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New synthetic indicators to assess community resilience and restoration success

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A B S T R A C T

The Strategic Plan for Biodiversity 2011–2020 sets as an objective the restoration of 15% of degraded ecosystems by 2020. This challenge raises at least two major questions: (i) How to restore and (ii) how to measure restoration success of said ecosystems? Measurement of restoration success is necessary to assess objective achievement and to adjust management with regard to objectives. Numerous studies are being conducted to try to work out synthetic indices to assess ecosystem diversity or integrity in the context of global change. Nevertheless, at the community level, there is no index that allows the assessment of community integrity regarding its restoration or resilience, despite the fact that a lot of indicators are used such as species richness, Shannon diversity, multivariate analyses or similarity indices. We have therefore developed two new indices giving new insights on community states: the first index, coined as the Community Structure Integrity Index, measures the proportion of the species abundance in the reference community represented in the restored or degraded community, and the second index, coined as the Higher Abundance Index, measures the proportion of the species abundance in the restored or degraded community which is higher than in the reference community. We illustrate and discuss the use of these new indices with three examples: (i) fictitious communities, (ii) a recent restoration (2 years) of a Mediterranean temporary wetland (Camargue in France) in order to assess restoration efficiency, and (iii) a recently disturbed pseudo-steppe plant community (La Crau area in France) in order to assess natural resilience of the plant community. The indices provide summarized information on the success of restoration or on the resilience of the plant community, which both appear less positive than with standard indicators already used. The indices also provide additional insights useful for management purposes: the Community Structure Integrity Index can indicate whether the improving target species abundance is needed or not while the Higher Abundance Index can indicate whether controlling the high abundance of some species is needed in order to approach a reference ecosystem. These relatively simple indices developed on community composition and structure state can provide a base to further indices focusing on ecosystem functioning or services not only calculating values as a static point but also its temporal or spatial dynamic.

1. Introduction

The Strategic Plan for Biodiversity 2011–2020 sets as an objective the restoration of 15% of degraded ecosystems by 2020 (Convention on Biological Diversity, 2011), and this challenge raises at least two major questions: (i) How to restore and (ii) how to measure restoration success of said ecosystems? The first question has been addressed and is still being addressed in a multitude of ecological systems and geographical areas (see for example (Perrow and Davy, 2002)) and for various restoration aims. Restoration targets are diverse: from rehabilitation, which is the restoration of one or some target ecosystem functions, to the restoration *sensu stricto*, which is the restoration of the whole ecosystem, i.e. its richness, composition, structure and functions (Society for Ecological Restoration International Science and Working Policy Group, 2004). Restoration is advocated for stopping the global erosion of biodiversity (Millennium Ecosystem Assessment, 2005; Nellemann et al., 2010), and is imposed by law in many countries for ecosystem destruction or degradation offsets (ten Kate et al., 2004). However, a recent meta-analysis conducted over 89 ecological restoration projects

Abbreviations: CSII, Community Structure Integrity Index; $CSII_{norm}$, normalized Community Structure Integrity Index; HAI, Higher Abundance Index.

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concluded that although restored ecosystems provide more biodiversity and ecosystem services than degraded ecosystems, these parameters still do not reach those of reference ecosystems (Benayas et al., 2009).

A community is defined as "an assemblage of populations of living organisms in a prescribed area or habitat" (Krebs, 1972). A multitude of indicators can be used to characterize a community (e.g. patchiness, nutrient cycling rate, interaction intensities, etc. (Noss, 1990)). To assess restoration success, most measures of biodiversity are related to abundance, species richness, diversity, growth, or biomass of organisms (Ruiz-Jaen and Aide, 2005). As strengthened by the analysis of 80 recent (2007–2011) papers comparing restored and reference communities, species richness and abundances are the most commonly used indicators of restoration (Appendix B). Species-richness is one of the simplest ways to describe a community (Magurran, 2004), however, many authors admit that species-richness, as well as diversity index (Shannon, Pielou, etc.), cannot be used alone (Noss, 1990). Indeed, completely different communities can be characterized by the same species-richness and diversity values. Our review analysis also pointed out an absence of consensus on indicators of community structure integrity: various multivariate analyses and various similarity-dissimilarity indices are widely used (52.5% and 20% of the studies respectively) (Appendix B). Nevertheless, all these indicators can have some drawbacks. Multivariate analyses are designed to maximize the variance while reducing the number of dimensions and provide a good overview of plant community composition and help to distinguish different plant communities (Borcard et al., 2011; McGarigal et al., 2000). While some methods allow us to significantly distinguish groups (Borcard et al., 2011; McArdle and Anderson, 2001), it is difficult to assess the magnitude of these differences between groups and impossible to compare, for example, the same restoration technique in two different ecosystems. Moreover, these types of analyses are not commonly used by practitioners because it is difficult to communicate their results to the general public. One-dimension measure, even if it summarizes more (and consequently reduces the amount of) information, is easier to interpret and can solve the problem of assessing magnitude differences. Examples of one-dimension community comparison measure are the widely used similarity-dissimilarity indices (such as Sorensen or Bray–Curtis) but these indices can be difficult to interpret: the dissimilarities can be attributed either to lower abundances of target species (i.e. species present in the reference community), or to higher abundances of target or non-target species compared with the reference community. These two explanations, which can occur concurrently, do not have the same implications in terms of community dynamics and hence of further management (Luken, 1990).

The objective of this work is therefore to develop an assessment method of community structure integrity after restoration (i.e. to measure restoration success) or after disturbance (i.e. to measure resilience) that measures the two types of community dissimilarities: lower and higher abundances in the restored or degraded community compared to reference communities. We have developed two indices giving additional insights on community states: the first index measures the proportion of the species abundance in the reference community represented in the restored or degraded community, and the second index measures the proportion of the species abundance in the restored or degraded community which is higher than in the reference community. We illustrate the use of these indices with fictitious communities, with an application to resilience and with an application to restoration in order to discuss the contribution of the new indices compared with existing ones, their perspective of utilization and limits.

2. Materials and methods

2.1. Indices description

The goal of our indices is to measure resilience or restoration success in a given community (the assessed community, AC), by comparison with a series of communities used as a reference (RC). Using a series of reference communities is crucial, as we expect undisturbed areas to present possible large variations in composition. Each community is characterized by a list of species each associated with a number (n) which reflects their abundance on a given area at a given date: size, biomass, abundance coefficient, percentage of cover, etc. The assessed community may be composed of target species (Clewell and Aronson, 2007), i.e. species present in the reference community, but also of non-target species. The idea behind our indices is to distinguish the species lower in abundance in the assessed community than in the reference communities, from the species higher in abundance in the assessed community than in the reference communities.

For a given species i, we note $\Delta i, j = \left| ni, \mathrm{AC} - ni, j \right|$ the absolute difference between the abundance in the assessed community and the abundance in reference community j. We indicate with a subscript whether the abundance in the assessed community is lower ($\Delta^{-}_{i,j}$) or higher ($\Delta^{+}_{i,j}$) than in the reference community.

We define 3 indices:

1) The Community Structure Integrity Index (CSII) measures the average proportion of species' abundance in the reference communities represented in the assessed community, and is defined as:

$$\mathrm{CSII} = \overline{\left[\frac{\sum_{i=1\ldots S}(n_i - \Delta^{-}_{i,j})}{\sum_{i=1\ldots S} n_{i,j}} \right]}^{j=1\ldots K}$$

with S the total number of species over all communities and K the total number of reference communities. The overbar stands for the arithmetic mean over all reference communities. The CSII index thus focuses on the "deficit" of abundance in the assessed community. It takes values between 0 and 1, and equals 1 when all species in the assessed communities are at least as abundant as in the reference communities.

2) The normalized Community Structure Integrity Index ($\mathrm{CSII_{norm}}$) is a normalized version of CSII. Indicators which represent measurable portions of a reference are the easiest to interpret and therefore the most convincing (Balmford et al., 2005; Duelli and Obrist, 2003). We calculate a normalized value of CSII as: $\mathrm{CSII_{norm}} = \frac{\mathrm{CSII}}{\mathrm{CSII_{RC}}}$ with $\mathrm{CSII_{RC}}$ the arithmetic mean of CSII calculated over all reference communities. Hence, reference communities have an average $\mathrm{CSII_{norm}}$ value of 1; this allows a meaningful comparison of $\mathrm{CSII_{norm}}$ values across ecosystems with different heterogeneity of reference communities.

3) The Higher Abundance Index (HAI) measures the average proportion of species' abundance in the assessed community higher than the reference communities, and is defined as:

$$\mathrm{HAI} = \overline{\left[\frac{\sum\limits_{i=1\ldots S} \Delta^{+}_{i,j}}{\sum\limits_{i=1\ldots S} n_{i,AC}} \right]}^{j=1\ldots K}$$

where the overbar stands for the arithmetic mean over all reference communities. HAI considers both target species having a higher abundance in the assessed community than in the reference community and non-target species.

No normalized version of HAI was developed as it is already a relative value to the whole assessed community structure.

We calculated the 3 indices and compared them to standard indicators in three case studies: one with fictitious communities, one in which resilience is assessed after disturbance, and one in which restoration is assessed.

2.2. Fictitious case study

New methods need to be tested rigorously before being applied to real data. We created fictitious communities which allowed us to confirm that the new indices show differences when they occur and do not show differences when they do not occur. We defined 10 types of fictitious communities: one reference, and nine assessed community types where the increase in target species abundances (T0, T0.5 and T1 having respectively 0×, 0.5× and 1× the abundance of target species in the reference) and the increase in non-target abundances (N0, N0.5 and N1 having respectively 0×, 0.5× and 1× the abundance of non-target species) were crossed, resulting in the following community types: T0N0, T0N0.5, T0N1, T0.5N0,T0.5N0.5, T0.5N1, T1N0, T1N0.5 and T1N1 (Fig. 1). As it is important that fictitious communities are the closest to what they are supposed to simulate (Zurell et al., 2010), we simulated 10 samples for each community type (representing the samples which could be surveyed in a community assessment), within which species abundances were characterized by means and variances similar to those found in an example of real plant communities assessed in a restoration context (Jaunatre et al., 2012).

2.3. Application to the resilience of a Mediterranean steppe after ploughing

La Crau area is the last xeric steppe in south-eastern France (ca. 10,000 ha; c. 43°33′ N, 4°52′ E) and has been shaped by (i) a Mediterranean climate: a mean annual temperature of 15 °C, a variable annual sum of precipitation between 400 and 600 mm concentrated in autumn and spring, with four months of summer drought, and more than 110 days with a >50 km h^{-1} wind; (ii) 40 cm deep soil composed with about 50% of silicaceous stones overlying a conglomerate layer, making the alluvial water table unavailable to the roots of plants and (iii) itinerant sheep grazing over a period of several thousand years (Buisson and Dutoit, 2006; Devaux et al., 1983). Although this area is protected by a French National Reserve status, a 5.7 ha area was accidentally ploughed in August 2010. Once the Reserve authorities were aware of the incident, the area was steamrollered in order to reduce the effects of ploughing. Vegetation relevés were carried out in order to assess the impact of such a disturbance: nine 4 m^2 quadrats were surveyed in the ploughed area and the unploughed area (reference community) in May 2011. Standard indicators and the three indicators presented above were calculated for both areas.

2.4. Application to the restoration by hay transfer of a Mediterranean meso-xeric grassland

The Camargue natural areas (Rhône Delta, south of France, 140,000 ha) have drastically declined with the combined effects of industrialization and agricultural development (Lemaire et al., 1987). Currently, opportunities arise to rehabilitate them on abandoned cultivated plots. The 70 ha Cassaïre site (c. 43°31′ N, 4°44′ E), is mostly composed of 70 former rice fields. The upper elevation of the site (3 m above sea level) is currently being restored by transferring hay from reference xero-halophytes communities of the Tour du Valat domain (Mesléard et al., 2011) located 10 km away from the restoration site. The hay was previously gathered by air-vacuuming in summer 2010 and transferred on five mesocosms (15 m × 5 m × 40 cm deep) randomly disposed on the site. Hay material was applied on a 2 m × 10 m plot (hay density = 11.5 g m^{-2}). Five control mesocosms where no hay transfer was applied were also randomly disposed. A vegetation survey was carried out in the hay transfer and the control using 50 cm × 50 cm grids in each mesocosm subdivided into 25 10 cm × 10 cm cells for each species recorded, giving a frequency. Five grids were also randomly surveyed in the reference community.

2.5. Analyses

We calculated standard indicators for the three case studies: species richness, Shannon index, Shannon evenness (Pielou, 1969) which are indicators of diversity, and Sorensen similarity and Bray–Curtis similarity (i.e. 1-Bray–Curtis dissimilarity index) which are both indicators of composition. The Sorensen index does not take abundances into account, while the Bray–Curtis index does (Borcard et al., 2011). In order to have one value of similarity for each assessed community sample, we calculated the mean of

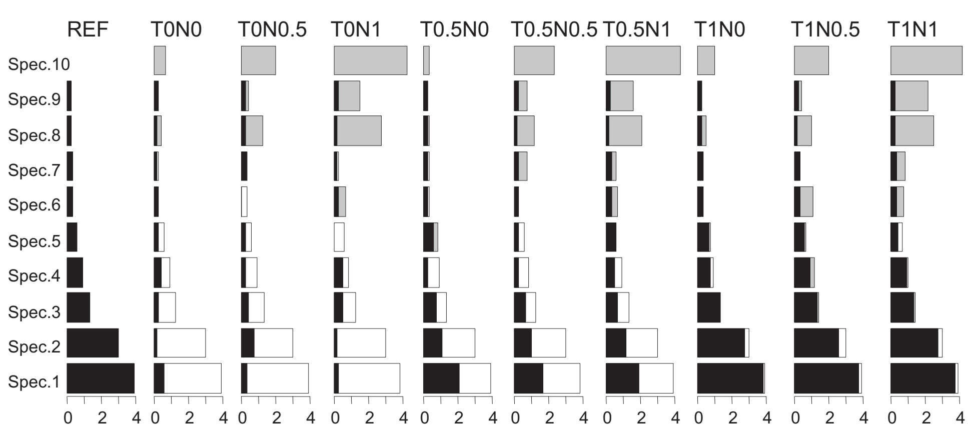

Fig. 1. Structure of the eight fictitious communities. White areas are missing abundances, black areas are abundances up to reference community abundances and grey areas are abundances higher than the reference community abundances. REF is the reference community, and the nine others are assessed community types where the increase in target species abundances (T0, T0.5 and T1 having respectively 0×, 0.5× and 1× the abundance of target species in the reference) and the increase in non-target abundances (N0, N0.5 and N1 having respectively 0×, 0.5× and 1× the abundance of non-target species). Data are mean ± SE, two bars with no letter in common are significantly different according to Tukey Honestly Significant Differences comparisons ($p < 0.05$).

Table 1
Description of standard indicators and of the new indices developed.

Indicators	Description of the indicators
Species-richness	Number of different species recorded in a delimited area.
Shannon index	Shannon index is a diversity index which expresses a ratio of proportion of species abundance relative to the whole community. The more one species dominates the community compared to other species, the higher Shannon index is. It is limited between 0 and a maximum potential which increases with species-richness.
Shannon evenness	Shannon evenness maximum potential value depends on the species-richness of the assessed community. Shannon evenness is relative to this potential maximum and is therefore limited to 1.
Sorensen similarity index	Sorensen similarity index is a similarity index between two samples which take into account only composition, not species abundance. It increases when two communities are close and is limited between 0 and 1.
Bray–Curtis similarity index	Bray–Curtis similarity index is a similarity index between two samples which take into account composition and species abundance. It increases when two communities are close and is limited between 0 and 1. Usually, Bray–Curtis dissimilarity is used but for clarity's sake, we used the similarity (1-Bray–Curtis similarity).
Community Structure Integrity Index (CSII)	CSII is an index calculated between a sample and one or several samples of a reference community. It measures the proportion of the species abundance in the reference community represented in the assessed community. It increases when target species abundance increases until their abundance reach those of reference community. It is limited between 0 and 1.
Normalized Community Structure Integrity Index ($CSII_{norm}$)	$CSII_{norm}$ is similar to the CSII but is normalized in a way that when it is calculated in the reference community it takes a 1 value. It is also limited between 0 and 1.
Higher Abundance Index (HAI)	HAI is an index calculated between a sample and one or several samples of a reference community. It measures the proportion of the species abundance in the assessed community which is higher than in the reference community. It increases when non-target species abundance increases or when target species abundance increases above their abundance in reference community. It is limited between 0 and 1.

similarities between that sample and each reference community sample. Then, in order to have one value of similarity for each reference community sample, we calculated the mean of similarities between that sample and each reference community sample. We also calculated the three new indices (HAI, CSII and $CSII_{norm}$,) for the three case studies.

After checking conformity to parametric conditions we performed *T*-tests for the Mediterranean steppe case study and an ANOVA followed by Tukey HSD post hoc tests for the fictitious and the Mediterranean xero-halophyte grassland case study to compare indicators between communities.

All calculations and analyses were performed with the package "stats" and "vegan" in R 2.13.0 (R Development Core Team, 2011) and we used the R code given in Appendix B for our three new indices (CSII, $CSII_{norm}$ and HAI) calculations and abundances plotting.

3. Results

3.1. Fictitious case study

Species-richness and Shannon index increased or decreased independently of which species occur in the assessed community. Obviously, the smaller species-richness was found in the T0N0 community and the highest species-richness in the T1N1 community (Figs. 1 and 2). The Shannon evenness, which is independent of species-richness, was the highest in the community with low abundances, and was not significantly different between the reference and the other community types. Sorensen similarity and Bray–Curtis similarity increased when target species abundances increased, but only Bray–Curtis similarity decreased when non-target species abundances increased. There was no significant difference in Bray–Curtis similarity indices between the T0.5N0 community, where target species abundances was lower than in the reference and non target species abundances null, and the T1N1 community, where target species abundances were equal to the reference and non target species abundances higher. CSII and $CSII_{norm}$ increased only when target-species abundances increased and were not significantly different from the reference when all the target species had the same abundance as in the reference. CSII and $CSII_{norm}$ were not influenced by the increase in non-target species abundances. On the contrary, HAI was significantly influenced by the increase in non-target species but not by target species abundances. However, when the overall abundance of community decreased, the HAI increased.

3.2. Resilience of a Mediterranean steppe

The reference and ploughed communities shared numerous species (Fig. 3), as expressed by their similar species-richness (Table 1). However many species have different abundances: some have higher abundance in the reference community (e.g. *Brachypodium distachyon*) or are absent in the ploughed community (e.g. *B. retusum*), whereas some have higher abundances in the ploughed community (e.g. *Bromus madritensis*), or were not recorded at all in the reference community (e.g. *Polycarpon tetraphyllum*). These differences in abundance were poorly shown by diversity indices: Shannon index was significantly different (1.68 ± 0.04 in the reference vs. 1.61 ± 0.07 in the ploughed community; $p = 0.04$) but Shannon evenness was not significantly different ($p = 0.38$). As for indices dealing with community composition (Sorensen similarity index, Bray–Curtis similarity index) and the three new indices (Community Structure Integrity Index, normalized Community Structure Integrity Index and Higher Abundance Index) we found significant differences between the reference and ploughed communities (Table 1). Sorensen and Bray–Curtis similarities were higher in the reference community than in the assessed community (ploughed community). The mean $CSII_{norm}$ reached 0.41 in the ploughed community meaning that 59% of the reference community was destroyed by the ploughing event. The reference community had a mean $CSII_{norm}$ of 1, while it had a mean CSII of 0.71. The reference community had a mean HAI of 0.29 significantly different from the ploughed community mean HAI of 0.64 meaning that 64% of the abundance in the ploughed community came from species in higher abundance than in the reference communities.

3.3. Restoration of a Mediterranean meso-xeric grassland

The restored hay transfer community shared more species with the reference community than with the control community (Fig. 4). However, as in the resilience case study, some species showed different abundances: some had higher abundance in the reference community (e.g. *Galium murale*) or were completely absent in the restored community (e.g. *B. phoenicoides*) whereas some had higher abundances in the restored community (e.g. *B. hordeaceus*), or were not recorded in the reference community (e.g. *Polygonum*

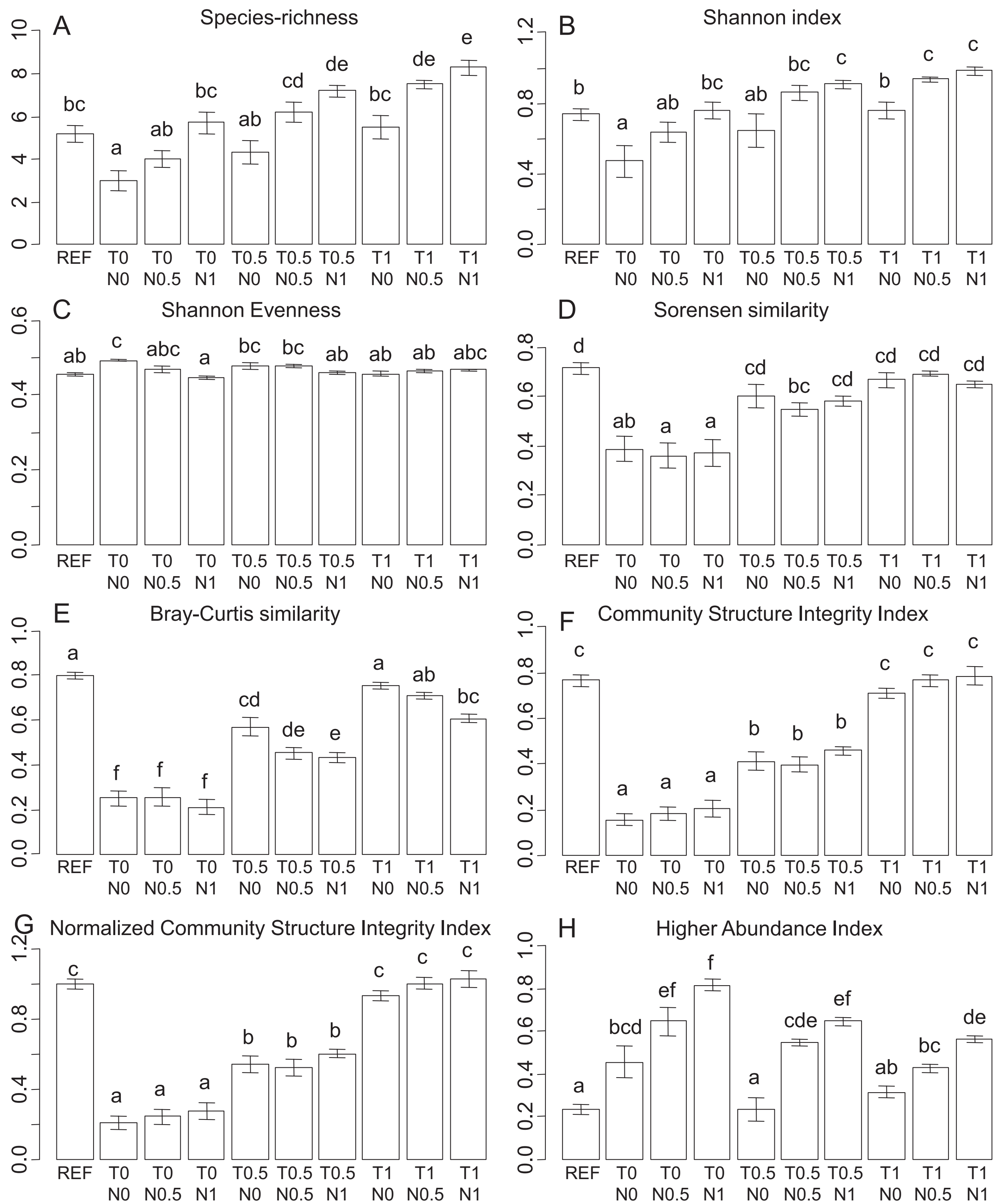

Fig. 2. Comparison of standard indicators (Species-richness, Shannon index, Shannon evenness, Sorensen similarity index, Bray–Curtis similarity index (1-Bray–Curtis dissimilarity index) and the three new indices (Community Structure Integrity Index, normalized Community Structure Integrity Index and Higher Abundance Index) in the ten fictitious communities. REF is the reference community, and the nine others are assessed community types where the increase in target species abundances (T0, T0.5 and T1 having respectively 0×, 0.5× and 1× the abundance of target species in the reference) and the increase in non-target abundances (N0, N0.5 and N1 having respectively 0×, 0.5× and 1× the abundance of non-target species). Data are mean ± SE, two bars with no letter in common are significantly different according to Tukey Honestly Significant Differences comparisons ($p < 0.05$).

aviculare). We did not find any differences in the Shannon index and species richness between reference and hay transfer community (Table 2). Nevertheless, Sorensen similarity index, Bray–Curtis similarity index and the three new indices ($CSII_{norm}$, CSII and HAI) were significantly different between the 3 communities ($p < 0.001$ for the five indices). Sorensen and Bray–Curtis similarities were the highest in the reference community and the lowest in the control. The mean $CSII_{norm}$ of the control was 0.01, meaning that only

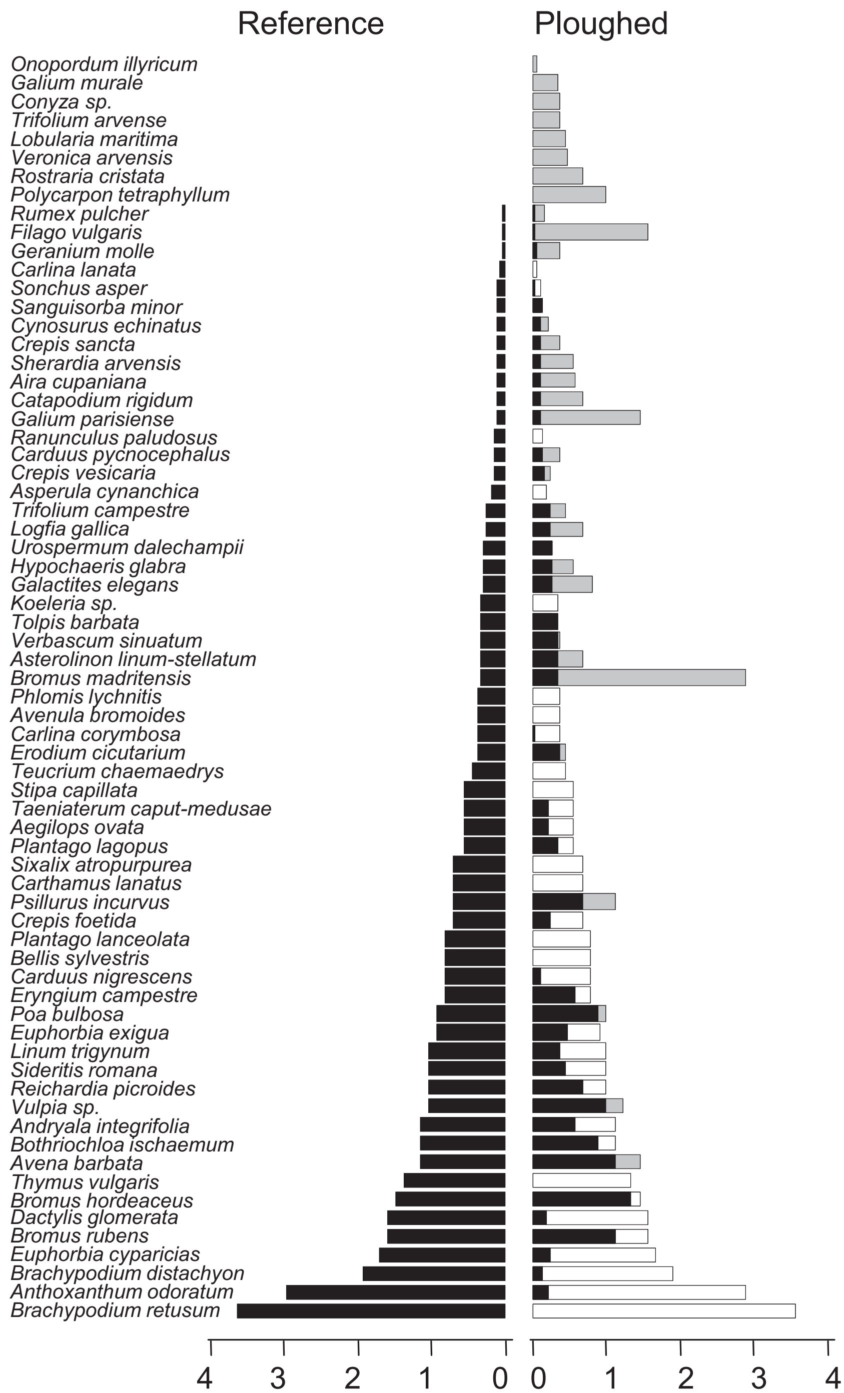

Fig. 3. Mean abundances of reference community and ploughed communities (assessed community) ($n = 9$). Black areas represent mean abundances in the reference communities. White areas represent mean missing abundances in the ploughed community, grey areas represent mean abundances in the ploughed community up to the mean abundances in the reference community and yellow areas represent abundances which are higher than in the reference community. For clarity purposes, only species which occur in more than 3 samples are shown (67 of the 119 species).

1% of the reference community abundance was expressed in this community. It reached a mean of 0.20 for the restored community, meaning that according to our index, 20% of the reference community has been restored. In the reference community the mean of the $CSII_{norm}$ and the CSII were respectively of 1 and 0.67. In this reference community the value of the mean HAI (0.32) was significantly different from the restored or the control (respectively 0.77 and 0.99) meaning the control community corresponded to 99% of the abundance of target species higher than the reference community or of non-target species. Table 3

4. Discussion

4.1. Comparison of standard indicators with CSII and HAI

Among the numerous indicators used to assess diversity (functional diversity, β diversity, etc.), some standard indicators are widely used in conservation biology (species-richness, Shannon or Shannon evenness) and provide useful information on community states. Nevertheless, when measuring resilience or restoration, they have to be cautiously interpreted. In our case studies we

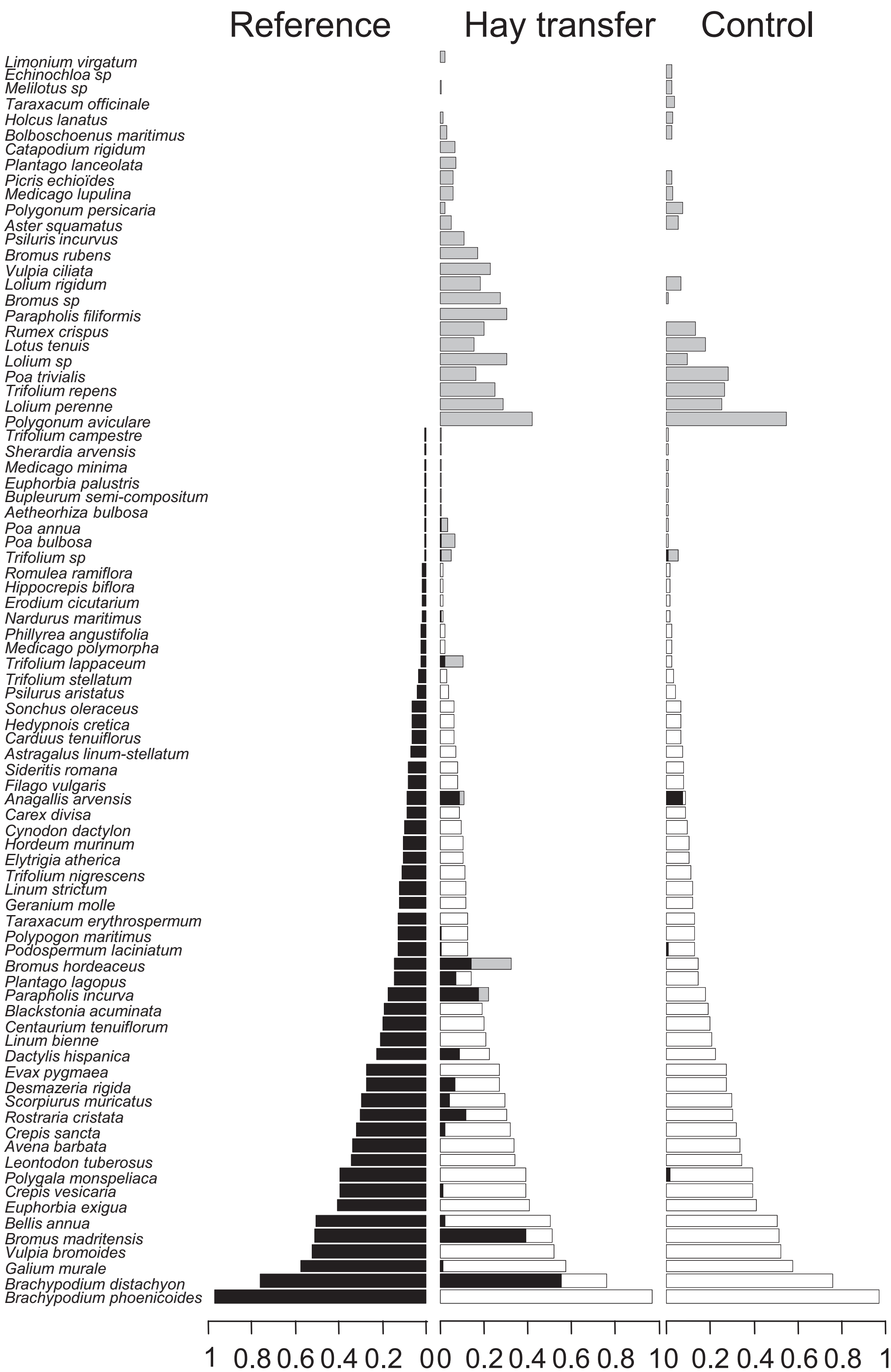

Fig. 4. Mean abundances of reference, hay transfer and control communities ($n = 5$). Black areas represent the mean abundances in the reference communities, white areas represent mean missing abundances in hay transfer and control communities, grey areas represent mean abundances in ploughed community up to the mean abundances in the reference communities and yellow areas represent mean abundances which are higher than in the reference communities. For clarity purposes, only species which occur in more than 2 samples are shown (83 on 97 species).

found no significant differences in the species-richness and evenness between the restored or ploughed community and their respective references, although the communities showed great differences in composition. More seriously, sometimes diversity indicators are higher in the assessed community than in the reference, despite the fact that the community is dominated by non-native or ruderal species (Balcombe et al., 2005). Even if species-richness and evenness were similar in the assessed

Table 2
Comparison of standard indicators (species-richness, Shannon index, Shannon evenness, Sorensen similarity index and Bray–Curtis similarity (i.e. 1-Bray–Curtis dissimilarity)) and the three new indices (Community Structure Integrity Index, normalized Community Structure Integrity Index and Higher Abundance Index) between the reference community and the ploughed area.

	Reference	Ploughed area	*t*	*df*	*p*	
Species-richness	33.78 ± 2.88	29.67 ± 4.43	1.90	14	0.078	
Shannon index	1.68 ± 0.04	1.61 ± 0.07	2.27	13	0.041	*
Shannon evenness	0.48 ± 0	0.48 ± 0	0.92	14	0.375	
Sorensen similarity index	0.71 ± 0.02	0.4 ± 0.08	9.71	9	<0.001	***
Bray–Curtis similarity index	0.71 ± 0.02	0.31 ± 0.06	14.50	10	<0.001	***
Community Structure Integrity Index	0.71 ± 0.03	0.29 ± 0.08	12.00	10	<0.001	***
Normalized Community Structure Integrity Index	1.00 ± 0.04	0.41 ± 0.11	12.00	10	<0.001	***
Higher Abundance Index	0.29 ± 0.03	0.64 ± 0.04	−17.47	14	<0.001	***

Reported values are means ± confidence interval (95%).
t is the statistic of the *t* test, *df* the degree of freedom.
p value (no sign: $p > 0.05$).
* $p < 0.05$.
*** $p < 0.001$.

communities and in their respective reference, we cannot consider that the meso-xeric grassland has been fully restored by hay transfer and that the ploughed steppe has fully recovered after one year. Similarity indices, which permit the comparison of the composition of two communities, are used to assess restoration or resilience (Appendix A). Some similarity indices, however, do not take abundance into account (e.g. Sorenson, Ochiai, etc. (Borcard et al., 2011)). Those indices cannot detect dissimilarities between two communities of identical composition but of different structure, as our fictitious communities example shows. Structure may be a determinant for ecosystem functioning (Chapin et al., 1997). Indices which depend on community structure should thus be preferred when assessing resilience or restoration (e.g. Bray–Curtis, etc. (Borcard et al., 2011)). In our case studies the Bray–Curtis similarity index is the standard indicator which expresses the largest difference between reference and assessed communities. Nonetheless, such indices, when deviating from the maximum similarity (i.e. 1 for similarity indices, 0 for dissimilarity indices), may reflect two different kinds of patterns: the species in the assessed community may have lower abundances than those in the reference community, or they may have higher abundances. Our three new indices permit disentangling these two different patterns, which can occur simultaneously. This is particularly illustrated by the fictitious case study. Indeed, when the abundances were higher in the assessed than in the reference community, Bray–Curtis similarity decreased. On the contrary, the CSSI does not depend on abundances that were higher than in the reference community and thus does not decrease. The similarity decreasing is expressed in the Higher Abundance Index, which then deviates from 0. The ploughed steppe community and the restored xero-halophytic grassland community had $CSII_{norm}$ of 0.41 and 0.20 respectively meaning that according to our indices, assessed communities contain 41% and 20% of abundances of their respective reference communities. Their mean HAI were 0.64 for the ploughed steppe community and 0.77 for the restored meso-xeric grassland community, meaning that, according to our indices, the assessed communities contained 64% and 77% of their respective total abundance which are higher abundances (i.e. non-target species or abundances of target species are higher than mean reference abundances).

4.2. Contribution of CSII and HAI to community assessment interpretation

The choice of an indicator depends on what one wants to measure, and on the objectives with which the measures are taken (Duelli and Obrist, 2003). Moreover, (Balmford et al., 2005) advocates using indicators that are rigorous, repeatable, and widely and easily understandable. $CSII_{norm}$ and HAI indices both represent

Table 3
Comparison of standard indicators (species-richness, Shannon index, Shannon evenness, Sorensen similarity index and Bray–Curtis similarity (i.e. 1-Bray–Curtis dissimilarity)) and the three new indices (Community Structure Integrity Index, normalized Community Structure Integrity Index and Higher Abundance index) between the reference community, the hay transfer community and the control community.

	Reference	Hay transfer	Control	*F*	*df*	*p*	
Species-richness	34.80 ± 4.95	25.00 ± 12.49	9.60 ± 8.31	18.69	2	<0.001	***
	a	a	b				
Shannon index	1.60 ± 0.09	1.41 ± 0.22	0.85 ± 0.68	8.71	2	0.005	**
	a	a	b				
Shannon evenness	0.45 ± 0.02	0.45 ± 0.03	0.44 ± 0.05	0.26	2	0.77	NS
Sorensen similarity index	0.71 ± 0.05	0.25 ± 0.16	0.03 ± 0.07	102.90	2	<0.001	***
	a	b	c				
Bray–Curtis similarity index	0.59 ± 0.06	0.16 ± 0.13	0.01 ± 0.01	128.86	2	<0.001	***
	a	b	c				
Community Structure Integrity Index	0.67 ± 0.07	0.13 ± 0.13	0.00 ± 0.01	170.56	2	<0.001	***
	a	b	c				
Normalized Community Structure Integrity Index	1 ± 0.11	0.20 ± 0.19	0.01 ± 0.02	176.56	2	<0.001	***
	a	b	c				
Higher Abundance index	0.32 ± 0.04	0.77 ± 0.18	0.99 ± 0.02	94.10	2	<0.001	***
	a	b	c				

Reported values are means ± confidence interval (95%).
Values on a line with a common letter are not significantly different (Tukey HSD test with a *p*-value adjustment according to Bonferroni's method).
F is the statistic of the ANOVA test, *df* the degree of freedom.
p the *p* value (NS: $p > 0.05$).
** $p < 0.01$.
*** $p < 0.001$.

easily understandable measurements for conservation biologists of a community state: $CSII_{norm}$ is the proportion of the reference community structure which can be found in the assessed community whereas HAI is the proportion of the assessed community structure that is represented by higher abundances than in the reference community. Knowing whether a community has a "deficit" of target species abundance or is characterized by higher abundances is of primary interest for practitioners who want to manage ecological succession (Kiehl and Pfadenhauer, 2007; Luken, 1990).

4.3. Applications of indices to restoration ecology and biological conservation

Low values of CSII express a lack of target species in the assessed community. Therefore identifying the reasons why these species do not reach the reference community abundances is of primary interest. If target species do not disperse, the propagule source may be too far away or the target species do not produce sufficiently dispersible propagules: management can be focused on strengthening dispersion processes (see Kiehl et al. (2010) for review). For example, the restored meso-xeric grassland case study shows that dispersion strengthening by hay transfer increases CSII value. Environmental conditions may be too far from the growth optimum of target species, in which case management should involve trying to restore suitable conditions (Bakker and Berendse, 1999; Dorland et al., 2005). Target species may also be in competition with non-target species (D'Antonio et al., 2003), which will be expressed with high values of HAI. Management should then involve trying to decrease abundances of these species with higher abundances, whether it concerns target species or not (Donath et al., 2003; Murray and Marmorek, 2003). More than a static measurement, these indices may be used to monitor the succession of assessed communities. Increasing CSII values could show that dispersion strengthening is not necessary. On the contrary, an increase of HAI, even if the values are low, can indicate the need for managing higher abundance (Donath et al., 2003; Haywood, 2009). In both real case studies, HAI are significantly higher than in the reference community. If HAI increases during forthcoming years, the actual site management, extensive sheep grazing, will have to be adapted to reduce higher abundance. Otherwise these species with higher abundance may have a negative feedback on the CSII values and thus threaten the maintenance of community integrity success.

4.4. Limits and constraints of CSII and HAI use

Particular attention should be paid to data gathering before performing indices calculations. Whether it is for assessing resilience or restoration efficiency, the definition and characterization of reference ecosystems are crucial (White and Walker, 2008). A broad part of ecological restoration literature deals with this issue (Egan, 2001; Ehrenfeld and Toth, 1997). In order to avoid bias in HAI or CSII calculations, similar community characterization protocol should be used in reference and assessed ecosystems (same sample size, working effort, plant identification skills and date of sampling). Communities are not static entities and, at least in the framework of restoration, the reference should be all the manifested or potential states that occur within a given historical and spatial variation (Landres et al., 1999; Society for Ecological Restoration International Science and Working Policy Group, 2004). Therefore, reference community characterization should take into account the natural variability of the reference, both spatially and temporally (White and Walker, 2008). Calculation of CSII and HAI should be performed in both the reference and assessed communities. Indeed the indices give information on the reference community variability and heterogeneity and allow statistical analyses comparing the reference and assessed communities. These comparisons provide an overview of the assessed community but do not account for the whole complexity of an ecosystem: functional, spatial or dynamic attributes are eluded. Therefore these indices should be used in addition to standard indicators or more specific ones adapted to each case study (see for example Raab and Bayley (2012)). Moreover, in a context of the evaluation of a restoration project, assessment of one community of the whole ecosystem is not sufficient to draw conclusions on the project. Several communities should be assessed (i.e. plants, insects, birds, mammals, microbes, etc.), as well as environmental characteristics (i.e. soil chemistry, disturbance regime, etc.) or landscape-scale indicators (i.e. fragmentation, etc.) (Palmer et al., 2005; Tasser et al., 2008).

4.5. Perspective of use and development of CSII and HAI

All species do not necessarily have the same status in a community, whether they could exert a more significant role in ecosystem functioning or services (Bullock et al., 2011; Funk et al., 2008) or they could be of high conservation value. It could have been relevant to give more weight to high conservation value species in the calculation of CSII indices or to give more weight to species with a high invasion potential for the HAI. However, these resulting indices would deviate from the original goal of these indices: measuring in an easily interpretable way the difference from a reference community.

To our knowledge, no meta-analyses have tried to measure the abilities of ecological restoration projects to restore reference community integrity. It has been proved that restoration exerts a significant positive effect on diversity or ecosystem services (Benayas et al., 2009). Regarding the high differences sometimes existing between standard indicators and CSII in our case studies, it would be interesting to perform these indices calculations in such meta-analyses.

Metaphorically speaking, if we compare restoration with assembling a jigsaw puzzle, species-richness would be equivalent to the colour palette of the puzzle and Shannon index, or evenness, would be the correct equilibrium of colours, whereas CSII could be compared to the number of correct pieces of the puzzle. This metaphor leads to two comments: (1) It seems obvious that even the correctly balanced colour palette is not enough to complete the puzzle if 50% of the pieces are missing and (2) Even with all the pieces, they have to be assembled adequately to obtain the desired picture. To our knowledge, there is no indicator which measures this community configuration (apart from random/aggregated distribution) although it has been proved to exert a significant effect on ecosystem functioning (Maestre et al., 2012). Consideration of how to measure the state of a community in a framework of restoration or resilience assessment should be continued to set realistic and measurable goals for ecosystem management as noticed by Ehrenfeld and Toth (1997).

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2013.01.023.

References

Bakker, J.P., Berendse, F., 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. Trends Ecol. Evol. 14, 63–68.

Balcombe, C., Anderson, J., Fortney, R., Rentch, J., Grafton, W., Kordek, W., 2005. A comparison of plant communities in mitigation and reference wetlands in the mid-appalachians. Wetlands 25, 130–142.

Balmford, A., Bennun, L., ten Brink, B., Cooper, D., Côté, I.M., Crane, P., Dobson, A., Dudley, N., Dutton, I., Green, R.E., Gregory, R.D., Harrison, J., Kennedy, E.T., Kremen, C., Leader-Williams, N., Lovejoy, T.E., Mace, G., May, R., Mayaux, P., Morling, P., Phillips, J., Redford, K., Ricketts, T.H., Rodìguez, J.P., Sanjayan, M., Schei, P.J., van Jaarsveld, A.S., Walther, B.A., 2005. The convention on biological diversity's 2010 target. Science 307, 212–213.

Benayas, J.M.R., Newton, A.C., Diaz, A., Bullock, J.M., 2009. Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. Science 325, 1121–1124.

Borcard, D., Gillet, F., Legendre, P., 2011. Numerical Ecology with R. Springer, New York.

Buisson, E., Dutoit, T., 2006. Creation of the natural reserve of La Crau: implications for the creation and management of protected areas. J. Environ. Manage. 80, 318–326.

Bullock, J.M., Aronson, J., Newton, A.C., Pywell, R.F., Rey-Benayas, J.M., 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends Ecol. Evol. 26, 541–549.

Chapin, F.S., Walker, B.H., Hobbs, R.J., Hooper, D.U., Lawton, J.H., Sala, O.E., Tilman, D., 1997. Biotic control over the functioning of ecosystems. Science 277, 500–504.

Convention on Biological Diversity, 2011. Ways and means to support ecosystem restoration, 17 pp, Available at: http://www.cbd.int/doc/meetings/sbstta/sbstta-15/official/sbstta-15-04-en.pdf

Clewell, A.F., Aronson, J., 2007. Ecological Restoration: Principles, Values, and Structure of an Emerging Profession, 1st ed. Island Press, Washington, DC.

D'Antonio, C., Bainbridge, S., Kennedy, C., Bartolome, J., Reynolds, S., 2003. Ecology and restoration of California grasslands with special emphasis on the influence of fire and grazing on native grassland species, 99 pp, Available at: http://www.cnga.org/library/DAntonioGrassReviewArticle.pdf

Devaux, J.P., Archiloque, A., Borel, L., Bourrely, M., Louis-Palluel, J., 1983. Notice de la carte phyto-écologique de la Crau (Bouches du Rhône). Biol. Ecol. Méditerranéenne 10, 5–54.

R Development Core Team, 2011. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Donath, T.W., Holzel, N., Otte, A., 2003. The impact of site conditions and seed dispersal on restoration success in alluvial meadows. Appl. Veg. Sci. 6, 13–22.

Dorland, E., Van Den Berg, L.J.L., Brouwer, E., Roelofs, J.G.M., Bobbink, R., 2005. Catchment liming to restore degraded acidified heathlands and moorland pools. Restor. Ecol. 13, 302–311.

Duelli, P., Obrist, M.K., 2003. Biodiversity indicators: the choice of values and measures. Agric. Ecosyst. Environ. 98, 87–98.

Egan, D., 2001. The Historical Ecology Handbook: A Restorationist's Guide to Reference Ecosystems. Island Press, Washington, DC.

Ehrenfeld, J.G., Toth, L.A., 1997. Restorarion ecology and the ecosystem perspective. Restor. Ecol. 5, 307–317.

Funk, J.L., Cleland, E.E., Suding, K.N., Zavaleta, E.S., 2008. Restoration through reassembly: plant traits and invasion resistance. Trends Ecol. Evol. 23, 695–703.

Haywood, J.D., 2009. Eight years of seasonal burning and herbicidal brush control influence sapling longleaf pine growth, understory vegetation, and the outcome of an ensuing wildfire. Forest Ecol. Manage. 258, 295–305.

Jaunatre, R., Buisson, E., Dutoit, T., 2012. First-year results of a multi-treatment steppe restoration experiment in La Crau (Provence France). Plant Ecol. Evol. 145, 13–23.

Kiehl, K., Kirmer, A., Donath, T.W., Rasran, L., Hölzel, N., 2010. Species introduction in restoration projects – evaluation of different techniques for the establishment of semi-natural grasslands in Central and Northwestern Europe. Basic App. Ecol. 11, 285–299.

Kiehl, K., Pfadenhauer, J., 2007. Establishment and persistence of target species in newly created calcareous grasslands on former arable fields. Plant Ecol. 189, 31–48.

Krebs, C.J., 1972. Ecology; the Experimental Analysis of Distribution and Abundance. Harper & Row, New York.

Landres, P.B., Morgan, P., Swanson, F.J., 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecol. Appl. 9, 1179–1188.

Lemaire, S., Tamisier, A., Gagnier, F., 1987. Surface, distribution et diversité des principaux milieux de Camargue: leur évolution de 1942 à 1984. Revue d'écologie, 47–56.

Luken, J.O., 1990. Directing Ecological Succession. Chapman and Hall, New York.

Maestre, F.T., Castillo-Monroy, A.P., Bowker, M.A., Ochoa-Hueso, R., 2012. Species richness effects on ecosystem multifunctionality depend on evenness, composition and spatial pattern. J. Ecol. 100, 317–330.

Magurran, A.E., 2004. Measuring Biological Diversity. Blackwell, Oxford, UK.

McArdle, B.H., Anderson, M.J., 2001. Fitting multivariate models to community data: a comment on distance-based redundancy analysis. Ecology 82, 290–297.

McGarigal, K., Cushman, S., Stafford, S., 2000. Multivariate Statistics for Wildlife and Ecology Research. Springer, New York, USA.

Mesléard, F., Mauchamp, A., Pineau, O., Dutoit, T., 2011. Rabbits are more effective than cattle for limiting shrub colonization in Mediterranean xero-halophytic meadows. Ecoscience 18, 37–41.

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-Being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Murray, C., Marmorek, D., 2003. Adaptive management and ecological restoration. In: Frederici, P. (Ed.), Ecological Restoration of Southwestern Ponderosa Pine Forests. Island Press, Washington, pp. 417–428.

Nellemann, C., Corcoran, E., Nellemann, C., Corcoran, E., 2010. Dead Planet Living Planet: Biodiversity and Ecosystem Restoration for Sustainable Development. United Nations Environment Programme, Nairobi.

Noss, R.F., 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conserv. Biol. 4, 355–364.

Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Follstad Shah, J., Galat, D.L., Loss, S.G., Goodwin, P., Hart, D.D., Hasset, B., Jenkinson, R., Kondolf, G.M., Lave, R., Meyer, J.L., O'donnel, T.K., Pagano, L., Sudduth, E., 2005. Standards for ecologically successful river restoration. J. Appl. Ecol. 42, 208–217.

Perrow, M.R., Davy, A.J., 2002. Handbook of Ecological Restoration: Volume 2, Restoration in Practice, 1st ed. Cambridge University Press, Cambridge.

Pielou, E.C., 1969. An introduction to Mathematical Ecology. Wiley-Interscience, New York.

Raab, D., Bayley, S.E., 2012. A vegetation-based index of biotic integrity to assess marsh reclamation success in the Alberta oil sands. Canada Ecol. Indic. 15, 43–51.

Ruiz-Jaen, M.C., Aide, M.T., 2005. Restoration success: how is it being measured? Restor. Ecol. 13, 569–577.

Society for Ecological Restoration International Science and Working Policy Group, 2004. The SER International Primer on Ecological Restoration, 15pp, Available at: http://www.ser.org/resources/resources-detail-view/ser-international-primer-on-ecological-restoration

Tasser, E., Sternbach, E., Tappeiner, U., 2008. Biodiversity indicators for sustainability monitoring at municipality level: an example of implementation in an alpine region. Ecol. Indic. 8, 204–223.

ten Kate, K., Bishop, J., Bayon, R., 2004. Biodiversity offsets: views, experience, and the business case. IUCN, Gland, Switzerland and Cambridge, UK and Insight Investment, London, UK.

White, P.S., Walker, J.L., 2008. Approximating nature's variation: selecting and using reference information in restoration ecology. Restor. Ecol. 5, 338–349.

Zurell, D., Berger, U., Cabral, J.S., Jeltsch, F., Meynard, C.N., Münkemüller, T., Nehrbass, N., Pagel, J., Reineking, B., Schröder, B., Grimm, V., 2010. The virtual ecologist approach: simulating data and observers. Oikos 119, 622–635.