lowest scores, whereas in contrast, the "outer reef flat" near from the wave actions had the highest scores, averaging 7.3/10 ± 1.3. Although we detected no significant difference between the habitats categorized as "channels" and "inner reef flat with branching corals" in terms of their benthic ecological state, with respect to the sub-index "coral community structure", the score obtained for the channel habitat were significantly lower than those for inner reef flat (4.2 ± 1.4 vs. 6.1 ± 2.0 respectively). Conversely, for the sub-index "benthic community vitality", the channel scores were significantly higher than those obtained for inner reef flat (6.0 ± 1.4 and 4.5 ± 1.5). Moreover, for this sub-index, these scores obtained for inner reef flat were found to be similar to those obtained for the basalt and emergent reef flat habitats.

3.2. Relationship between index and sub-indexes according to abiotic environmental variables

3.2.1. Assessment of relationships using a multivariate approach

The standardized and normalized BBIs are positively correlated (Fig. 4a). These BBIs were divided into two categories, namely, those related to the "coral community structure", including live coral cover, structural complexity, the mean diameter of coral colonies, and percentage of *Acropora* genus within coral communities (lower right of the PCA), those related to the "benthic community vitality", including fleshy algae cover, opportunistic species cover, coral state of health and juvenile coral density (upper right of the PCA). The "benthic ecological state" index is between the "coral community structure" and "benthic

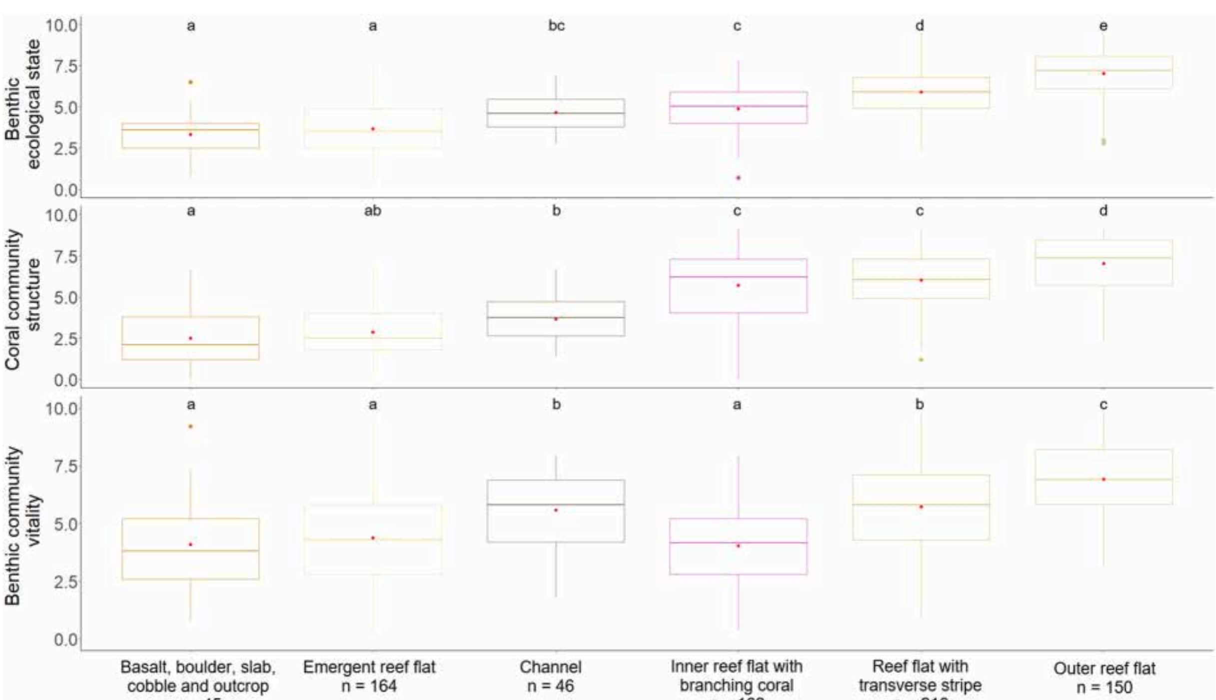

Fig. 3. Boxplot of "benthic ecological state" index and the both sud-indexes "coral community structure" and "benthic community vitality" in relation to reef habitats. The red dots represent the mean values. Significant differences between habitats represented by a letter (ANOVA tests following by Student-Newman-Keuls test, p-value threshold = 0.001).

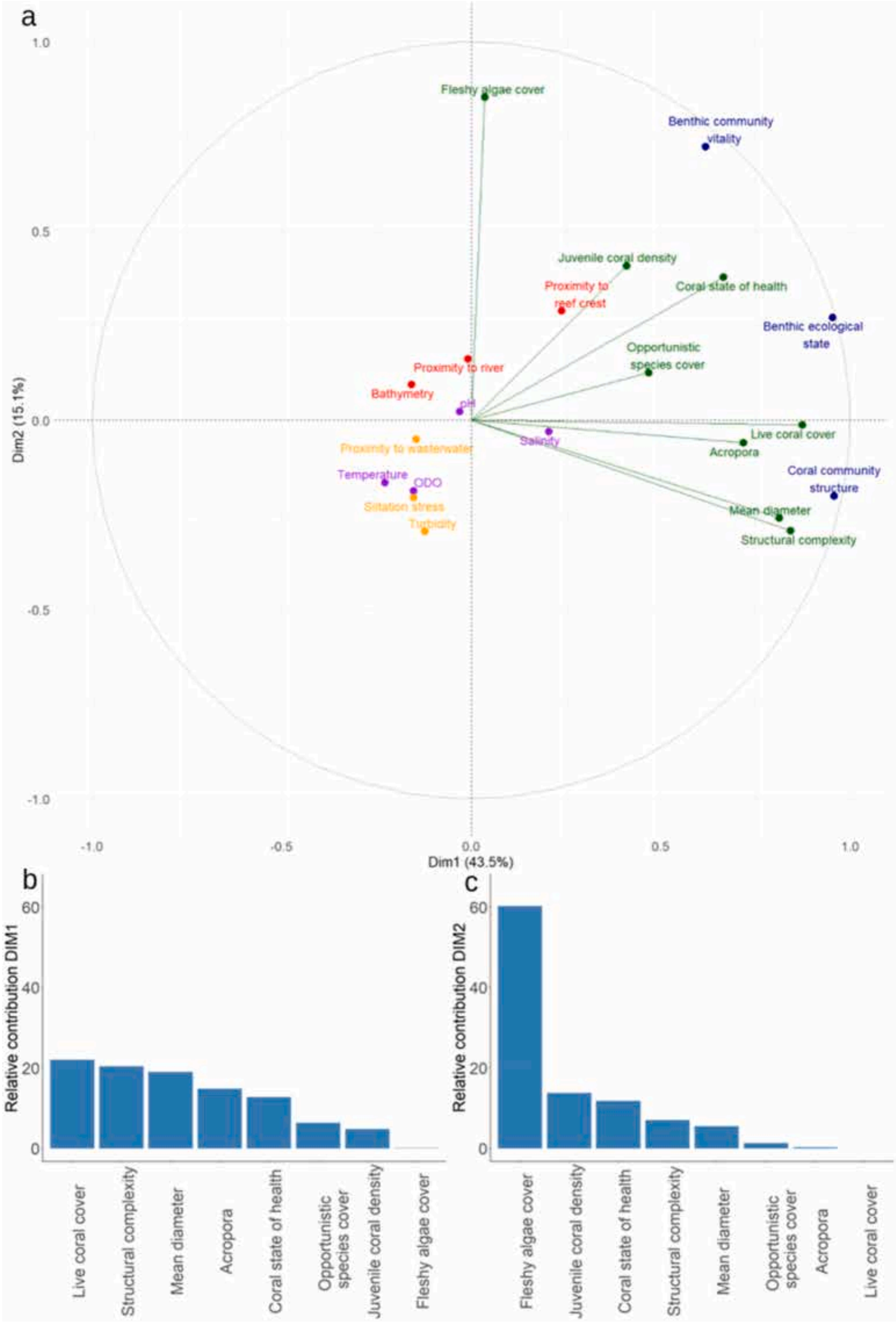

Fig. 4. Results of the principal composante analysis performed on the 786 circular plots for the BBIs. a: The 8 BBIs in green, index and sub-indexes in blue and the 10 abiotic environmental variables are grouped by categories (ODO: dissolve oxygen, colors, yellow: watershed influence, mauve: physicochemical parameters and red: geomorphological influence). Only the BBIs contribute to the PCA (active variables), while the environmental variables are included in the analysis as supplementary variables. Axis 1 accounts for 43.5 % and axis 2 for 15.1 % of the data's variability b: Histogram showing eigenvalues of the composant analysis for the DIM1, c: Histogram showing eigenvalues of the composant analysis for the DIM2.

community vitality". Abiotic environmental variables related to watershed influence and physico-chemical parameters were grouped together and inversely associated with "benthic community vitality", whereas those associated with geomorphological influences, such as bathymetry and proximity to rivers, were inversely correlated with "coral community structure". With respect to the ecological data, axes 1 and 2 of the PCA explained 58.6 % of the total variance, of which 43.5 % and 15.1 % was explained by axes 1 and 2, respectively. Furthermore, 22 % of the axis 1 was by live coral reef, 20 % by structural complexity, 19 % by mean diameter and 15 % by percentage of the genus *Acropora* within the coral population. Axis 2 is 60 % explained by fleshy algae cover, 14 % by juvenile coral density, 12 % by coral state of health and 1 % by opportunistic species cover (Fig. 4b and c).

The indices and watershed influence were significantly and negatively correlated (Fig. 5), as were temperature and dissolved oxygen. The indices showed the strongest positive correlations with proximity to the reef crest, particularly the "Coral Community Structure" index. Proximity to the reef crest was also identified as the abiotic environmental variable most strongly correlated with all BBIs (Fig. S2).

The correlations between the sub-indexes and the abiotic environmental variables were stronger than those observed for the "benthic ecological state" index. "Coral community structure" and "benthic community vitality" indexes being significantly negatively correlated with watershed influence, including turbidity, proximity to wastewater, and siltation terrigenous stress. The physico-chemical parameters were positively correlated with salinity and negatively correlated with temperature and dissolved oxygen level. With the exception of the positive correlation with proximity to the reef crest, the correlations with geomorphological influence variables differed between these two sub-indexes. Moreover, whereas the "coral community structure" was positively correlated with proximity to the reef crest, although not significantly correlated with bathymetry, the "benthic community vitality" in contrast, showed no significant correlation with proximity to the reef crest, although was negatively correlated with bathymetry.

3.2.2. Spatial correlations

The variance maps and mean residuals of the sub-indexes and index are presented in the Supplementary Information (Figs. S3 and S4).

By overlaying the maps of "benthic ecological state", turbidity, sedimentation, temperature (as a proxy for water residence time), and reef habitats, we were able to spatialized the correlations highlighted in Fig. 5 (Fig. 6). Areas of high turbidity and sedimentation were accordingly found to coincide with areas of low "benthic ecological state", particularly in the vicinity of the Sorema channel (zone 1) and the "Bora-Bora" and "Petite Sirène" rainwater pipelines (zone 3). The heavy sedimentation in zone 4 was mainly confined to the inner reef flat with branching coral habitat, in the same area in which the ecological state was also low. Emergent reef flat habitats were established to be areas with low "benthic ecological states". Finally, the highest temperatures were recorded in zones 2 to 4, with an increasing gradient moving toward the open sea. Areas with a high "benthic ecological state" were located in areas in the outer reef flat, in which turbidity, sedimentation (except in zone 5), and temperature were at their lowest level.

3.2.3. Index and sub-indexes spatialization and quantification

Of the entire Saint-Pierre reef flat, 5 % was in a very good "benthic

Fig. 5. Correlation matrix between the index and both sub-indexes and abiotic environmental variables. The values shown are the correlation coefficients, which indicate the strength (values between −1 in blue and 1 in red) and direction (negative or positive values) of the linear relationship between each pair of variables following a Pearson correlation analysis. Only significantly related relationships are associated with a correlation coefficient (p-values <0.05). The colored squared indicate the category of abiotic environmental variables, i.e. yellow: watershed influence, mauve: physicochemical parameters and red: geomorphological influence. ODO: dissolve oxygen.

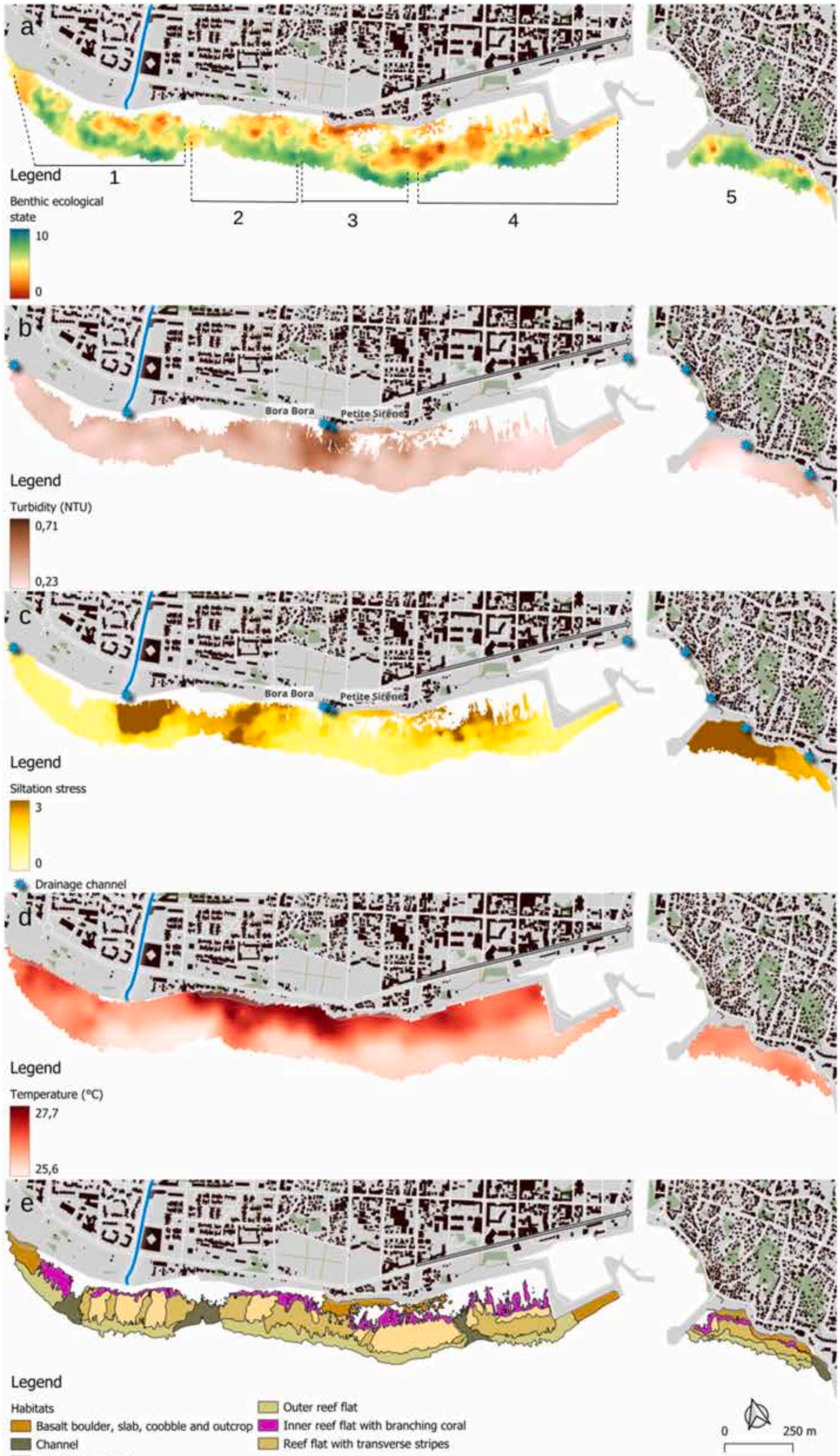

Fig. 6. Correspondence between "benthic ecological state" and three abiotic environmental variables. a: "benthic ecological state", b: turbidity (NTU), c: siltation index and d: temperature (°C) and e: habitats. Temperature is a proxy for water mass residence time: the warmer the temperature, the longer the water mass tends to remain in a restricted zone.

ecological state" (score >8) (Fig. 7a). These areas had a strong "coral community structure" and "benthic community vitality" (Fig. 7, picture 2). In contrast, 2 % of the reef flat was in a very bad "benthic ecological state" (Fig. 7, pictures 5 and 6). Areas with moderate scores for "benthic ecological state" (between 2 and 8), represented 42 % of the reef flat, and were distributed in different configurations (Fig. 7, pictures 1, 3 and 4). Comparatively, 7 % of the Saint-Pierre reef flat was in a very good "coral community structure" (scores >8) (Fig. 7b), with these areas characterized by high coral cover (>50 %), large coral colonies (>100 cm), mainly of the genus *Acropora,* and branching and tabular growth forms (Fig. 7, pictures 1 and 2). A further 6 % was established to be covered by a very bad "coral community structure" (scores ≤2), characterized by low coral cover (<5 %), small coral colonies (<10 cm) and more simple and poor growth forms (encrusting, massive and sub-massive) (Fig. 7, pictures 4, 5, 6). The most structured zones were located at the periphery of the reef flat, specifically in the inner reef flat

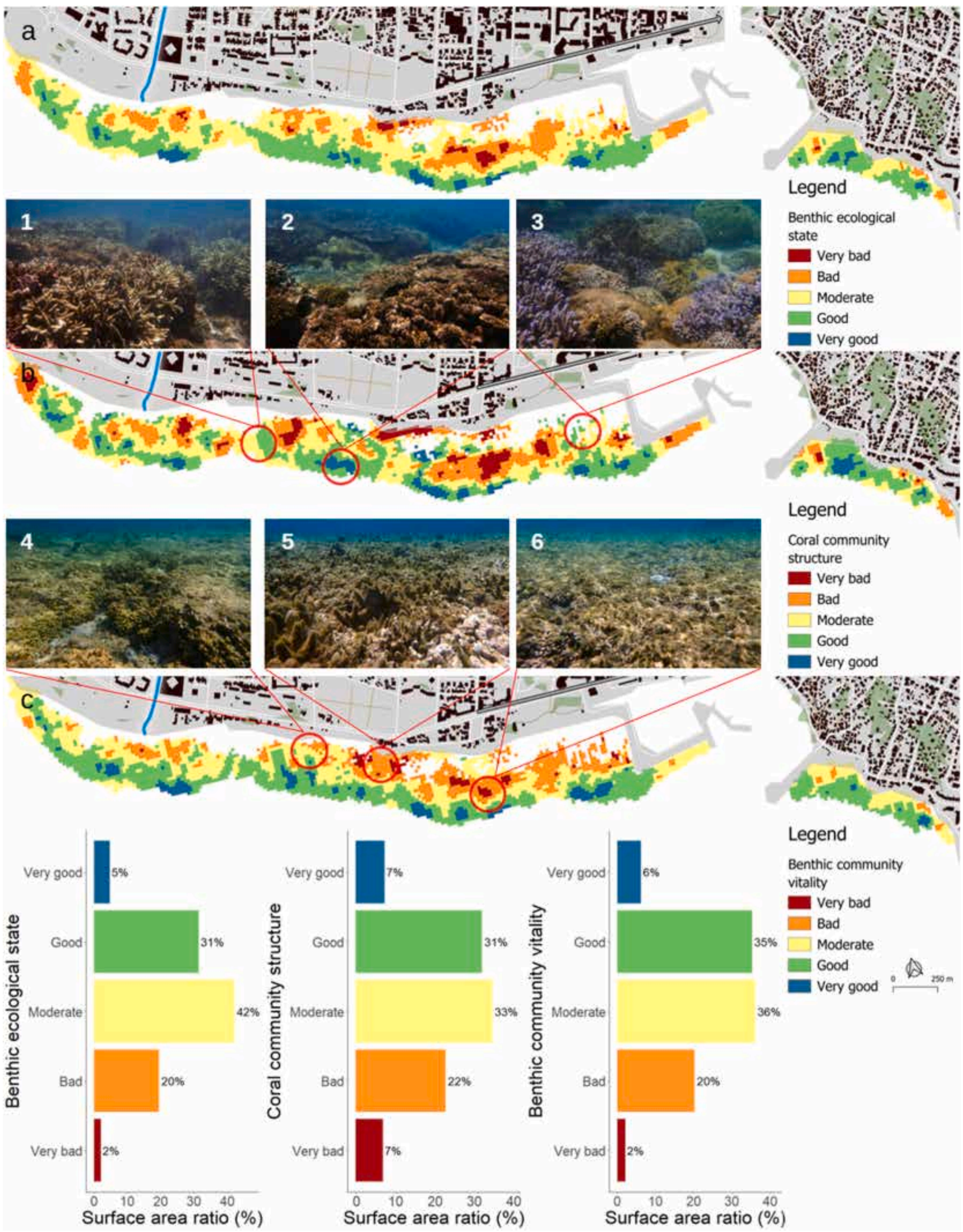

Fig. 7. Spatialization and quantification of surface area ratios of the index and sub-indexes on the Saint-Pierre reef flat. a: "benthic ecological state", b: "coral community structure", c: "benthic community vitality". Overlay of models and reef habitats most relevant to index scores. The emergent reef flat habitat is a proxy for very low bathymetry, and the outer reef flat is a proxy for proximity to the reef crest. The photos correspond to the following cases 1: Moderate score for "coral community structure" and "benthic community vitality", 2: Very good score for "coral community structure" and "benthic community vitality", 3: Good score for "coral community structure" and moderate score for "benthic community vitality", 4: very bad score for "coral community structure" and moderate score for "benthic community vitality", 5: very bad score for "coral community structure" and "benthic community vitality" in a habitat conducive to coral development, and 6: very bad score for "coral community structure" and "benthic community vitality" in a habitat not conducive to coral development.

with branching coral and the outer reef flat. The least structured zones were distributed in discontinuous patches located at the center of the reef flat. With regard to “benthic community vitality”, 6 % of the Saint-Pierre reef flat was in very good high vitality (scores>8) (Fig. 7c), reflected by a particularly low cover of fleshy algae and opportunistic species (<5 %), coral colonies with very few signs of disease and stress, and a high density of juvenile corals (>4 juveniles. m^{-2}) (Fig. 7, picture 1). Conversely, 2 % of the reef was in very bad vitality (scores from ≤2), as indicated by a high cover of fleshy algae and opportunistic species (>20 %), a high proportion of diseased coral colonies (characterized by white and black streaks), signs of stress and extensive necrosis with rubble, and few or no juvenile corals (≤1 juvenile. m^2) (Fig. 7, pictures 4, 5, 6). The most degraded areas was located close to the shore, mainly in areas without beaches and with extensive urbanization. Conversely, the healthier areas were distributed along offshore section of the reef, following a coast-wide gradient, even if lower vitality is noticed around the harbor dock and estuaries.

3.3. “Benthic ecological state”, “reference ecological state” and “DeltaRef” by habitat

Although the “basalt boulder, slab, cobble and outcrop” habitat had the lowest “benthic ecological state” and “reference ecological state” scores of 3.3 ± 1.2 and 5.6 ± 0.8, respectively (Table S1), the $\mathrm{DeltaRef}_{\text{basalt boulder, slab, cobble and outcrop}}$ index indicated this habitat to be the second most degraded habitat in terms of the “reference ecological state” with a mean score difference of 2.3 ± 0.4 (a deficiency of performance of 40 % ± 22 %) (Fig. 8). As designated by “DeltaRef”, the most degraded habitat was the $\mathrm{DeltaRef}_{\text{Emergent reef flat}}$, with a mean score difference of 3.4 ± 2.2 (a deficiency of performance of 48 % ± 23 %). Conversely, the outer reef flat had the highest “benthic ecological state” and “reference ecological state” values of 7.0 ± 1.4 and 9.2 ± 0.1 respectively, and the lowest $\mathrm{DeltaRef}_{\text{Outer reef flat}}$, with a mean score difference of 2.2 ± 1.3 (a deficiency of performance of 23 % ± 16 %).

4. Discussion

4.1. Benthic ecological state index

The new “benthic ecological state” index is calculated and spatialized using the CORRAM method. These novel approach can be applied to rapidly collect a set of integrative ecological information relating to the ecological state of a reef across a large number of stations (McClanahan et al., 2012). In this study, 8 BBIs were estimated based on sampling within 786 circular plots during a single field season. We then spatialized the “benthic ecological state” index, which, to the best of our knowledge, is the first time this approach has been adopted for an assessment of coral reefs. To assess the validity of the index, we evaluated its sensitivity based on 10 key environmental variables identified by the Biodiversity Indicators Partnership (BIP, Bubb et al., 2010). BIP is a global initiative designed to promote and coordinate the development and delivery of biodiversity indicators for use by the Convention on Biological Diversity, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, and sustainable development goals in the European countries.

The “benthic ecological state” index corresponds to the average of the eight BBIs, scaled to a range from 0 to 10, each of which responds to one or more know pressures. Our results revealed that the index is strongly influenced by geomorphology, physicochemical parameters, and watershed of the reef. Areas characterized by the most degraded states were found overlap with patches showing high siltation terrigenous stress and turbidity, which are assumed to be attributable to wastewater discharge (Alongi and McKinnon, 2005; Reopanichkul et al., 2009). The division of this index into two sub-indexes provide a more comprehensive understanding of the ecological processes operating at different spatial and temporal scales. For example, the inner reef flat with branching coral was established to have a strong “coral community structure” although was assessed to have a low “benthic community vitality”. Near Bora-Bora and Petite Sirène, the current regime is weak (Naim et al., 2001) and subject to high turbidity and sedimentation. The multiple interactions of pressures in this area create a "cocktail effect" (Wear and Thurber, 2015) which is reflected in low scores for all three indices. The geomorphological conditions also significatively influence the BBIs representative of “coral community structure”. The areas where water is rapidly renewed, such as the proximity to the reef crest (Darling et al., 2019; Faure, 1982; Graham et al., 2014; Liddell and Ohlhorst, 1987) and depth (Bajjouk et al., 2019), favor high values of the sub-index. On the other hand, proximity of a watershed has a negative influence on the inner reef flat with branching coral. This may be explained by the fact that this habitat is the receptacle of loaded terrigenous inputs. The effects are all the more noticeable on BBIs composing “benthic community vitality” sub-index (Cleary et al., 2016;

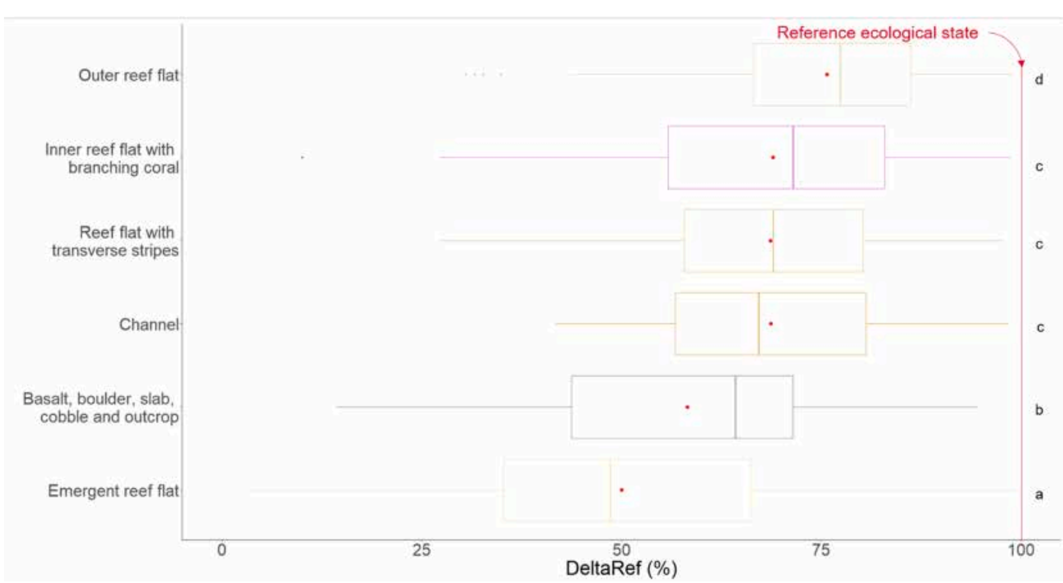

Fig. 8. Boxplot of “DeltaRef” index in relation to reef habitats. The red line represents 100 % of the “reference ecological state”. The red dots represent the mean values. Significant differences between habitats represented by a letter (ANOVA tests following by SNK test, p-value threshold = 0.001).

Gittenberger et al., 2015; Ogden and Gladfelter, 1983).

The data used to calculate BBIs are comparable to those obtained by Global Coral Reef Monitoring Network of the Reunion Island Marine Reserve (Réserve Naturelle Marine de La Réunion, 2024). Although only two GCRMN stations are present on the Saint-Pierre reef flat, the results obtained for these sites are consistent with the BBIs, thereby reinforcing the reliability of our findings (Fig. S5 and Appendix S1). By using an index that reflects responses to environmental pressures, CORRAM could be considered a valid additional tool to conventional monitoring. The "benthic ecological state" index can also be used to complement the relative index developed by Jouval et al. (2023) or the relative resilience potential index proposed by Maynard et al. (2015), both of them assess the resilience potential of benthic community. **Stakeholders** could therefore combine these resilience indices with our ecological state index to identify areas where management measures should be prioritized. Moreover, the use of **CORRAM** and the calculation of the **"benthic ecological state"** do not require advanced taxonomic expertise, making them easily accessible to managers. The advantage of RAMs is that having initially undergone training and inter-calibration of field operators, the managers, consultants, and associations can rapidly initiate monitoring programs (Fennessy et al., 2007). However, visual estimation of RAMs will always be less accurate than methods using measuring tools (Hill and Wilkinson, 2004).

To meet stakeholders' expectations, as the BIP points out (Bubb et al., 2010), the 'benthic ecological state' index can be adapted. Indeed, new BBIs can be introduced or removed according to the specific characteristics of each country and coral reefs. For example, the proliferation of species, such as sea urchins (Echinoderma), sea cucumbers (*Holothuria* spp), sea star (*Acanthaster planci*), which is recognized in several territories (Ditzel et al., 2022; Pierrat et al., 2024; Randall, 1972), plays a major role in the ecological mechanisms of the coral reefs. A BBI score based on species proliferation can thus be integrated into the "benthic ecological state" index. However, the adaptability of the index must be parsimonious, particularly in terms of calculation, in order to remain, as the BIP emphasizes, readily comprehensible, both in terms of the conceptual approach as in its presentation and interpretation.

One of the major advantages of "benthic ecological state" index based on CORRAM's is its spatialization. Maps are relevant educational and practical tools for locating areas of concern according to specific objectives (Hamylton, 2017; Monnier et al., 2021). The distinction between (*i*) "Very Bad/Bad" and "Good/Very Good" areas, and (*ii*) "coral community structure" and "benthic community vitality", facilitates spatial comparisons. This differentiation also supports the implementation of an "early warning system", helping to detect emerging issues before the reef's bio-physical structure is adversely affected (Bubb et al., 2010). These distinctions can be readily assessed using maps by virtues of the normalization process of the index, which sets a score between 0 and 10 with an average close to 5 (Andersen et al., 2004). Areas that deviate from the mean state are thus considered either degrade or in a good state of conservation (Brandl et al., 2024; Castro-Sanguino et al., 2021; Jouval et al., 2023; Maynard et al., 2015; Mumby and Harborne, 2010; Reverter et al., 2024; Thompson et al., 2020). Although absolute index values can also be used (e.g. without normalization process), using relative values makes it easier to compare. Consistent with the indices developed, particularly within the framework of the European Framework Directive, we transformed our quantitative indices (ranging from 0 to 10) to qualitative indices (ranging from very poor to very good). The description of qualitative values, associated with field photographs, significantly reduces the degradation of information related to the shift from quantitative to qualitative measures (Keeney and Gregory, 2005) and is better suited to fulfilling the normative tool of institutionalized indicators (e.g. EFI+: Solana et al. (2009), I2M2: Mondy et al. (2012), IBML: Tison-Rosebery et al. (2023).

If the proof of concept is demonstrated, it is necessary to compare our results with other reefs. For the purposes of this study, we used a restricted dataset developed by Broudic et al. (2025) in which the authors compared BBIs of the reefs of Reunion Island and among reef habitats, notably between the reef flats and the outer slopes. Given the different geographical scope, the normalization formulas had to be adapted to be relative to the entire set of reefs in Reunion Island, thereby enabling inter-reef comparisons. In this regard, current limitations with respect to data availability can be address by CORRAM implementation in coral new coral reefs (Pinault et al., 2025).

This study is limited by the abiotic environmental variables available, other information such as currentology or the diffusion of nutrient salts by runoff water would have been interesting to study. Especially as the exploration of other spatialization methods, such as the interpolation of ecological information under the constraint of abiotic environmental variable (e.g. co-kriging), seems relevant to explore in the future.

4.2. An operational index and tool for coral reef management

The "benthic ecological state" index addresses the need for a more integrative assessment of the ecological status of coral reefs (Díaz-Pérez et al., 2016; Hughes et al., 2018) particularly in the definition of collapse thresholds for this ecosystem (Obura et al., 2022). In addition to assessment, the definition of ecologically preserved zones, such as the outer reef flat of Saint-Pierre, or vulnerable areas, such as the inner reef flat highlighted by branching coral communities, is essential for policymakers and managers. Monnier et al. (2021), through the Water Framework Directive, emphasized the need to spatialize ecological information to facilitate coral reef management. The identification of major pressures sources through a sub-index concerning the benthic community vitality, more sensitive to water quality, would enable stakeholders to take action in a specific area before the structure of a reef deteriorates beyond irreparable levels (De'ath and Fabricius, 2010). Notably in this regard, comparisons of vulnerable areas with the most preserved areas (i.e., the reference ecological state) are necessary to gain a better understanding of the remedial measures necessary to re-establish the abiotic conditions favorable to the resilience of coral populations (Morizot, 2020).

The concept of reference ecological state is established after significant ecological degradation, which means that the baseline chosen may not represent a true natural or pristine state, but rather an ecological starting point shaped by previous human impacts. In the present context, it is the most favorable state at a given time and place (Clewell and Aronson, 2012; Folke, 2006). Accordingly, our finding in the present study are specific to the Saint-Pierre reef flat in 2021. The comparison of certain BBIs with GCRMN data revealed that the reference ecological states we assessed in 2021 are very similar to the GCRMN data from 2002 (with the exception of erect algae, see Table S2). These findings accordingly highlight one of the notable advantages of the CORRAM approach with the spatialization of multiple plots and the capacity to identify all potential variations in an indicator across the spatial heterogeneity of a site and within habitats (Legendre, 1990). This advantage integrates the notion of space-for-time substitution, which indicates that the spatialization of an ecosystem can encompass successive temporal states (Lovell et al., 2023; Pickett et al., 1989). The gaps in the temporal monitoring of the "benthic ecological state" index appear to be compensated for by space-to-time substitution, at least for the period from 2002 to 2021. However, in the 1980s, coral cover was estimated to be around 60 % on the reef flat of Reunion Island (Faure, *pers. comm.*). The definition of a reference ecological state for the Saint-Pierre reef flat in 2021 would remain lower than the pre-2002 periods. These elements are important for managers when defining management measures.

By using normalized indices and their deviations from the ecological reference (DeltaRef), allow to estimate the expected ecological losses and gains in the context of restoration or development projects. This approach aligns with one of Ifrecor's objectives aimed at fulfilling the "no net loss" of biodiversity principle (Levrel and Pioch, 2012; Bas et al., 2016; Bigard et al., 2018), a legal requirement adopted in several countries, including the United States and France (French Biodiversity

Law – 2016). The use of DeltaRef values, combined with a diachronic spatial representation of the "benthic ecological state" index, would enable an evaluation of the effectiveness of management decisions within marine protected areas as well as political choices related to spatial planning and zoning. An increasing DeltaRef indicates ecological degradation, while a decreasing DeltaRef suggests a habitat's improvement toward its reference state. Optimization data collection by streamlining the sampling plan resulted in an estimation cost of \$3 to \$7. ha^{-1} for the outer slope and \$10 to \$19. ha^{-1} for the reef for a circular plot of 5 min and a cost of \$80 per hour (Broudic et al., 2025). Currently, coral reef preservation is under the responsibility of managers, thereby distancing those who have a direct influence on reefs (Morrison et al., 2020). The use of the "benthic ecological state" index and "DeltaRef" by a range of stakeholders would facilitate the implementation of collective measures to preserve and restore the ecological functioning of coral reef ecosystems, and avoid governance traps and placebo policies (Morrison et al., 2020).

5. Conclusion

Our evaluation of the "benthic ecological state" index using the success criteria of the Biological Indicators Partnerships (BIP) is well-suited for operational ecological assessments. For the first time, a field-based method is combined with a spatially explicit index reflecting the ecological state of benthic communities. The spatialization of the index and pressures provides intuitive information on the location of vulnerable and critical coral reef areas. This tool helps identify at-risk areas and supports the development of targeted conservation, remediation, and restoration measures. This approach is based on the application of CORRAM, which effectively addresses the challenges of collecting extensive ecological data from multiple stations. By providing tailored density recommendations for the number of stations per hectare according to the geographic scale, it strikes a balance between field investigation time and the reliability of spatial analyses. Our findings highlight the potential of the "benthic ecological state" index which can be reproduced in other territories across multiple geographical scales. Future studies should also focus on the temporal evolution of reference states of different reefs. Nevertheless, although the benthic ecological state index meets the criteria set by the Biodiversity Indicators Partnership, its adoption by decision-making bodies will require structured institutional support. Closer collaboration with the International Coral Reef Initiative (ICRI), the international counterpart of Ifrecor, could help promote the dissemination of the CORRAM method and the ecological state indices developed in this study.

CRediT authorship contribution statement

L. Broudic: Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Data curation. **M. Pinault:** Methodology, Formal analysis, Data curation, Conceptualization. **R. Claud:** Formal analysis, Data curation. **J.B. Nicet:** Methodology, Conceptualization. **J. Wickel:** Methodology, Conceptualization. **T. Bajjouk:** Validation, Formal analysis. **T. Rungassamy:** Project administration, Data curation. **L. Bigot:** Methodology, Conceptualization. **N. Nikolic:** Validation, Project administration. **E. Crochelet:** Validation, Project administration. **M. Thibault:** Visualization, Project administration. **C. Maze:** Validation, Supervision. **B. Bergerot:** Validation, Supervision, Conceptualization.

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.

Acknowledgements

We acknowledge the association ARBRE and BESTRUN which supported the UTOPIAN project. This research was co-financed by European subsidies (LIFE4BEST – 2020-IO71), the SOS Corail program of the Fondation de la Mer and IFRECOR, with the support of du ministère de la transition écologique et de la cohésion des territoires. Special thanks to Claire Bissery for her proofreading and expert advice.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.indic.2025.100811.

Data availability

https://doi.org/10.17882/106549

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