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Ecological quality assessment: A framework to report ecosystems quality and their dynamics from reference conditions

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Abstract

Worldwide, ecosystems are suffering important taxonomic and functional modifications in response to anthropogenic disturbances, operating at multiple spatial and temporal scales. Awareness on biodiversity losses has led to the adoption of conservation policies and the development of programs devoted to the conservation and the restoration of terrestrial, freshwater, and marine ecosystems. The assessment of the ecological health of ecosystems requires measuring and characterizing restoration or degradation dynamics and their consequences on the ecological quality with respect to reference conditions defined pragmatically as conservation targets. Methodological innovations, in terms of data collection, analysis, and visualization, have an important influence on the ability of ecologists to understand biodiversity changes. The assessment of the quality of ecosystems with respect to reference conditions requires to address, notably, three main challenges: the definition of reference conditions, the assessment of the degree of achievement of conservation objectives, and the qualitative and quantitative characterization of recovering and departing patterns. We propose here the ecological quality assessment (EQA) framework as a data-driven approach to track ecological quality focusing on the distance of the tested stations with respect to a chosen reference envelope using fuzzy logic and trajectory analysis. We take advantage of those analytical tools to propose a general and flexible multivariate framework by quantifying the achievement of reference conditions, measuring restoration and degradation dynamics when temporal series are available, and representing and synthesizing this information. To take into account the natural spatiotemporal variability of sites considered as reference, we gave two variants to our framework: a state-based variant when no temporal replications are available and a trajectory-based variant specially devoted to compare whole trajectories to a

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trajectory reference envelope defined by a set of reference trajectories. These two complementary approaches were illustrated through two terrestrial and marine ecological applications using the R package "ecotraj" in order to evidence ecological observations that meet conservation objectives from those that do not meet them. EQA constitutes a flexible framework for the assessment and reporting of ecosystem quality, including restoration and degradation dynamics adaptable to multiple questions in the different fields of ecology and conservation.

KEYWORDS
BACI, ecological dynamic regime, ecological quality status, ecological trajectory analysis, experimental design, impact studies, recovering, restoration, shifting baselines

INTRODUCTION

Worldwide, ecosystems are suffering important taxonomic and functional modifications in response to anthropogenic disturbances, including overfishing, pollution, global climate change, habitat degradation, and introduction of nonindigenous species (Carmona et al., 2021; Claudet & Fraschetti, 2010; de Lima et al., 2020; Pereira et al., 2012). These disturbances, often cumulative, induce both acute and chronic effects over various temporal and spatial scales. They can ultimately lead to broad-scale loss of habitats as well as to the alteration of community structure, functioning, and associated ecosystem services (Convention on Biological Diversity, 2010; Ellis et al., 2000; UNEP, 2011).

The current loss of biodiversity has led to the adoption of conservation policies (Convention on Biological Diversity, 2010; UNEP, 2011) by many countries and to the development of several programs devoted to the conservation and the restoration of terrestrial, freshwater, and marine ecosystems. In Europe for instance, the Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC), the Water Framework Directive (WFD, Directive 2000/60/EC), and the Habitats Directive (HD, Directive 92/43/EEC) require that Member States implement measures to rapidly reach or maintain a good status of all their marine and terrestrial habitats. Assessing the efficiency of conservation policies requires the capacity to measure and characterize temporal ecological changes occurring within ecosystems, and ultimately, to define the dynamics of ecological quality with respect to reference conditions defined pragmatically as conservation targets (Borja et al., 2012; Hess et al., 2020; Pardo et al., 2012).

For a long time, biodiversity indices have remained the main way for reporting ecological quality. Scientists, stakeholders, and administrative bodies requirement for a synthetic approach to the assessment of ecosystems quality have favored a still ongoing development of biotic indices devoted to synthesize community responses to perturbations and ecological quality assessment (EQA) (Blandin, 1986; Buckland et al., 2005; Diaz et al., 2004; Labrune et al., 2021; Washington, 1984). Some of these biotic indices integrate multivariate data into a single, site-specific, numeric score that can be straightforwardly interpreted by nonspecialists within a "good" versus "bad" range of values, most often in relation with legislative requirement (Diaz et al., 2004). Two main approaches have often been followed: the "sensibility/tolerance to pressure" and the "deviation from reference conditions." For this second category, the number of reference stations must be sufficient in order to capture the spatial and temporal variability of reference conditions (Basset et al., 2013; Briton et al., 2019; Coates et al., 2018; Lavesque et al., 2009).

In practice, some marine biotic indices based on a classification of taxa on a scale of sensitivity/tolerance to a defined pressure (e.g., AMBI; Borja et al., 2000) have proved to be limited in some situations because the knowledge available for linking the species sensibility to a range of pressures is partial, and their use is limited to a specific type of pressure. These approaches have also proven to be limited in assessing the effect of anthropogenic disturbances in naturally stressed ecosystems, such as estuaries (Elliott & Quintino, 2007). In response, indices based on the deviation of the taxonomic composition of tested community with respect to stations of reference have been developed (Flåten et al., 2007; Johnson et al., 2008; Labrune et al., 2021) but defining thresholds remains a challenge to determine the achievement of reference conditions: that is, distinguish stations that meet reference conditions from those that do not meet them, with respect to ecological quality ratios generally ranging from 0 to 1 (Hiddink et al., 2023). What is an acceptable ecological quality assessment? What is a good or bad ecological status? What level of synthesis can satisfy scientists, managers, and stakeholders? The high diversity of indices reveals that the unanimity on any specific single metric by managers and scientists remains a complex task (Diaz et al., 2004). Therefore, this still represents an important challenge for reporting the quality of ecosystems as part of restoration programs, impact studies, or intergovernmental agreements (Borja et al., 2012; McNellie et al., 2020).

In EQA, defining reference conditions is a central question that has been challenging conservation practitioners for decades (Borja et al., 2012; Samhouri et al., 2012). In many fields of ecology, long-term monitoring data sets are consequently increasingly compared with reference data sets that should describe historical ecological states, benchmarks to be achieved, and/or represent objective conservation targets (Bioret et al., 2009). Historical reference states are rare and remain often debatable, as they have potentially been defined in ecosystems under levels of pressure already well established. Alternatives for reference conditions definition can be in unaffected, or less affected habitats (Borja et al., 2012; Coates et al., 2018). For instance, McNellie et al. (2020) propose the complementary concept of contemporary reference state that focuses on current ecological patterns and the definition of reference in areas with higher biodiversity.

The definition of quantitative, data-driven, reference conditions requires sampling designs similar to those underpinning long-term monitoring (Tomczak et al., 2022). Such assumptions have been recently summarized by Labrune et al. (2021): (1) reference stations should be defined within a same or at least a similar ecological entity (Borja et al., 2012); (2) a set of reference stations is preferable that one station, as it allows a better integration of the spatial variability of reference conditions (Lavesque et al., 2009); (3) even if they do not represent an optimum, historical data may represent an alternative in the absence of reference stations (Borja et al., 2012; Tomczak et al., 2022); (4) shifting-baselines claims for a synchronous monitoring of both potential references and tested stations over longer time periods to highlight their dynamics and disentangle natural from anthropogenic drivers (Basset et al., 2013; Hess et al., 2020). If fulfilled, these assumptions require analytical tools tailored to deal with this design.

Methodological innovations, in terms of data collection, analysis, and visualization, have an important influence on the ability of ecologists to advance understanding of biodiversity changes (Magurran et al., 2019). In changing ecosystems, even if spatial variability is taken into account, temporally static reference conditions may be limited to measure the quality of ecosystems with respect to rolling reference conditions (Thorpe & Stanley, 2011; White & Walker, 1997), as "baselines" or "reference states" are better viewed as envelopes that are dependent on the time window of observation (Hawkins et al., 2017; Samhouri et al., 2012).

Hiers et al. (2012) proposed the dynamic reference concept to incorporate the temporal and spatial changes in reference ecosystems. They used Mahalanobis distance with nonmetric multidimensional scaling ordination to quantify the dynamics of reference plots and determine restoration plots that were within the 90% confidence region of initial benchmark species compositions. Although this work has provided substantial new perspectives in the consideration of reference conditions in restoration ecology, a more formal, flexible, and explicit multivariate framework is clearly still lacking to comprehensively address the spatial and temporal variation of ecosystems for EQA in different restoration and conservation contexts for the diverse fields of ecology.

Hiddink et al. (2023) reviewed the most frequently used approaches to set thresholds for good ecosystems state (among which Babcock et al., 2010; ICES, 2021; Jac et al., 2020; Lester et al., 2013; McNellie et al., 2020; Ricard et al., 2012; Rice et al., 2012; Rossberg et al., 2017; Wedding et al., 2013). They pointed methods focusing on "staying within natural variation" as the most relevant and easiest to operationalize the distinction between good and degraded states. Belonging to this category, the assessment of the ecological quality with respect to reference conditions imposes to address, notably, three main challenges: (1) the selection of ecosystem states considered as reference conditions, (2) the assessment of the achievement of reference conditions and conservation objectives, and (3) the qualitative and quantitative characterization of recovering and departing patterns with respect to reference conditions. Here, we propose the EQA framework, as a data-driven approach to track ecological quality focusing on the inclusion or exclusion of tested stations with respect to a reference envelope using fuzzy logic and trajectory analysis. Fuzzy logic allows expressing the uncertainty of the EQA over a continuous scale. The hard statement "the tested ecosystem falls within the reference envelope" can be expressed in a fuzzy way with a degree of membership [0–1] (Krishnapuram & Keller, 1993) expressing to which degree the statement is true according to a quantification method (here our quality function). Ecological trajectory analysis (ETA) is a framework to analyze ecosystem dynamics described as trajectories in a chosen space of ecosystem resemblance with no limit in the number of included dimensions (De Cáceres et al., 2019; Sturbois, Cucherousset, et al., 2021; Sturbois, De Cáceres, et al., 2021). ETA considers trajectories as objects to be analyzed and compared geometrically in a chosen multivariate space. Trajectory analyses are used here to better represent the ecosystem dynamics in both reference envelop and tested ecosystem.

Here, we take advantage of fuzzy logic and ETA to propose a general and flexible EQA framework for assessing and reporting the quality of ecosystems by (1) testing the achievement of reference conditions and conservation objectives, (2) quantifying and qualifying restoration and degradation dynamics when temporal series are available, and (3) representing and synthesizing information for stakeholders and managers.

To take into account for natural temporal variability of sites used to define the reference envelope, our framework includes two variants: a *state-based* variant when ecosystem dynamics are not explicitly considered (even though temporal replications may be present) and a *trajectory-based* variant devoted to compare the whole trajectory of the tested ecosystem to a reference envelope also defined by a set of ecological trajectories. After implementing new functions to the R package "ecotraj," the EQA framework was tested through two examples from different fields of ecology and conservation, and complementary contexts where the assessment of the ecological quality of communities with respect to reference conditions was relevant: (1) conservation assessment of terrestrial habitat in a management and restoration context, (2) impact of fishing activities on the taxonomic properties of a marine habitat under an experimental design. Rules for future use were finally discussed in relation with the concept of reference conditions.

CONCEPTS

Comparing ecological quality of ecosystems requires defining the ecological attributes to be compared and how the corresponding multivariate space is defined, which will depend on ecological questions determined by the user. The different steps of EQA are supported by this chosen multivariate space of ecological resemblance, including the definition of the reference envelopes, the test of reference conditions achievement for tested ecosystems, and the quantification and qualification of restoration and degradation dynamics. EQA is based on a flexible definition of the reference conditions which is not limited to pristine or climax states. It includes for instance reference data sets that describe historical ecological states, benchmarks to be achieved and/or represent objective conservation targets (Bioret et al., 2009), to facilitate the exploration of complementary ecological questions at different scales in the fields of ecology.

On the multivariate space Ω

More formally, let Ω be a multivariate space representing the resemblance between a set of ecological observations. We assume that Ω is defined by the resemblance between pairs of observations, measured using a dissimilarity coefficient d. All following analyses are based on the dissimilarity values contained in a distance matrix $\Delta = [d]$. Let x_i contain the coordinates, or ecological state, of an ecological observation i in Ω.

The ecological applications used in this paper are based on species composition at the community scale, but note that EQA is applicable for different ecological and conservation questions requiring multivariate analysis (trace element at individual scale, functional trait, or environmental variables at ecosystem scale; Sturbois, Cucherousset, et al., 2021; Sturbois, De Cáceres, et al., 2021).

EQA is not limited in the number of dimensions taken into account, and d may measure differences in species composition or any other characteristic considered as being ecologically relevant. Furthermore, the coordinates x_i do not need to be explicit, because the distance matrix Δ contains all the information that is relevant (i.e., the relationships between ecological states).

Reference envelopes and their variability

State reference envelope

Ecological states considered as conservation targets are used to define the reference envelope (E), represented by a set of r observations (o_1, o_2, ..., o_r) with their corresponding ecological states (x_1, x_2, ..., x_r) in a chosen multidimensional space Ω (Figure 1A). Even though the state reference envelope does not explicitly consider ecosystem dynamics, users can integrate the spatial and temporal variability of reference conditions in the assessment of ecological quality, by including spatial and/or temporal replicates in the set of r ecological states. This variability of the reference envelope in Ω can be estimated using (Legendre & De Cáceres, 2013):

$$\mathrm{Var}(E) = \frac{\sum_{i=1}^{r-1}\sum_{j=i+1}^{r} d\left(x_i, x_j\right)^2}{r\times(r-1)}. \quad (1)$$

$\mathrm{Var}(E)$ can be interpreted (and is equal, in the case of an Euclidean metric) as the squared average distance from states to the centroid of the envelope (Figure 1B). When observations o_1, ..., o_r represent spatial variability in community composition, $\mathrm{Var}(E)$ is a measure of beta diversity which is the variation in species composition

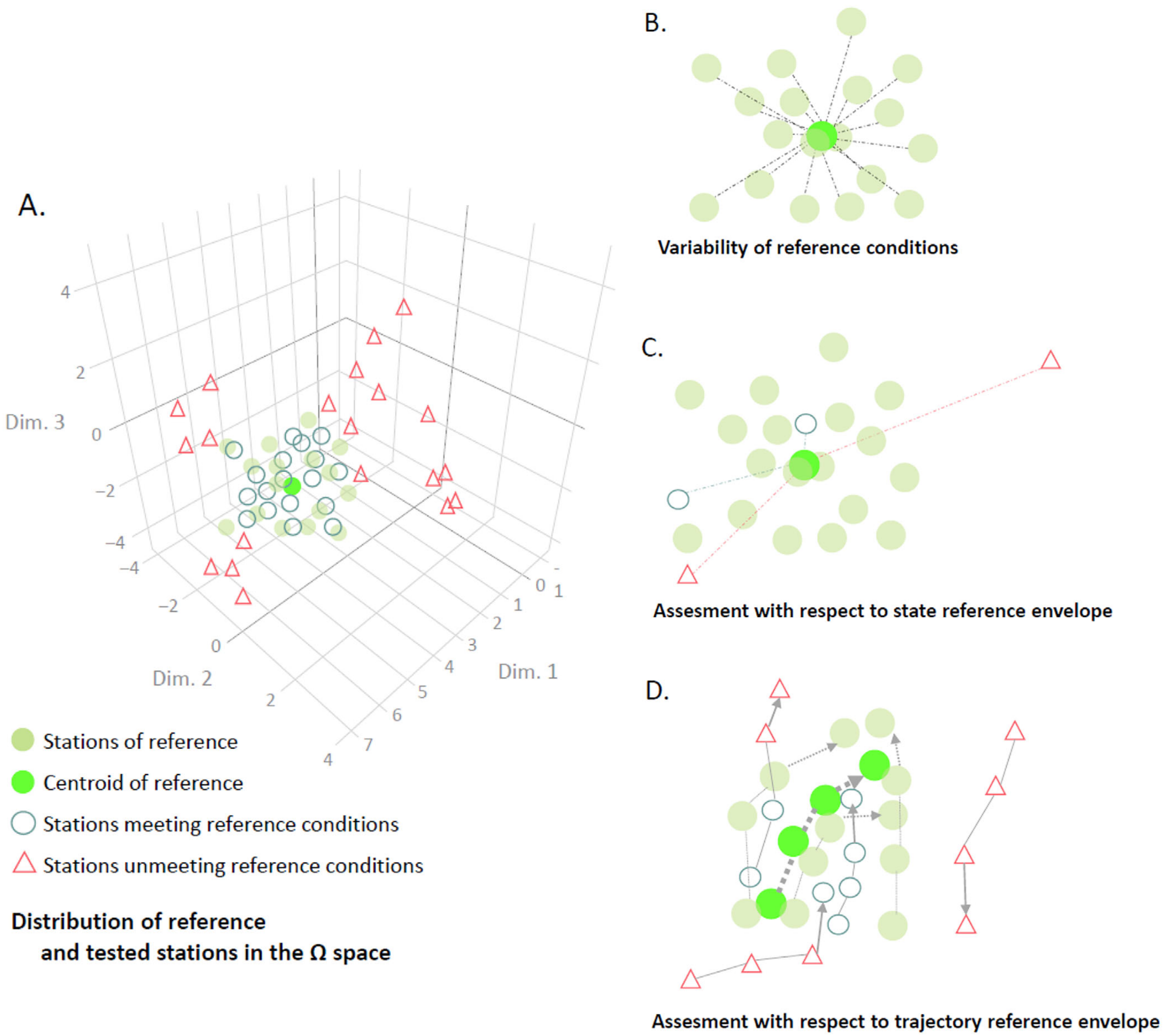

FIGURE 1 Ecological quality assessment. (A) Distribution of stations of reference and their centroid within the reference envelope (E) defined in the three first dimensions (Dim.) of the Ω space and distribution of assessed stations with respect to E. (B) Measure of Var(E) (i.e., spatial and/or temporal variability) of the reference envelop in Ω. (C) State-based ecological quality assessment in Ω. (D) Trajectory-based ecological quality assessment. Note that the trajectory-based approach implies a new space of resemblance Ω_T defined by dissimilarities between trajectories.

among sites within a geographical area of interest (Legendre et al., 2005; Legendre & De Cáceres, 2013; Whittaker, 1960). Note, however, that Var(E) can represent more generally spatiotemporal variability in the reference conditions. Using the state reference envelope, the ecological quality of assessed ecosystems can be assessed by comparing their coordinates with respect to the states conforming the reference envelope (Figure 1). However, it does not formally integrate the dynamics at the scale of reference stations. In other words, one is assuming that temporal and spatial replicates are equivalent with respect to the definition of the envelope.

Trajectory reference envelope

Whenever it is relevant to focus the EQA on the intrinsic dynamics of a set of reference stations, the reference envelope (E) will be defined in terms of their natural ecological dynamics (i.e., ecological dynamic regime; Sánchez-Pinillos et al., 2023; Figure 1D). Following the framework of De Cáceres et al. (2019) we define ecological trajectory in space Ω as the sequence $T = \{(x_1, t_1), ..., (x_n, t_n)\}$, where n is the size of the trajectory. Let now $\{T_1, T_2, ..., T_r\}$ be the set of r ecological trajectories conforming the trajectory reference envelope to which particular ecosystems are to be compared, and $D(T_i, T_j)$ be the dissimilarity between trajectories T_i and T_j in Ω_T, calculated from d values as explained in De Cáceres et al. (2019). The variability of ecosystem dynamics (or dynamic beta diversity) of the trajectory reference envelope is now (De Cáceres et al., 2019; Sánchez-Pinillos et al., 2023):

$$\mathrm{Var}(E) = \frac{\sum_{i=1}^{r-1} \sum_{j=i+1}^{r} D\left(T_i, T_j\right)^2}{r \times (r-1)}. \qquad (2)$$

With the trajectory envelope, the ecological quality of the assessed ecosystems can be tested by comparing its coordinates with respect to the states conforming the trajectories of the envelope. More interestingly, one can compare the temporal dynamics (i.e., the trajectories) of the assessed ecosystems to the dynamics observed in the reference envelope. Finally, note that the trajectory reference envelope can be considered as a generalization of the state reference envelope if $D(T_i, T_j) = d(x_i, x_j)$ for trajectories having a single observation.

THE DISTANCES BETWEEN ASSESSED ECOSYSTEMS AND REFERENCE ENVELOPES

Distance between an ecological state and a set of reference ecological states

Let y be the ecological state in Ω of the ecosystem whose quality is to be tested against a reference envelope E (Figure 1C). With the state reference envelope, the distance between the assessed state y and that of each reference station i is given by $d(y, x_i)$, and the distance from an assessed state to the reference envelope is calculated using (Legendre & De Cáceres, 2013):

$$D(y,E)^2 = \frac{1}{r} \times \sum_{i=1}^{i=r} d(y,x_i)^2 - \frac{1}{2} \times \mathrm{Var}(E), \qquad (3)$$

where $\mathrm{Var}(E)$ has been estimated using Equation (1).

Distance between an ecological state and a set of reference trajectories

When using a trajectory reference envelope, the distance from the assessed state y to any reference trajectory T_i is naturally defined as the minimum distance between y and the various segments of T in Ω (Besse et al., 2016; De Cáceres et al., 2019). Knowing the distance of the community state to all r reference trajectories and the dynamic variability of the reference envelope, $\mathrm{Var}(E)$, one can estimate the distance to the centroid of the reference envelope using an equation analogous to Equation (3):

$$D(y,E)^2 = \frac{1}{r} \times \sum_{i=1}^{i=r} D(y,T_i)^2 - \frac{1}{2} \times \mathrm{Var}(E). \qquad (4)$$

Recovering or departing from the reference envelope

Let o_1 and o_2 be two consecutive observations of the assessed ecosystem, whose corresponding states are y_1 and y_2. The assessed ecosystem will be recovering (i.e., decreasing distance from the reference envelope) if $D(y_1, E) > D(y_2, E)$ and departing (i.e., increasing distance from the reference envelope) from the reference envelope if $D(y_1, E) < D(y_2, E)$ (Sturbois, Cucherousset, et al., 2021; Sturbois, De Cáceres, et al., 2021). Since departing or recovering patterns depends on $D(y_1, E)$ and $D(y_2, E)$, the same concepts apply to both the state and trajectory reference envelope, depending on whether these distances are calculated using Equation (3) or (4).

Distance between an assessed trajectory and a reference envelope

The assessed ecosystem may have been sampled several times. Let now $T_y = \{(y_1, t_1), \ldots, (y_n, t_n)\}$ be the trajectory of our assessed ecosystem in Ω. We can estimate the distance between T_y and each of the trajectories of a dynamic reference envelope E, and estimate the distance between the dynamics of the assessed ecosystem and E in Ω_T analogously (Figure 1D):

$$D\left(T_y,E\right)^2 = \frac{1}{r} \times \sum_{i=1}^{i=r} D\left(T_y,T_i\right)^2 - \frac{1}{2} \times \mathrm{Var}(E). \qquad (5)$$

Different distance measures could be used to compare trajectories, based on d values (De Cáceres et al., 2019, for a discussion).

ECOLOGICAL QUALITY ASSESSMENT

The quality of the assessed ecosystem with respect to a state or trajectory reference envelope can be assessed by comparing $D(y, E)^2$ (from Equation 3 or 4) in relationship to $\mathrm{Var}(E)$ (from Equation 1 or 2, respectively). This can be done using a fuzzy typicality function, taken from the possibilistic C-means clustering algorithm (Krishnapuram & Keller, 1993):

$$u(y, E) = \frac{1}{1 + \left(\frac{D(y,E)^2}{\mathrm{Var}(E)}\right)^{1/(m-1)}}. \quad (6)$$

The *typicality* function u is bounded between 0 and 1. All assessed ecosystems whose squared distance to the centroid of the reference envelope is smaller than Var(E) will obtain a membership value higher than 0.5 according to the possibilistic C-means clustering algorithm (Krishnapuram & Keller, 1993). The fuzzy exponent $m \in (1, \infty)$ should be decided conventionally and modulates the shape of the typicality function, with values closer to 1 leading to a steepest change around Var (E). The smaller the value of m, the closer to a hard partition will be the result. If m is set too high and the data are noisy, the resulting u values may be completely fuzzy and therefore uninformative.

$u(y, E) = 0.5$ whenever $D(y, E)^2 = \mathrm{Var}(E)$ meaning that the squared distance between the assessed ecosystem and (the centroid of) the reference envelope is equal to the average, across reference states, of the same squared distance. In the EQA framework, we assume that the assessed ecosystem is considered completely included in the envelope of reference whenever $D(y, E)^2 \leqq \mathrm{Var}(E)$ which should lead to an EQA that should be equal to one. Consequently, we define a *quality function* Q as a simple transformation of u, as follows:

$$Q(y, E) = \min(1, 2 \times u(y, E)). \quad (7)$$

Like u, the *quality* function Q is also bounded between 0 and 1, which facilitates its interpretation. By definition, $Q(y, E) = 1.0$ whenever $D(y, E)^2 \leqq \mathrm{Var}(E)$, that is, whenever the squared distance between the tested ecosystem and the reference envelope is *lower than* or *equal to* the corresponding squared distance average across reference states. In other words, the tested ecosystem is, on average, as far from the envelope's centroid as reference states and hence, it can be considered as completely included in the envelope (i.e., $Q(y, E) = 1.0$). If $1 > Q \geqq 0.5$, the tested ecosystem can still be qualitatively included within the reference envelope (Krishnapuram & Keller, 1993), whereas if $Q < 0.5$, the tested ecosystem will be considered outside (Figure 1A,C,D). Like u, the steepness of the Q decline for larger $D(y, E)$ values will depend on the fuzzy exponent m. Note that the quality function Q can be also straightforwardly assessed for a trajectory T_y of the target ecosystem, by simply replacing $D(y, E)$ with $D(T_y, E)$ in Equation (8):

$$u(T_y, E) = \frac{1}{1 + \left(\frac{D(T_y, E)^2}{\mathrm{Var}(E)}\right)^{1/(m-1)}}, \quad (8)$$

and, analogously, in Equation (9):

$$Q(T_y, E) = \min(1, 2 \times u(T_y, E)). \quad (9)$$

When relevant, the framework allows performing EQA with respect to different reference envelopes in a single Ω space. This flexibility would be particularly useful in uncertain situations (e.g., diverging species, different potential endpoints). In this case, users can state that the tested ecosystems belong to the reference envelope with the higher value of Q.

EXTENDING THE ECOTRAJ SOFTWARE

The R package "ecotraj" (De Cáceres et al., 2019; Sturbois, Cucherousset, et al., 2021; Sturbois, De Cáceres, et al., 2021) assists ecologists in the analysis of ecosystems' temporal changes, defined as trajectories within a chosen multivariate space. It includes functions to perform trajectory plots and to calculate a set of distance and direction-based metrics (length, directionality, and angles) as well as metrics to relate pairs of trajectories (dissimilarity and convergence). New "ecotraj" functions are available on CRAN and GitHub repositories (https://zenodo.org/records/10053448) to perform state- and trajectory-based EQA, including the calculations of the variability of reference envelopes, the distance between tested ecosystems and reference envelope, and the calculation of the ecological quality. The data set and R codes used for this article are included in the R package "ecotraj" and associated documentation.

ECOLOGICAL APPLICATIONS

Mapping conservation status of vegetation habitat

Context

Ecological succession is the process by which species and habitat change over time in an area. Gradually, communities replace one another until a stable "climax community" is reached (e.g., mature forest) or until a disturbance occurs. The maintenance of cultural landscapes associated with a long human history consequently depends on the use made of them and the conservation of vegetated habitat at some chosen ecological states. This often requires restoration operations to limit their evolution (e.g., reaping or grazing). In heathlands, notably characterized by the dominance of Ericoid low shrubs, the cessation of agricultural activities leads to shrub and tree encroachment at varying rates, which affects their conservation status. In this context, managers need tools to (1) assess the conservation status of such habitat with respect to reference conditions defined as conservation targets, and (2) follow their dynamics to plan restoration operations.

Methods

The nature reserve of Landes et Marais de Glomel (Brittany, France) is composed of temperate Atlantic wet heaths whose reference state is commonly considered dominated by plant communities associated with acid, nutrient-poor soils that are at least seasonally waterlogged and dominated by *Erica tetralix* and *E. ciliaris*. This habitat is considered of community interest as part of the European directive on the conservation of natural habitats and of wild fauna and flora (97/62/CEE). This requires their maintenance or recovering into a good conservation status. As this conservation target belongs to intermediate states of the vegetation serial, it does not correspond to the final climax forest state. Managers of the nature reserve consequently lead restoration actions mainly through grazing and reaping to maintain this intermediate state. They need a tool to assess the efficiency of these operations.

The reference envelope, defined as the objective of conservation to be achieved with the managers of the nature reserve, was quantitatively identified on the field and characterized by vegetation surveys at the scale of homogeneous areas (abundance–dominance coefficient expressed in percentages for a 4-m^2 quadrat; Braun-Blanquet et al., 1932). For the definition of the reference envelope, surveys integrated the variability of reference states from youngest to early senescent stages, based on expert assessment with respect to the European Directive (97/62/CEE) requirements (Figure 2A).

Surveys were simultaneously performed at five stations used to define the reference envelope, and 18 stations for which the conservation status was to be assessed. A distance matrix was computed using the Bray–Curtis coefficient of dissimilarity (Ω space) and used as input of a state-based EQA ($Q \geq 0.5$: inside the reference envelope; $Q < 0.5$: outside the reference envelope). The squared distance to the centroid of the reference state envelope was calculated for each assessed station. A fuzzy exponent value equal to 1.7 was used, as managers do not consider the sporadic presence of woody vegetation as a degradation, a situation representing pragmatically the limit of the reference envelope along the senescence gradient that was not optimally expressed by reference stations. Ecological states of reference and tested stations were represented in the two first dimensions of a principal coordinates analysis (PCoA) performed on the Bray–Curtis dissimilarity matrix.

Results

The mean squared distance dissimilarity of reference stations to the centroid of reference, 0.06 ± 0.02 illustrated the spatial variability of the reference envelope in the Ω space, with a squared distance to the centroid of the reference envelope ranging from 0.04 (station 1) to 0.09 (stations 3 and 6). Among the 18 tested stations, 9 were included in the reference envelope ($Q \geq 0.5$; mean squared distance $\pm$ SD $= 0.08 \pm 0.18$), while 9 of them, distant from the reference envelope ($Q < 0.5$; 0.24 ± 0.12), were outside (Table 1, Figure 3). Stations meeting reference conditions were typified by higher characteristic species abundance–dominance values (*Erica tetralix*, *Calluna vulgaris*, *Ulex gallii*, and *Erica ciliaris*; Appendix S1, Figure S1). The first axis of the PCoA (27.7% of the data set inertia) separated stations characterized by a higher distance to the centroid of the reference envelope (Figure 2B). Some stations (e.g., 10, 15, and 16) that did not meet reference conditions were located close to the limit of the reference envelop in the Ω space due to the presence of woody vegetation indicating a most advanced successional process. Among the stations not meeting reference conditions, five of them were characterized by the highest distances to the centroid of the reference envelope (stations 11, 18, 14, 17, and 23) in relation with the dynamic of species responsible for shrub and tree encroachment (e.g., *Rhamnus frangula*, *Betula pubescens*, *Pteridium aquilinum*, *Quercus robur*, *Genista anglica*), or overgrazing (station 13) (Figure 2B; Appendix S1: Figure S1).

Discussion

This application based on vegetation habitat in a restoration context illustrates the interest of the EQA framework for the analysis of conservation status. The calculation of

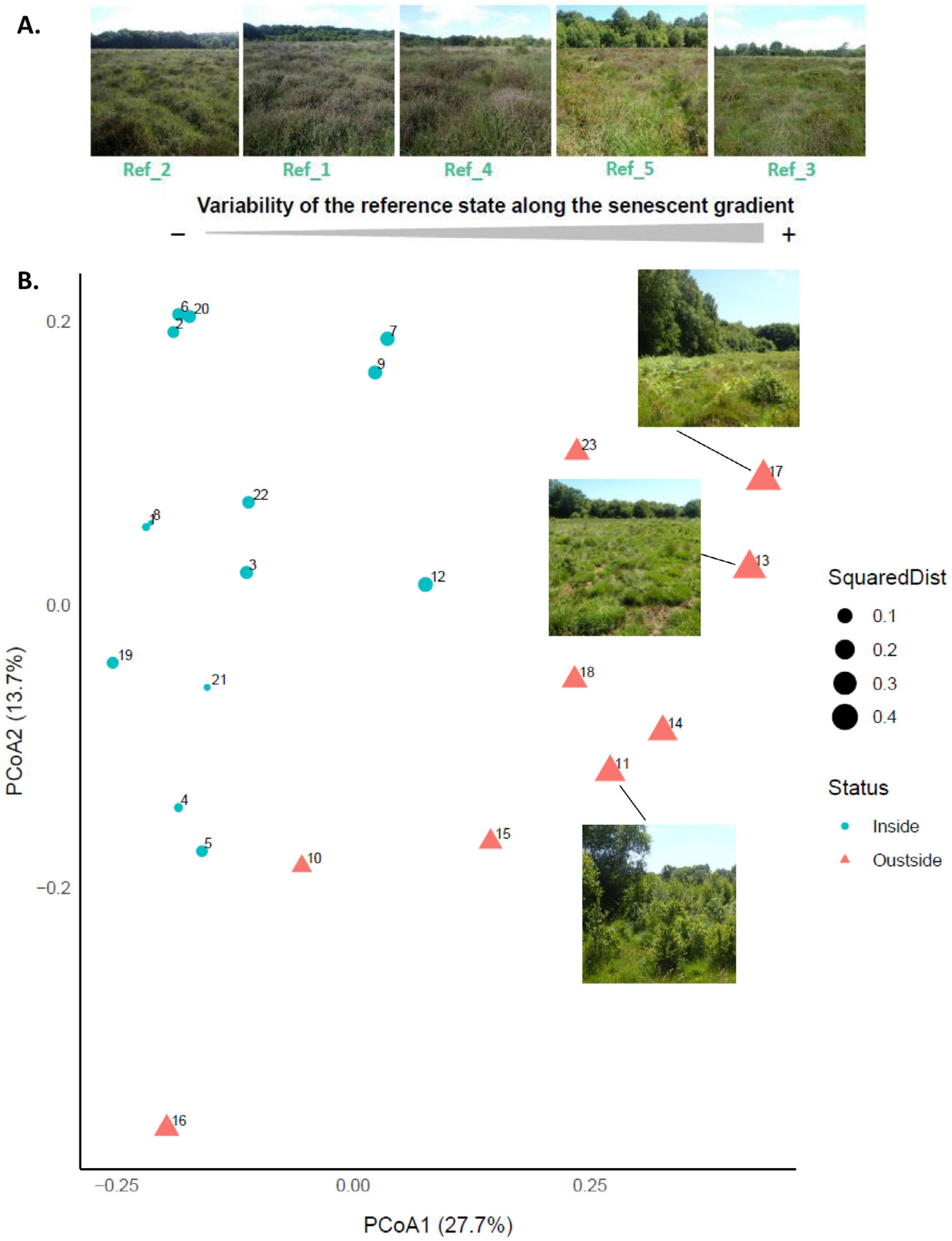

FIGURE 2 Conservation state assessment of wet heathlands in the nature reserve of the marsh and heathlands of Glomel as part of the European directive on the conservation of natural habitats and of wild fauna and flora (97/62/CEE). (A) Variability of data-driven reference conditions, defined as the conservation objective to be achieved by managers of the nature reserve, with respect to the gradient of senescence. (B) Representation of the results in the two first dimensions of the principal coordinate analysis (PCoA). Only the two first PCoA dimensions are shown but ecological quality assessment integrated all the Ω space dimensions. Green circles indicate stations that meet reference conditions ($Q \geqq 0.5$) while stations that do not meet the characteristics of the reference conditions are represented in red triangles ($Q < 0.5$). The size of symbols is proportional to the squared distance to the centroid of the reference envelope (SquaredDist). Points numbered from 1 to 5 were used to define the reference envelope, while points from 6 to 23 were those whose conservation status was to be assessed (Table 1). Photo credit: Anthony Sturbois.

TABLE 1 Results of the state-based ecological quality assessment of heathlands from the Glomel nature reserve including squared distance (SquaredDist) and *Q* values, and corresponding status.

Observations	SquaredDist	*Q*	Status
1	0.04	1.00	Reference
2	0.07	0.83	Reference
3	0.09	0.68	Reference
4	0.04	1.00	Reference
5	0.07	0.87	Reference
6	0.09	0.69	Inside
7	0.11	0.57	Inside
8	0.03	1.00	Inside
9	0.10	0.59	Inside
10	0.13	0.48	Outside
11	0.32	0.15	Outside
12	0.12	0.53	Inside
13	0.39	0.12	Outside
14	0.30	0.17	Outside
15	0.17	0.33	Outside
16	0.20	0.28	Outside
17	0.46	0.10	Outside
18	0.21	0.26	Outside
19	0.07	0.83	Inside
20	0.09	0.69	Inside
21	0.03	1.00	Inside
22	0.08	0.75	Inside
23	0.21	0.26	Outside

Q points out stations where restoration actions are necessary, and the squared distance to the centroid of the reference envelope provides an additional precision that could be relevant for managers when planning restoration actions (e.g., years of reaping operations, adaption of the grazing gradient). Ideally, such approach must be performed at the scale of habitat polygon characterized by homogeneous vegetation rather than at the scale of land parcel potentially characterized by different dynamics and state of conservations (e.g., stations 4 and 10). When resurveying, heathland managers would be able to (1) measure accurately the restoration or degradation dynamic (including trajectory speed) of heathlands, (2) compare different restoration methods, and (3) consequently the effectiveness of management operations. Depending on management ambitions (strictness of conservation goals), the severity degree of degradation process along a gradient, new knowledge, and/or human and financial means, managers could also adapt the value of the fuzzy component *m*.

Impact of fishing activities on marine habitats

Context

Assessment of the ecological quality status of marine habitats as requested by the European Water Framework Directive and the European Marine Strategy Framework requires measuring species composition differences between reference conditions and tested stations in different natural and anthropogenic contexts. Maerl beds refer to free-living coralline algae (*Corallinophycidae*, *Rhodophyta*) accumulating on soft sediment to form highly structured and productive biogenic benthic habitats. Considering their ecological value and multiple threats (e.g., extraction, eutrophication, fishing, climate change), maerl beds are under national, European, and international conservation legislation, which imposes the assessment of their ecological quality status.

Methods

We used an experimental data set built by Tauran et al. (2020) to study the impact of fishing dredges and varying fishing pressures on maerl beds, in the bay of Brest (Brittany, France). The study follows a before-after-control-impact design (Stewart-Oaten et al., 1986). Briefly, three control stations were surveyed and compared with five treatment stations characterized by different fishing dredges and pressure levels (Appendix S2: Figure S1; Tauran et al., 2020): (1) a clam dredge (CD), 70–90 kg, 1.5-m wide, 40 teeth of 11 cm each; (2) a queen scallop dredge (QSD), 120 kg, 1.8-m wide, with a blade; and (3) a king scallop dredge (KSD), 190 kg, 1.8-m wide, 18 teeth of 10 cm each every 9 cm. CTRL stations were used to define the reference envelope. The main aim of the BACI design was to study the impact of different dredging treatments at a given time. The study area was not free from other sources of natural and anthropogenic variability implying that reference conditions do not describe pristine habitats but were defined with the best available contemporary data in the sense of McNellie et al. (2020) or Coates et al. (2018).

Pressure levels were measured as the number of dredge tows performed on the zone during the experimental dredging session: 0 (i.e., control), 10, or 30 dredge tows. Samples were collected from April 2016 to April 2017. Session 1 was sampled just before the experimental dredging (t_0); session 2 at $t_0 + 1$ week; session 3 at $t_0 + 1$ month; and session 4 at $t_0 + 12$ months. Nine replicates were sampled for all treatments and sessions with a Smith–McIntyre grab (0.1 m^2). More information

FIGURE 3 Map of the conservation status assessment of the wet heathlands in the nature reserve of the marsh and heathlands of Glomel as part of the European directive on the conservation of natural habitats and of wild fauna and flora (97/62/CEE). Green circles indicate stations that meet reference conditions (i.e., inside $Q \geqq 0.5$) while stations that unmeet reference conditions are represented in red triangles (i.e., outside, $Q < 0.5$). The size of symbols is proportional to the squared distance (i.e., distance to the centroid of the reference envelope).

on laboratory processes can be found in Tauran et al. (2020).

Abundance was pooled at the treatment levels (i.e., CTRL1, CTRL2, CTRL3, KSD_10, CD_10, CD_30, QSD_10, and QSD_30) and log-transformed as initially performed in Tauran et al. (2020). A Bray–Curtis matrix of dissimilarity was computed with the resulting data set, including raw data for 250 species at 32 observations (i.e., treatments × sessions) and used for both state- and trajectory-based EQAs. Results were represented in the two first dimensions of a PCoA. For the state-based approach, all CTRL samples as well as those collected at t_0 before the experimental dredging at tested stations were used together to define the state reference envelope (i.e., composed of 16 reference ecological states) compared to remaining observations ($n = 16$). For the trajectory-based approach, trajectories of the three CTRL stations ($n = 3$) were used to define the dynamic reference envelope compared with the trajectories of the five experimental treatments. For both approaches, the ecological quality of remaining stations was assessed with respect to the reference envelope by the calculation of Q and the squared distance to the centroid of the reference envelopes.

Results

Observations performed before the dredging experiment were characterized by low Bray–Curtis dissimilarity values and were well grouped in the two first dimensions of the PCoA, indicating that conditions at the beginning of the experiment were similar at control and assessed stations (Figures 4 and 5A). All stations mainly exhibited departing pattern during the experiment from session 2 (one week later) to session 4 (one year later): natural dynamics for control stations and natural as well as dredging dynamics for tested stations. The spatiotemporal variability of the squared distance to the centroid of the reference envelope was lower at control stations (0.029 ± 0.005) than at tested stations (0.044 ± 0.013) indicating that the dredging experiment induced a higher variation in community composition than natural dynamics (Figure 5B). The dissimilarity between control and tested stations globally increased at sessions 2 and 3 (Figure 4) in relation with a decline in density and species richness. While at session 2, treatment characterized by lower pressure levels (i.e., CD_10, QSD_10, and KSD_10) was still included in the reference envelope, CD_30 and QSD_30 fell outside of the reference envelope, which pointed a higher and quicker impact for higher level of pressure treatment (Table 2, Figure 5).

At session 3, all assessed stations were excluded from the reference envelope and CD_30 (0.063), QSD_10 (0.073), and QSD_30 (0.064) were characterized by the highest squared distance to the centroid of the reference

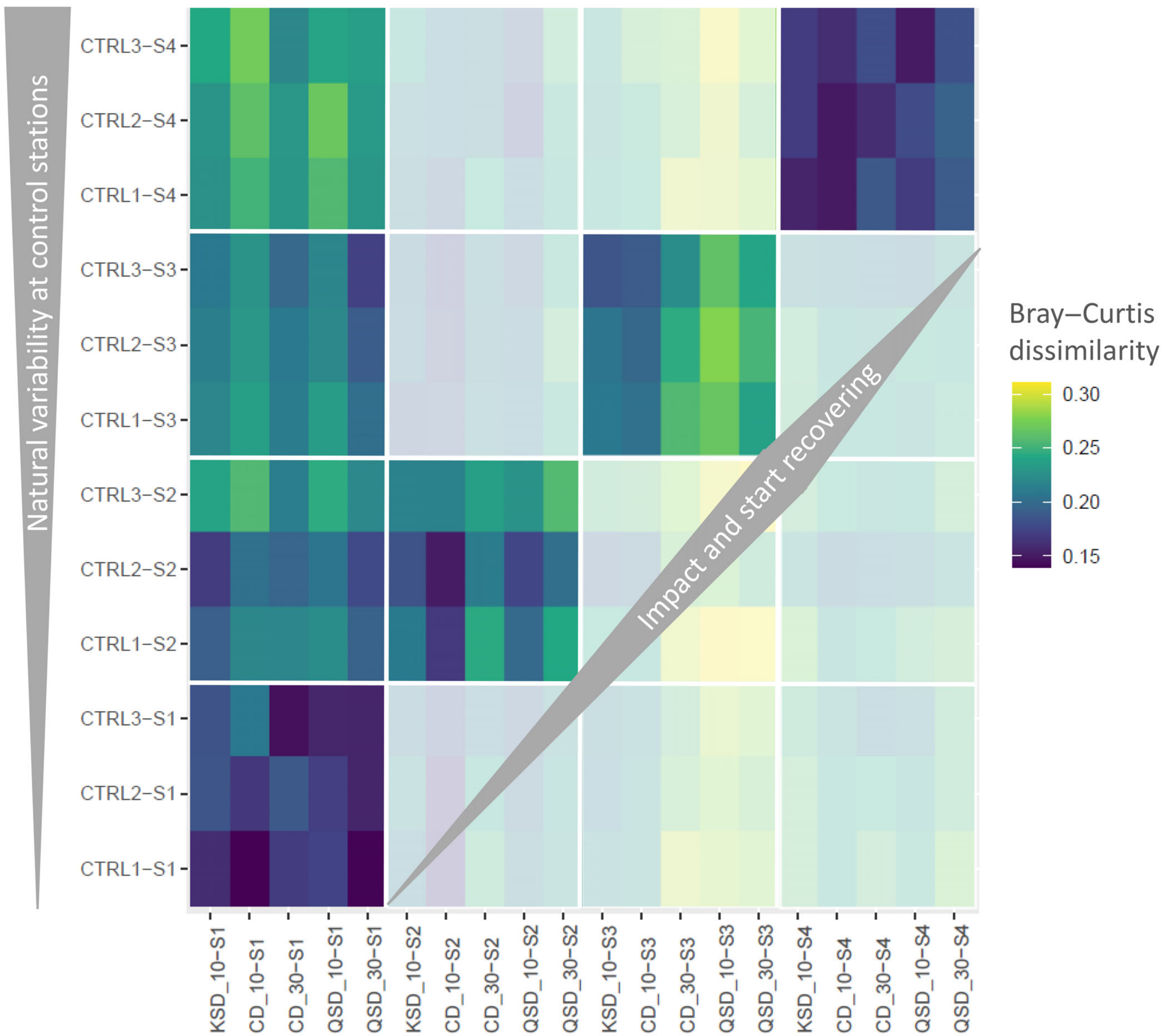

FIGURE 4 Heat map of the Bray–Curtis dissimilarity matrix between control (row) and tested stations (column). Colors correspond to Bray–Curtis dissimilarity values. Some parts appear in transparency to focus the heatmap on natural variability at control stations and impact and start recovering processes at dredged stations.

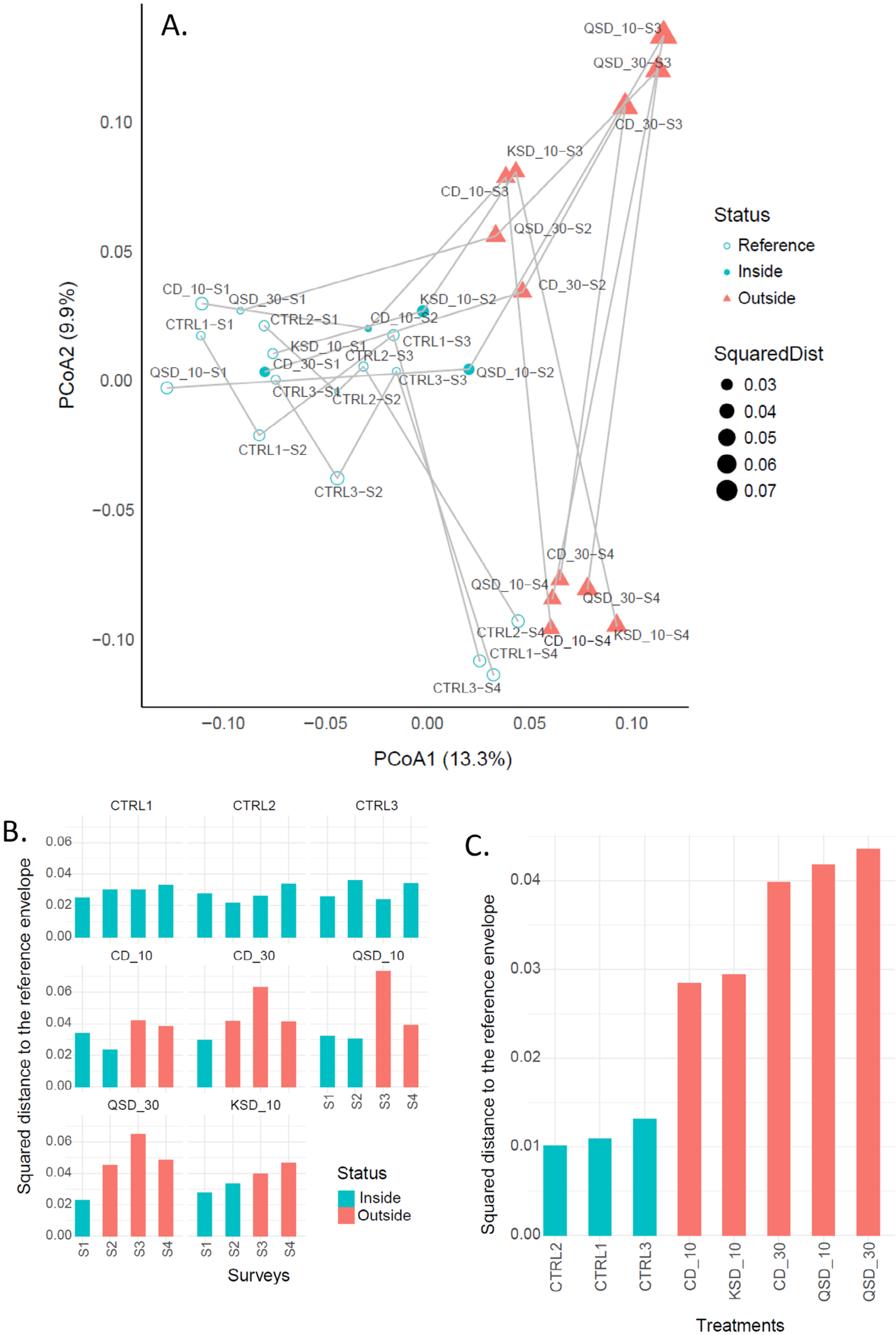

FIGURE 5 Legend on next page.

envelope. During this session, impact of dredging compared with the reference envelope was maximal, except for the KSD_10 treatment.

Although the dissimilarity between reference and tested observations decreased at session 4 (Figures 4 and 5), tested stations were still located outside of the reference envelope, indicating an incomplete recovering dynamics 12 months after dredging for all treatments. The trajectory-based analysis allowed a further integration of dynamics in the quality assessment process by considering whole trajectories at both control and tested stations. All trajectories of tested stations were located outside of the envelope of reference defined by natural trajectories at control stations within the trajectory reference envelope (Table 2). Compared to the natural trajectories, experimental dredging CD_30 (0.040, squared distance to the centroid of the reference envelope), QSD_10 (0.042), and QSD_30 (0.044) implied a higher deviation, than treatments CD_10 (0.028) and KSD_10 (0.029).

Discussion

Using univariate and multivariate analyses and a weighted linear mixed-effect model combined with photographic monitoring, Tauran et al. (2020) pointed (1) that the consequences of dredging were taxonomically and structurally visible from session 2, (2) ongoing taxonomic impacts at the end of the experiment, and (3) effects of dredge types and pressure levels. Our results were in accordance with this study, which aimed at describing the response of maerl habitat to different dredging pressures. EQA brings interesting additional and synthetic conclusions by making the spatiotemporal variability of the reference envelope central in the analysis in order to disentangle natural and dredging-induced dynamics (i.e., $\mathrm{Var}(E_{\mathrm{CTRL}}) < \mathrm{Var}(E_{\mathrm{tested}})$). With the state-based EQA, we were able to distinguish accurately when stations fell outside of the reference envelope pointing in the same time differences in response of maerl habitat immediately and one month after the dredging experiment, CD_30 and QSD_10&30 being the most damaging treatments. We also captured the early recovery for tested stations and were able to measure accurately the recovering gradient among treatments with the calculation of Q and the squared distance to the centroid of reference envelope. Furthermore, the trajectory-based EQA allowed pointing trajectories that deviated the most from the seasonal dynamics exhibited by control stations within the trajectory reference envelope, CD_10, and KSD_10 exhibiting more similar trajectories with control stations than other treatments.

GENERAL DISCUSSION

Assumptions and limitations of the proposed framework

EQA provides a simple, yet quantitative, assessment of the ecological quality of tested ecosystems and its temporal dynamics. As most multivariate ecological methods, which are often descriptive by nature, EQA as well as ETA suffer the same limitations. Future users should consequently consider EQA outputs together with a strong examination of ecological attributes used to define the Ω space (i.e., species or any other variables). Choosing the appropriate attributes constitutes an essential step in order to obtain meaningful EQA outputs. Otherwise, conclusions of the analysis may provide an incomplete picture and can mislead the description of ecological processes with potential misdirecting conservation actions (Jetz et al., 2019; Pereira et al., 2013; Schmeller et al., 2018; Turak et al., 2017). When necessary, additional analyses providing statistical background on ecosystem quality can be used (Buckley et al., 2021), as well as bootstrapping the reference sample to generate CIs (see function arguments in the R package "ecotraj"). The analysis of the distribution of distances to the centroid among reference samples using the quantiles (i.e., Q90, Q95, Q99) would also allow a probabilistic

FIGURE 5 State-based ecological quality assessment of maerl benthic habitats under dredging pressures. Dredge type: king scallop dredge (KSD), clam dredge (CD), queen scallop dredge (QSD) control stations with no dredging (CTRL). Pressure levels at dredged stations: 10 and 30 dredge tows. Samples were collected from April 2016 to April 2017. Session 1 (S1) was sampled just before the experimental dredging (t_0) and used to define the state reference envelope; session 2 (S2) at t_0 + 1 week; session 3 (S3) at t_0 + 1 month; and session 4 (S4) at t_0 + 12 months. (A) Representation of the results in the two first dimensions of the principal coordinate analysis (PCoA). Only the two first PCoA dimensions are shown but EQA integrated all the Ω space dimensions. The shape of symbols corresponds to ecological quality (reference; inside the reference envelope $Q \geqq 0.5$; stations outside the reference envelope $Q < 0.5$) and the size is proportional to the squared distance to the centroid of the reference envelope (SquaredDist). (B) Bar plot showing the evolution of the squared distance to the centroid of the state reference envelope during the experiment at control and tested stations from session 1 to session 4. (C) Bar plot showing the squared distance to the centroid of the trajectory reference envelope.

TABLE 2 Results of the state- and trajectory-based ecological quality assessment of maerl habitat under different dredging pressures including squared distance (SquaredDist) and *Q* values, and corresponding status.

State-based EQA				Trajectory-based EQA			
Observations	**SquaredDist**	***Q***	**Status**	**Treatments**	**SquaredDist**	***Q***	**Status**
CTRL1-S1	0.025	0.837	Reference	CTRL1	0.011	1.000	Reference
CTRL1-S2	0.030	0.657	Reference				
CTRL1-S3	0.030	0.658	Reference				
CTRL1-S4	0.033	0.576	Reference				
CTRL2-S1	0.028	0.731	Reference	CTRL2	0.010	1.000	Reference
CTRL2-S2	0.022	0.971	Reference				
CTRL2-S3	0.026	0.792	Reference				
CTRL2-S4	0.034	0.560	Reference				
CTRL3-S1	0.026	0.805	Reference	CTRL3	0.013	1.000	Reference
CTRL3-S2	0.036	0.506	Reference				
CTRL3-S3	0.024	0.872	Reference				
CTRL3-S4	0.034	0.555	Reference				
KSD_10-S1	0.027	0.742	Reference	KSD_10	0.029	0.356	Outside
KSD_10-S2	0.033	0.576	Inside				
KSD_10-S3	0.040	0.440	Outside				
KSD_10-S4	0.046	0.341	Outside				
CD_10-S1	0.034	0.559	Reference	CD_10	0.028	0.377	Outside
CD_10-S2	0.023	0.903	Inside				
CD_10-S3	0.042	0.404	Outside				
CD_10-S4	0.038	0.468	Outside				
CD_30-S1	0.029	0.676	Inside	CD_30	0.040	0.211	Outside
CD_30-S2	0.042	0.408	Outside				
CD_30-S3	0.063	0.199	Outside				
CD_30-S4	0.041	0.412	Outside				
QSD_10-S1	0.032	0.598	Reference	QSD_10	0.042	0.194	Outside
QSD_10-S2	0.031	0.643	Inside				
QSD_10-S3	0.073	0.151	Outside				
QSD_10-S4	0.039	0.453	Outside				
QSD_30-S1	0.023	0.925	Reference	QSD_30	0.044	0.179	Outside
QSD_30-S2	0.045	0.358	Outside				
QSD_30-S3	0.065	0.191	Outside				
QSD_30-S4	0.048	0.316	Outside				

Note: Dredge type: king scallop dredge (KSD), clam dredge (CD), queen scallop dredge (QSD) control stations with no dredging (CTRL). Pressure levels at dredged stations: 10 and 30 dredge tows. Samples were collected from April 2016 to April 2017. Session 1 (S1) was sampled just before the experimental dredging (t_0) and used to define the state reference envelope; session 2 (S2) at t_0 + 1 week; session 3 (S3) at t_0 + 1 month; and session 4 (S4) at t_0 + 12 months. The trajectory reference envelope was defined by trajectory at control stations.

interpretation (i.e., if $D(Y, E) > Q90$, then it is farther from the centroid than the 10% farthest stations in the reference).

Note that ordination spaces are specifically constructed for each given data set. Therefore, any data transformation on the raw data or sampling decision is likely to affect the space of resemblance, and subsequently, all EQA metrics to be calculated. We alert future users and urge them to test for these effects before any overall transformations, change in sampling design, and/or suppression of rare species in a community data set. Furthermore, when choosing a dissimilarity coefficient, users should check the properties the coefficient has, to determine whether they are suitable for the objectives of the study (Anderson et al., 2011; Koleff et al., 2003; Legendre & De Cáceres, 2013) and implications in EQA performing.

All observations influence the properties of the multivariate space of resemblance used as input for EQA. Consequently, the construction of Ω should integrate observations belonging to similar ecological entities. For instance, if the reference envelope characterized a specific ecological state along a particular vegetation serial, only surveys from this serial, which are characterized by a real reachability with respect to the reference envelope, should be used to define Ω. For instance, in the Glomel application, we perform EQA for wet heathlands and it would not be relevant to add surveys from dry heathlands to define the space of resemblance. Overall, our framework claims for strong ecosystems knowledge for the definition of reference conditions and EQA analyses. Similarly, in the second ecological applications about dredging impact on maerl benthic communities, dealing with another habitat type (e.g., fine sands) implying different species pools must impose performing EQA in an additional taxonomic space of analysis. Then qualitative EQA outputs can be compared (i.e., inclusion or exclusion of respective reference envelope, recovering, or departing patterns) but not quantitative outputs.

Rerunning EQA does not change the results for newly tested ecosystems, as long as the reference states/trajectories remain the same and the choices underlying Ω do not change. Of course, the addition of new reference ecosystems will change the value of Var(E) and consequently and potentially the results. On longer time scales and facing shifting baseline effects, users should decide whether states used to define the reference envelope are still reachable or not and decide to define appropriately the reference envelope.

EQA allows quantifying the departure from (or recovery of) reference conditions, but does not inform on what restoration measures should be applied. For instance, in the first application, stations 11 and 13 were characterized by similar squared distance to the centroid of the reference envelope values but contrasted ecological processes in relation with different restoration techniques. In response, restoration strategies should be adapted: for example, reaping operations with an exportation of materials for station 11 versus decrease of the grazing pressure for station 13.

On the definition of reference envelopes

The EQA framework is based on a data-driven definition of reference conditions used to compute a reference envelope, from a set of observations chosen by the user, to be compared with observations to be assessed in a multivariate space of resemblance. In this sense, our approach meets the criteria of Hiddink et al. (2023) who privileged methods using the range of natural variation as reference conditions for ecosystem state assessment. Using EQA implies that users accept the concept of reference conditions responsible of its underlying assumptions, have a sufficient knowledge of the ecosystem considered, and that such conditions can be quantitatively defined.

The concept of reference conditions is debated in ecology (Corlett, 2016; Hobbs et al., 2014; Hughes et al., 2017; McNellie et al., 2020). The respective intrinsic properties of the different fields of ecology fuel such debates and influence the degree of accuracy and maturity by which different scientists deal with the concept of reference conditions. For instance, the definition of reference conditions, including dynamics, seems more evident when dealing with local communities composed of few species in a restoration context than in a larger scale setting involving several hundred species under complex natural and anthropogenic dynamics. In this context, detaching the concept of reference conditions from "good" or "bad" subjective judgments and integrating the dynamic dimension in the EQA should probably facilitate a common acceptance. When accepted, the concept of reference conditions implies a quantitative definition, a step that also make debates as there is virtually no relevant benchmark data describing natural standards and alterations of most environmental systems concerned by cumulative impacts (Ellis et al., 2000). Stoddard et al. (2006) proposed different alternatives for the definition of reference conditions to face the uneven availability of baseline data sets, among which: describing conditions at minimally or least-disturbed sites (in the sense of the contemporary reference state; McNellie et al., 2020), interpreting historical conditions, extrapolating empirical models. The concept of reference condition also fits with experimental studies. Experimental designs allow a more intuitive definition of reference conditions, as shown in the second ecological application under before-after-control-impact design (Stewart-Oaten et al., 1986). In not-managed vegetation applications, reference envelope can also be defined as being observations belonging to natural potential vegetation, even it is not expressed yet on the studied sites.

In dynamic ecosystems or facing the current multiscale changing environment, shifting baselines also influence reference conditions and therefore all the processes of EQA (Thorpe & Stanley, 2011; White & Walker, 1997). Hiers et al. (2012) propose the dynamic reference concept to incorporate the temporal and spatial variation of reference ecosystems such that targets reflect ecological dynamism. Monroe et al. (2022) and Rydgren et al. (2011) also used a dynamic reference approach to model time to ecosystems recovery. EQA provides a more formal, flexible, and explicit multivariate framework, based on two complementary state and trajectory reference envelopes concepts, for the EQA in restoration and conservation contexts of the diverse fields of ecology.

Even if the EQA framework is flexible regarding data input properties, it is necessary to define reference envelopes with a sufficient set of reference observations integrating spatial and/or temporal replications, synchronously sampled with tested stations. Such design will allow taking the best of EQA under state and/or trajectory-based approaches. When performing EQA at large spatial scale, future users should check whether the chosen spatial scale of analysis does not include diverging species pool, which will affect the analysis, regardless of being state-based or trajectory-based. If necessary and relevant, the definition of reference conditions should integrate appropriately those diverging species pools.

Similarly, measurements of restoration success and ecological quality must deal with the rapidly changing no-analogue future (Hiers et al., 2012). For instance, a studied ecosystem can be included in a relevant region located at a moving biogeographic frontier (e.g., along a south/north gradient) in the context of global change. Because of diverging species pools, one can be interested in performing EQA with respect to reference envelopes defined for both biogeographic areas, rather than a global reference envelope. The flexibility of the EQA framework allows such distinction by performing analysis for different chosen "local" reference envelopes within a similar Ω space. In such case, the quality function Q will provide a degree membership for both envelopes which is relevant for uncertain situations. Thus, Q for the conservation target is then the maximum of Q over the different local reference envelopes.

Conservation and research perspectives

During the last century, biodiversity erosion has led to the adoption of conservation policies requiring the reporting of ecosystems ecological status. The EQA framework has been specially designed to support and facilitate such reporting of the quality of ecosystems at the frontier of conservation and science in the different fields of ecology. Other approaches exist for that purpose, but they often require long time series and a vast amount of data (Laurila-Pant et al., 2021; Östman et al., 2020). EQA, although remaining data demanding, seems more flexible and is also developed for multivariate data environments, while existing approaches are mostly univariate so that they ignore a large part of the potential response of the ecosystem to anthropogenic source pressures. By keeping its multivariate nature intact, the EQA framework encompasses a large part of the ecosystem variability and overcomes this issue. The need to keep all the "multivariate" ecosystem variability is in line with the current philosophy in assessing the ecological status (for instance, Labrune et al., 2021).

The ability of EQA to distinguish objectively observations that fall inside or outside reference envelopes makes it relevant to test the behavior of the different available biotic indices, which are more or less potential candidates for the reporting of environmental policies (Teixeira et al., 2016). Taking up the challenge of their respective thresholds would be a direct interesting application (Hiddink et al., 2023). The flexibility of the approach is also inherent to the diversity of variables, which can potentially define multivariate spaces of resemblance. It allows the adaptive application of EQA concepts and metrics to any situation involving reference conditions from individual to community and ecosystems scales: for example, effect of protected areas, impact studies, restoration project, and experimental design. Interestingly, the EQA framework also fits the ecological restoration principles, a promising field of applications to measure the achievement of conservation objectives or compare the effectiveness of different restoration techniques (Saunders et al., 2020): ecological restoration practice is informed by native reference ecosystems; while considering environmental change, ecosystem recovery is assessed against clear goals and objectives; use of measurable indicators (Gann et al., 2019).

The EQA framework also brings relevant insights for multivariate analyses of impact studies under BACI design. Recent methodological innovations have been proposed in response to the call of ecologists for more robust studies on the impact of anthropogenic pressures, conservation actions, or environmental events. Chevalier et al. (2019), Thiault et al. (2017), and Wauchope et al. (2021) recently brought interesting new frameworks or metrics to formally identify and characterize changes through the analysis of data sets collected under BACI designs. To our knowledge, such recent innovations focused on univariate response and the study of multivariate response in impact studies needs to be improved. In this perspective, we demonstrated that the EQA framework perfectly fits to experimental designs and provides valuable multivariate approaches to BACI studies complementary to available univariate frameworks.

Another perspective for future conservation/research applications would be to implement other ETA metrics for EQA. For instance, the use of the trajectory speed would allow to study the speed of ecosystems responses in different management or pressure contexts complementary with time to recovery modeling (Monroe et al., 2022; Rydgren et al., 2011). To go further in the analysis of recovering and departing patterns with respect to the dynamic reference envelopes, one could also find interesting to decompose trajectory patterns focusing on trajectory segment distance to the centroid of the reference envelopes, or the use of trajectory convergence/divergence metrics (De Cáceres et al., 2019).

In these perspectives, the flexibility and complementarity of the state- and trajectory-based combination, the adaptability to different types of ecological questions (compositional, functional trait, structural, trophic, environmental variables, etc.) given by the choice of the space of analysis Ω (i.e., raw variables and dissimilarity metric choice) constitute the major strengths of the approach. We strongly believe that coupling EQA with traditional methods of analysis in experimental or field studies dealing with long-term integrative data sets in an ecological quality purpose could bring interesting perspectives for a better understanding of ecosystem functioning, protected areas efficiency, restoration operations effectiveness, trends in ecosystems quality, and past and present global changes.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

Data set and R code (De Cáceres, 2023) are included in the R package "ecotraj" and are available from Zenodo: https://zenodo.org/records/10053448.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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