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Vegetation richness and rarity in habitats of European conservation value in Ireland

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ABSTRACT

Species richness is widely used as a measure of biodiversity in quantifying the conservation value of sites or habitats and in researching ecological functioning, but it gives equal weighting to different species irrespective of their rarity. We developed a Rarity Co-efficient and related indices utilising plant species distribution records to investigate the significance of this limitation and applied these to quantitative vegetation plot data from Irish habitats of European importance. The conservation value of habitats and ecosystems differed markedly depending upon whether it was measured by species richness, species rarity or both richness and rarity. Application of abundance-weighting had noteworthy effects on index values at the community level; dominance of non-native or very common species reduced values, while abundance of rare species increased values. Index values varied along successional gradients occurring in coastal dune systems and saltmarsh. Almost no significant concordance was found between the rankings of habitats based on richness or rarity index values derived from vascular plant data and those based on index values derived from bryophyte data, therefore these taxonomic groups are poor surrogates for each other. Habitats were divided into three categories based on similar patterns in these rankings: (a) habitats with specialist vascular plants, (b) species-rich vascular plant habitats and (c) important bryophyte habitats. Habitat assessment criteria should reflect these broad differences in conservation value. Integration of rarity indices into site conservation value assessments is recommended.

1. Introduction

Species richness is the iconic measure of biodiversity (Mace et al., 2012). It has been widely used when investigating the linkages between biodiversity, ecological functioning and ecosystem services (e.g. García and Martínez, 2012; Maestre et al., 2012) and also to quantify the conservation value of sites and habitats (e.g. Costanza et al., 2007; Paillet et al., 2010). This conceptually simple measure may also be employed as part of monitoring protocols for communities at sites of recognized biodiversity value (e.g. Chiarucci et al., 2001; Freeman et al., 2017) and is one score used for prioritizing locations for conservation efforts (Orme et al., 2005).

There are, however, limitations when using species richness in isolation as a surrogate for biodiversity values, as such data do not convey information on native status, endemism, rarity, abundance, function or sensitivity (Fleishman et al., 2006). For example, species richness does not take into account the varying contribution that different species make to biodiversity when assessed at particular geographical scales. These contributions can be evaluated at a community level by indices that incorporate some measure of rarity. Such indices have been applied in a variety of contexts, for example rich-fen vegetation (Wheeler, 1988), bumblebees (Williams, 2000), spiders (Samu et al., 2008), urban plants (Godefroid and Koedam, 2007), freshwater habitats (Davies et al., 2008; Williams et al., 2004) and marine fish (Daan, 2001). Some of these rarity indices have been based solely on distribution or range data (e.g. Williams, 2000) while others have also taken into account protected or threatened status (e.g. Red Data Book status, Davies et al., 2008).

Here, we apply a group of related rarity indices to vegetation samples from a range of habitats in Ireland. We focus on a selection of habitats listed in Annex I of the EU Habitats Directive (92/43/EEC), to which Ireland is a signatory and has reporting obligations (NPWS, 2019), as these are recognised as being of high conservation value at a continental level. Our primary aims were to: (i) determine if the conservation valuation of these habitats differed between measures reflecting species richness only, species rarity only or both species richness and species rarity; (ii) determine if these conservation valuations differed when these measures were applied to data representing

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Table 1
Habitats and ecosystem groupings used in the analysis, number of plots after stratification and data sources. * indicates an Annex I priority habitat in whole or part.

Annex I code	Habitat	Ecosystem	*n*	Source
1210	Drift lines	Shingle	13	Sand Dunes Monitoring Project
1220	Perennial vegetation of stony banks	Shingle	8	Sand Dunes Monitoring Project
1310	*Salicornia* mud	Saltmarsh	67	Saltmarsh Monitoring Project/Penk et al. (2020)
1330	Atlantic salt meadows	Saltmarsh	424	Saltmarsh Monitoring Project/Penk et al. (2020)
1410	Mediterranean salt meadows	Saltmarsh	204	Saltmarsh Monitoring Project/Penk et al. (2020)
1420	Halophilous scrub	Saltmarsh	4	Saltmarsh Monitoring Project
2110	Embryonic shifting dunes	Dunes	24	Sand Dunes Monitoring Project
2120	Marram dunes	Dunes	21	Sand Dunes Monitoring Project
2130*	Fixed dunes (grey dunes)	Dunes	22	Sand Dunes Monitoring Project
2170	Dunes with creeping willow	Dunes	4	Sand Dunes Monitoring Project
2190	Dune slack	Dunes	19	Sand Dunes Monitoring Project
21A0*	Machair	Dunes	8	Sand Dunes Monitoring Project
4010	Wet heaths	Heath	181	National Survey of Upland Habitats
4030	Dry heaths	Heath	127	National Survey of Upland Habitats
4060	Alpine and subalpine heath	Heath	62	National Survey of Upland Habitats
6210*	Orchid-rich calcareous grassland	Grassland	217	Irish Semi-natural Grasslands Survey
6230*	Species-rich *Nardus* grassland	Grassland	68	Irish Semi-natural Grasslands Survey
6410	*Molinia* meadows	Grassland	160	Irish Semi-natural Grasslands Survey
6510	Lowland hay meadows	Grassland	49	Irish Semi-natural Grasslands Survey
7130*	Blanket bogs (active)	Bog	236	National Survey of Upland Habitats
7140	Transition mires	Bog	21	National Survey of Upland Habitats
7150	*Rhynchosporion* depressions	Bog	25	National Survey of Upland Habitats
8110	Siliceous scree	Scree	39	National Survey of Upland Habitats
–	Rush pasture	Rush pasture	589	Irish Semi-natural Grasslands Survey

different major taxonomic groups; (iii) determine if categorisations of habitats could be made based on patterns in these conservation values; and (iv) examine the application of these indices at ecosystem, habitat and community levels. Based on our findings, we also sought to make some practical recommendations for monitoring and assessing the biodiversity of these assemblages.

2. Methods

2.1. Data preparation

Quantitative vegetation plot data were sourced for 23 Annex I habitats (Table 1) that we grouped by seven ecosystem types: saltmarsh, coastal shingle, coastal dunes, grassland, bog, heath and scree. To provide an additional comparison, we also included plot data from rush pasture as an example of a non-Annex I habitat of perceived low conservation value. All rush pasture plots were classified as either GL2B *Juncus effusus – Holcus lanatus* grassland or GL2D *Juncus effusus – Rumex acetosa* grassland of the Irish Vegetation Classification (IVC) (Perrin et al., 2018).

With the exception of some of the saltmarsh data, which were collected by Penk et al. (2020), all data were collected during national surveys in Ireland for the National Parks & Wildlife Service. The Sand Dunes Monitoring Project was reported in Delaney et al. (2013), the Saltmarsh Monitoring Project in McCorry (2007) and McCorry and Ryle (2009), and the Irish Semi-natural Grasslands Survey in O'Neill et al. (2013). The National Survey of Upland Habitats was reported in a sequence of reports, for example Perrin et al. (2014). Only 2 m × 2 m plots were used to prevent species-area effects. This size of plot is appropriate for the stature of vegetation in the selected habitats but we do not purport that this is necessarily the most appropriate plot size for analysis of species richness or rarity as we have not yet tested the effect of varying plot sizes on the indices reported here. The dataset for each habitat was stratified such that for each monad (1 km × 1 km grid square of the Irish National Grid) from which plots had been recorded, only one randomly selected plot per habitat was retained. This reduced the potential for pseudoreplication (Hurlbert, 1984) as plots were often clustered at the site level due to the recording methodology of the original habitat surveys. Cover scores were converted from the Domin scale to mid-range percentages where required as follows: 10 = 96%; 9 = 83%; 8 = 63%; 7 = 42%; 6 = 30%; 5 = 18%; 4 = 8%; 3 = 3%; 2 = 0.5%; 1 = 0.3%; + = 0.1%. Plant records made only to the genus level were excluded. If the sum of the cover-abundance of genus-level records within a plot was 5% or greater, then the whole plot was excluded. The sample size for a minority of habitats was relatively small (Table 1) as some of these habitats have restricted areas. The corresponding results should thus be treated with some caution.

2.2. Data analysis

For each native species of vascular plant and bryophyte in the dataset we found the number of hectads (10 km × 10 km grid squares of the Irish National Grid) from which it has been recorded across the island of Ireland (H). Hectad distribution data were from Preston et al. (2002) for vascular plants and largely from Blockeel et al. (2014) for bryophytes, but where there were taxonomic differences between Blockeel et al. (2014) and the compiled dataset, unpublished hectad data provided by O. Pescott of the Centre for Ecology & Hydrology, UK, were used. Records from all years were included to reduce the effect of recording effort varying by time period; hectad coverage of bryophyte recording in Ireland has much improved in recent years, however, the H values for bryophytes may still be influenced by a degree of under-recording (Blockeel et al., 2014). Archaeophytes and species regarded as having uncertain native status in Ireland were treated as native species. H values were then used to calculate a Rarity Co-efficient (R) for each native species using a subjectively chosen exponential decay relationship (Fig. 1). This curve was selected so that for lower values of H (< 200 hectads) values of R decline rapidly with increases in H, but for higher values of H (> 600 hectads) values of R decline only slowly with increases in H. Furthermore, assuming there are 985 Irish hectads (Hill et al., 2004), values of R for native species can conveniently vary from 1.02 (species present in all hectads) to 9.95 (species present in only 1 hectad). As rarity was being used as a measure of conservation value, non-native species were assigned an R of zero. Examples of R values for a selection of saltmarsh plants are given in Table 2. Amongst these examples, it can be seen that *Atriplex portulacoides* L., a species largely restricted to saltmarsh, has a high R value, whereas *Festuca rubra* L., a species found commonly in several habitats, has a low R value.

Using these data, we calculated four different indices in addition to

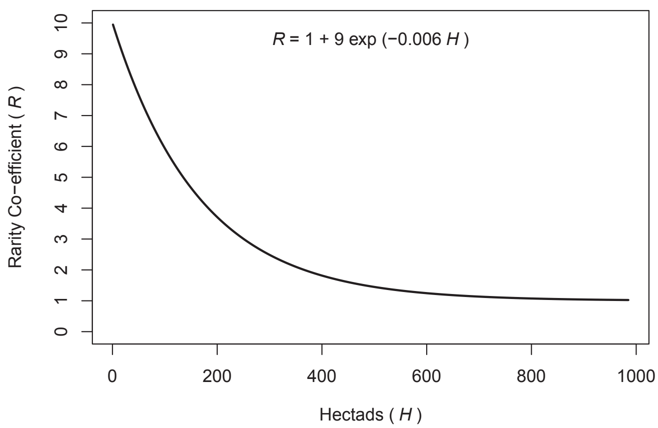

Fig. 1. Exponential decay curve relating the number of hectads from which each native species has been recorded in Ireland with its Rarity Co-efficient.

Table 2
Examples of Rarity Co-efficient (R) values for species found in Irish saltmarsh.

Species	Native status	Number of hectads (H)	Rarity Co-efficient (R)
Spartina anglica (C.E. Hubb.)	Non-native	71	0
Festuca rubra L.	Native	974	1.03
Plantago maritima L.	Native	405	1.79
Puccinellia maritima (Huds.) Parl.	Native	201	3.69
Spergularia media (L.) C. Presl	Native	186	3.95
Atriplex portulacoides L.	Native	68	6.98
Juncus acutus L.	Native	28	8.61
Puccinellia fasciculata (Torr.) E.P. Bicknell	Native	11	9.43
Sarcocornia perennis (Mill.) A.J. Scott	Native	4	9.79

species richness (S, which, for the sake of simplicity, we will also hereafter refer to as an ‘index’). Indices were calculated for each plot separately for vascular plants and for bryophytes. As a measure of both richness and rarity, we calculated the sum of Rarity Co-efficient values for each plot (Sum R):

$$\text{Sum } R = \sum_{i=1}^{S} R_i \tag{1}$$

where R_i is the R of species i. For a closer measure of just rarity, we used the mean Rarity Co-efficient ($\bar{R}$):

$$\bar{R} = \frac{\sum_{i=1}^{S} R_i}{S} \tag{2}$$

To describe rarity weighted by species abundance, an abundance-weighted mean Rarity Co-efficient ($\bar{R}_w$) was calculated as:

$$\bar{R}_w = \frac{\sum_{i=1}^{S} R_i A_i}{\sum_{i=1}^{S} A_i} \tag{3}$$

where A_i is the percentage cover of species i. We also calculated the reciprocal form ($1/D$) of Simpson’s index of diversity, which takes into account species richness and the relative abundance of each species:

$$1/D = 1/\sum_{i=1}^{S} p_i^2 \tag{4}$$

where p_i is the proportion of species i. If a plot lacked bryophytes or vascular plants it was assigned a $1/D$ value of zero for that component. Relationships between these five indices are examined in Fig. S1.

All statistical tests were conducted within the R statistical environment (R Core Team, 2018). Differences in indices between ecosystems were analysed using Kruskal-Wallis tests with *post hoc* tests for multiple comparisons (package *pgirmess;* Giraudoux, 2017). At the habitat level, we calculated the mean values by habitat for each of the five indices applied to both vascular plant and bryophyte data and ranked these scores. We then used k-means clustering to create several partitions of the habitats based on these ten rankings (k = 1, 2, 3 … 10) and selected the optimal number of clusters with the Calinski-Harabasz criterion (package *vegan*; Oksanen et al., 2017). To examine how these rankings at the habitat level differed between indices, we used Kendall’s coefficient of concordance (package *synchrony*; Gouhier and Guichard, 2014); rather than correct p-values for multiple comparisons, which can lead to unacceptable levels of type II errors, we present effect sizes (Moran, 2003; Nakagawa, 2004). To examine patterns at the community level, we classified the saltmarsh plots using both vascular plant and bryophyte data to communities of the current version of the IVC using the ERICA (Engine for Relevés to Irish Communities Assignment) v4.0 application (Perrin et al., 2018); this is an R-based application that utilises a noise-clustering approach (package *vegclust*; De Cáceres et al., 2010). Rush pasture plots were also identified using this application.

3. Results

3.1. General

Stratification yielded a dataset of 2592 plots containing 454 vascular plant species and 269 bryophyte species. Of the native vascular plant species within the dataset, 60.4% can be regarded as very common ($1 < R \le 2$) at a national level, with the frequency of R values generally decreasing as R increased (Fig. 2), resulting in a median R value of 1.29. In contrast, only 28.3% of native bryophyte species were categorised as very common, with a median R value of 3.63 (Fig. 2). There were relatively few non-native vascular plants and only a single non-native bryophyte – *Campylopus introflexus* (Hedw.) Brid.

3.2. Ecosystem-level analysis

There were significant differences between ecosystems in the distribution of values of all indices applied to the vascular plant data (Fig. 3). Grassland had a higher species richness (median S = 26 species/4 m^2) than all other ecosystems, with saltmarsh, scree and shingle being the most species-poor with medians of 6, 5 and 4 respectively. Notably, rush pasture, comprising non-Annex I habitat, was the second most species-rich category (median S = 14). Simpson’s index of diversity ($1/D$) yielded a similar pattern of results (Fig. 3). In contrast, the median value of $\bar{R}$ for grasslands (i.e. the median of all the mean R values for grassland plots) of 1.15 was lower than that of all other habitat groups except rush pasture (median $\bar{R}$ = 1.05). Shingle (median $\bar{R}$ = 3.60) and saltmarsh (median $\bar{R}$ = 2.65) had the highest values, whilst dunes displayed a notably greater variation in values than other ecosystems. When the elements of richness and rarity were combined as Sum R, grassland (median Sum R = 30.52) scored highest again, but rush pasture (median Sum R = 14.73) was now statistically comparable with shingle (median Sum R = 14.34) and dunes (median Sum R = 16.66). Bog (median Sum R = 10.88), scree (median Sum R = 11.09) and heath (median Sum R = 11.14) had the lowest scores for this index. $\bar{R}_w$ produced a similar pattern to its unweighted counterpart.

There were also significant differences between ecosystems in the distribution of values of all indices applied to the bryophyte data (Fig. 4). Bryophytes were largely absent from saltmarsh and shingle and from many of the dune plots. Bryophyte species richness was highest in heath, bog and scree (median S = 10 for all three ecosystems) with intermediate levels of richness in grassland (median S = 5) and low levels in rush pasture (median S = 2). Unlike the vascular plant data, the same pattern of results was obtained for the other four indices; this is an outcome of there being proportionately more rare bryophytes than rare vascular plants (Fig. 2).

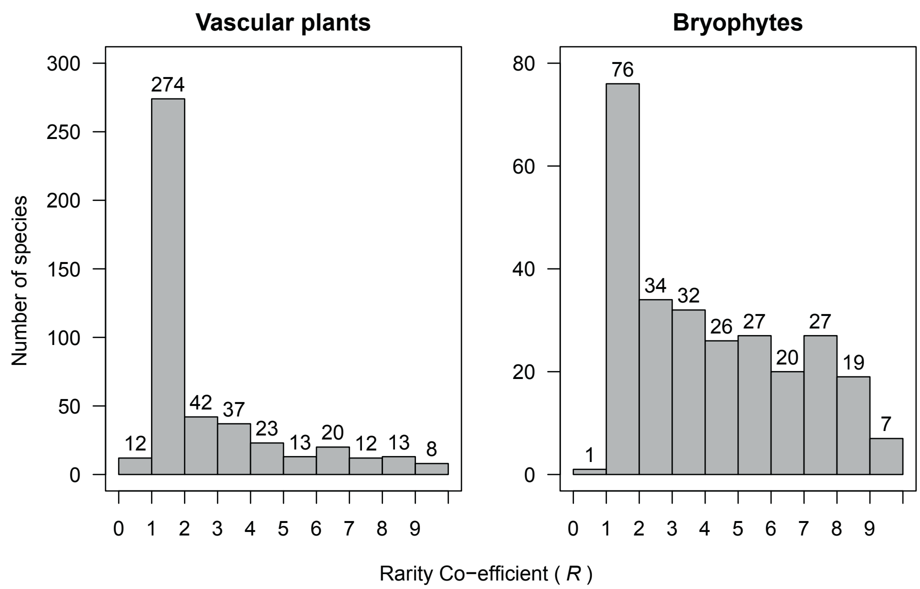

Fig. 2. Distribution of Rarity Co-efficient (R) values of the vascular plant and bryophytes species (native and non-native) within the stratified dataset.

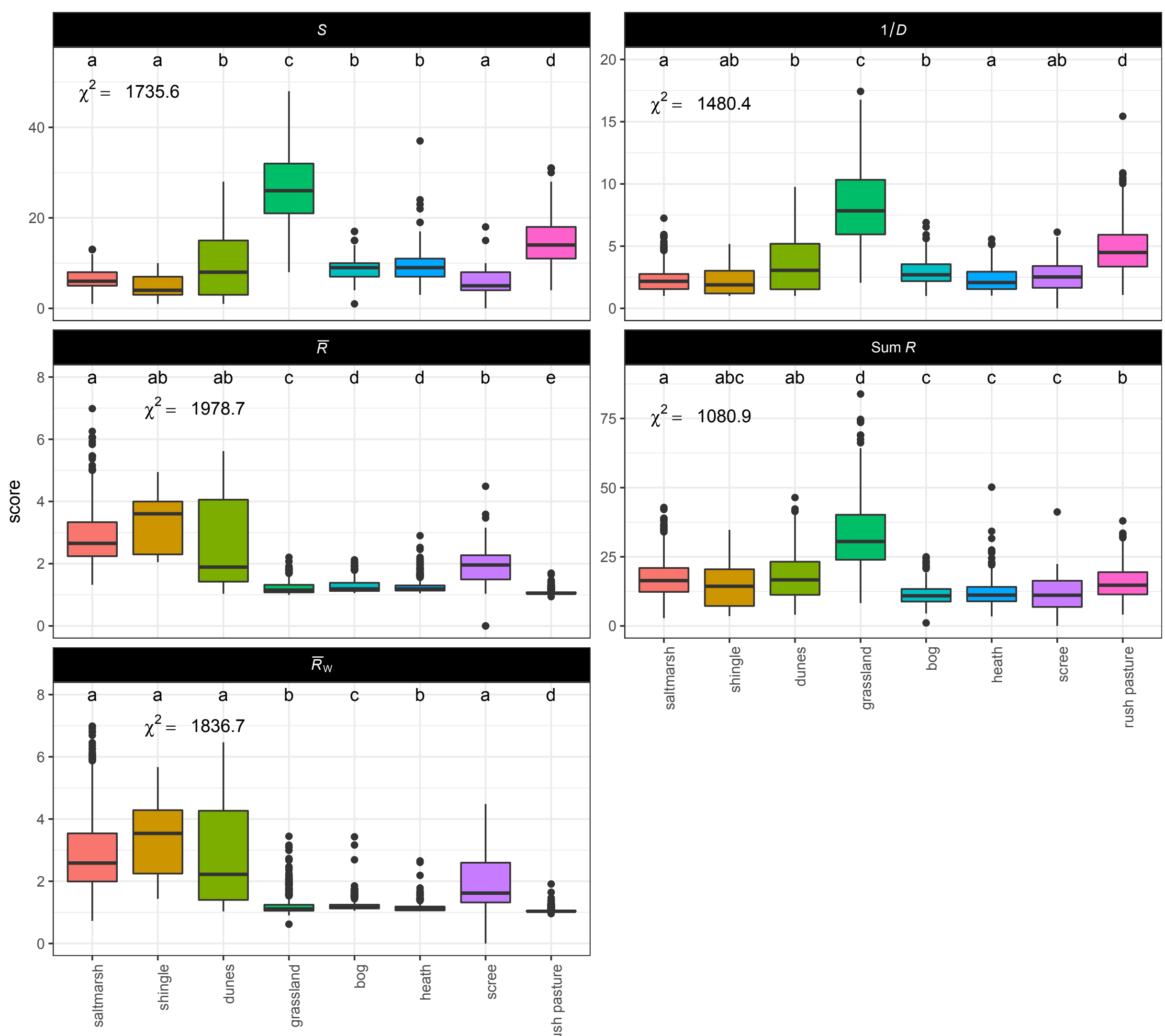

Fig. 3. Kruskal-Wallis rank sum test analysis of five indices applied to vascular plant data from eight ecosystems. Significant differences among ecosystems were detected for each index (χ^2 as shown in each panel, d.f. = 6, $p < 0.001$). For each index, ecosystems with different lower-case letters are significantly different ($p < 0.05$) according to *post hoc* pairwise comparisons adjusted for multiple comparisons.

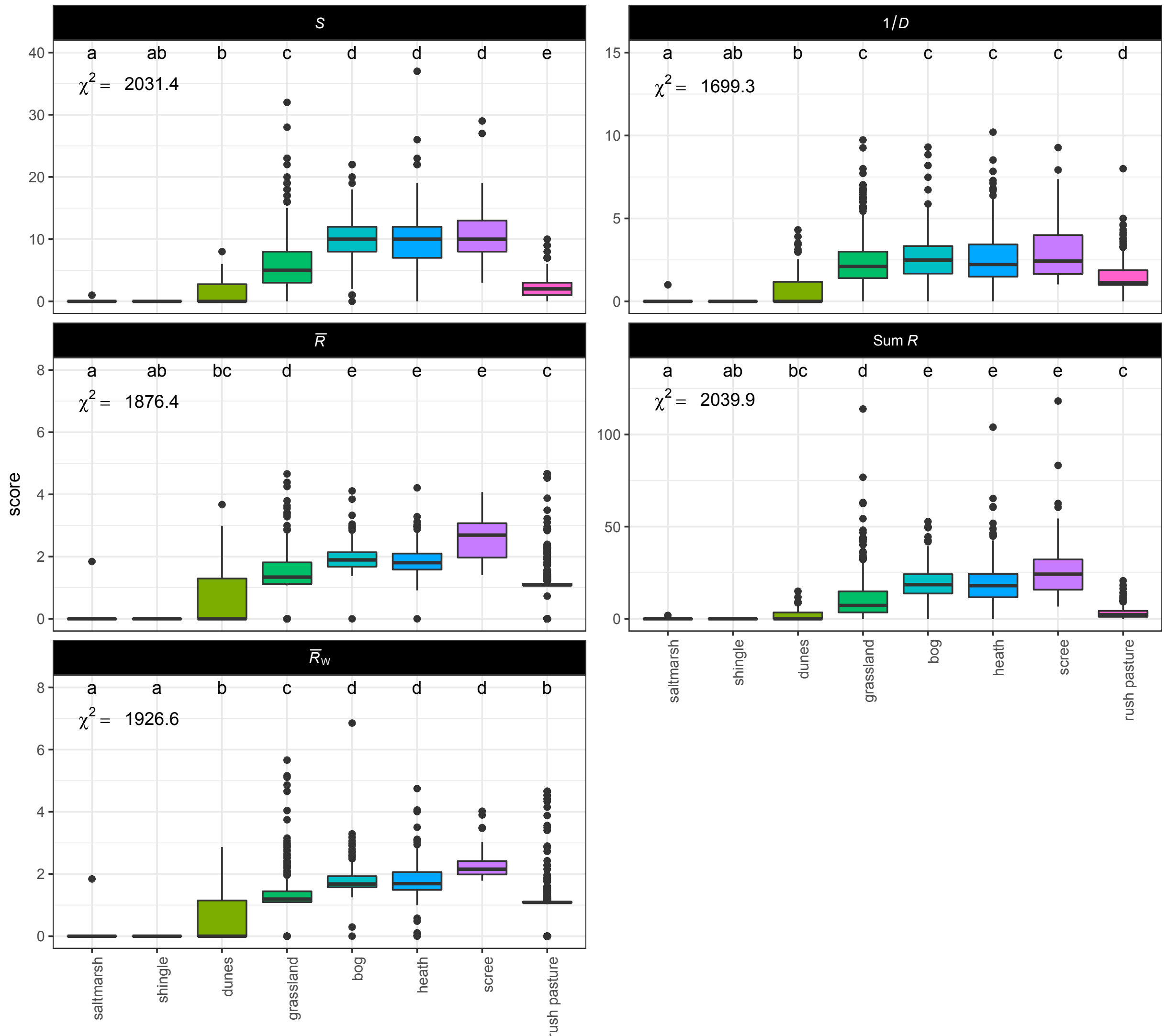

Fig. 4. Kruskal-Wallis rank sum test analysis of five indices applied to bryophyte plant data from eight ecosystems. Significant differences among ecosystems were detected for each index (χ^2 as shown in each panel, d.f. = 6, $p < 0.001$). For each index, ecosystems with different lower-case letters are significantly different ($p < 0.05$) according to *post hoc* pairwise comparisons adjusted for multiple comparisons.

3.3. Habitat-level analysis

The *k*-means procedure resulted in the optimal partitioning of habitats into three types based on their rankings (Table 3). Type A consisted of all shingle and saltmarsh habitats together with early successional dune habitats. These habitats typically lacked bryophytes. Rankings here based on vascular plant data were low for S and $1/D$, intermediate for Sum R, but high for both $\bar{R}$ and $\bar{R}_w$, therefore we categorised this coastal type as being one of *habitats with specialist vascular plants.* Type B consisted of the late successional dune habitats, the four Annex I grassland habitats and rush pasture. Rankings based on vascular plant data were typically high for S, $1/D$ and Sum R, and low to intermediate for $\bar{R}$ and $\bar{R}_w$, whilst rankings based on bryophyte data were intermediate. This group we thus categorised as one of *species-rich vascular plant habitats*. Type C contained the bog, heath and scree habitats which were ranked highly based on bryophyte species richness and rarity and lowly or intermediately based on vascular plant data. We therefore categorised type C as one of *important bryophyte habitats*.

There was significant concordance between the rankings of S, $1/D$ and Sum R and between $\bar{R}$ and $\bar{R}_w$ for vascular plant data, while rankings of all five indices applied to the bryophyte data were also significantly concordant (Table 4). There was, however, almost no significant concordance between the rankings for vascular plants and those for bryophytes. These results are related in part to the differences in the distributions of R values for vascular plants and bryophytes (Fig. 2).

3.4. Community-level analysis

Saltmarsh plots were classified to seventeen communities of the IVC (Fig. 5; Table 5), belonging to six IVC groups, SM1-6, which sequentially represent a general progression from lower marsh to upper marsh vegetation. As bryophytes were so rare in this ecosystem, we present index values for vascular plants only. There were trends for S to increase and $\bar{R}$ to decrease up the saltmarsh elevation gradient, resulting in the highest levels of Sum R being at intermediate elevations in the SM2 *Puccinellia maritima – Spergularia media* group (highest median Sum R = 25.54 in SM2C). A comparison of $\bar{R}$ and $\bar{R}_w$ indicates noteworthy effects of the application of abundance weighting, with a reduction in the index scores of communities typically dominated by non-native or nationally common species (e.g. *Spartina anglica* C.E. Hubb. in SM1B, *Festuca rubra* in SM4D and *Agrostis stolonifera* L. in SM6B) but an increase in the index scores of communities characteristically dominated by saltmarsh specialist species (e.g. *Salicornia* L. spp. in SM1A,

Table 3
Ranking of habitats for each of the five indices applied to vascular plant and bryophyte data. Green cells indicate higher rankings, yellow cells indicate intermediate rankings and red cells indicate lower rankings. Dashed lines indicate optimal partitioning by *k*-means clustering with type descriptions in the right-most column. Rush indicates rush pasture, other habitat codes are explained in Table 1.

	Vascular plants					Bryophytes					
Habitat	S	$1/D$	Sum R	$\bar{R}$	$\bar{R}_w$	S	$1/D$	Sum R	$\bar{R}$	$\bar{R}_w$	*k*-means group
1210	23	22	19	5	5	21	21	21	21	21	(A) Habitats with specialist vascular plants
1220	18	11	10	7	6	21	21	21	21	21	
1310	24	23	21	1	1	21	21	21	21	21	
1330	19	18	11	6	8	21	21	21	21	21	
1410	17	16	12	8	7	17	17	17	17	17	
1420	15	10	2	2	4	21	21	21	21	21	
2110	22	20	15	3	2	21	21	21	21	21	
2120	21	24	16	4	3	21	21	21	21	21	
2130*	7	9	7	10	10	14	14	12	11	11	(B) Species-rich vascular plant habitats
2170	9	7	9	14	12	12	9	13	13	12	
2190	8	6	8	15	15	16	16	16	15	14	
21A0*	5	4	3	12	16	11	8	10	10	4	
6410	2	2	4	22	22	10	10	11	12	13	
6510	3	3	5	23	23	13	13	14	14	15	
Rush	6	8	13	24	24	15	15	15	16	16	
6230*	4	5	6	21	18	4	5	5	7	10	
6210*	1	1	1	16	14	8	3	8	8	8	
4010	10	13	17	20	20	2	2	2	5	5	(C) Important bryophyte habitats
4030	14	21	24	19	21	5	6	7	9	9	
4060	16	19	20	11	17	7	12	6	3	3	
7130*	13	12	23	18	19	3	4	3	6	6	
7140	12	14	18	17	13	9	11	9	4	7	
7150	11	17	14	13	11	6	7	4	2	2	
8110	20	15	22	9	9	1	1	1	1	1	

Atriplex portulacoides in SM2B and *Juncus maritimus* Lam. in SM5B). Dominance of *Salicornia* spp. in SM1A, *Puccinellia maritima* (Huds.) Parl. in SM2D and *Festuca rubra* in SM4D resulted in these communities having the lowest measures of Simpson's diversity.

4. Discussion

Several forms of rarity have been recognised based on three factors: (i) geographic distribution (large or small), (ii) habitat specificity (wide or narrow) and (iii) local population density (locally abundant or sparse) (Rabinowitz, 1981; Rabinowitz et al., 1986). R values are a measure of geographic distribution, possibly conflated with elements of habitat specificity when applied, as here, at the hectad level, whilst by applying abundance-weighting local population density is also accounted for.

Mathematically, the approach of the rarity indices is similar to that applied in Floristic Quality Assessments (FQAs), which are widely used in the United States, but it differs conceptually in that the co-efficient R is a measure of rarity reflecting conservation value, whereas FQAs use C, a co-efficient reflecting tolerance to habitat degradation and faithfulness to natural habitat remnants (Freyman et al., 2016).

There has been little published in the context of Irish vegetation ecology on using quantitative approaches, such as those presented here, to evaluate conservation value. One exception is the work of Walsh et al. (2019). They used national grid square frequency to derive, for individual species, a measure of conservation value, but weighted this value by a measure of perceived conservation importance (e.g. protected status). Their approach also differed significantly in that these values were applied to species presence records at a hectad or tetrad

Table 4
Kendall's co-efficient of concordance (W; corrected for tied ranks) between rankings of habitats for each of the five indices applied to vascular plant and bryophyte data. Underlined values indicate significant concordance ($p < 0.05$).

		Vascular plants					Bryophytes				
		S	$1/D$	Sum R	$\bar{R}$	$\bar{R}_w$	S	$1/D$	Sum R	$\bar{R}$	$\bar{R}_w$
Vascular plants	S	1.00									
	$1/D$	0.95	1.00								
	Sum R	0.83	0.89	1.00							
	$\bar{R}$	0.10	0.20	0.43	1.00						
	$\bar{R}_w$	0.12	0.22	0.45	0.98	1.00					
Bryophytes	S	0.73	0.64	0.39	0.17	0.17	1.00				
	$1/D$	0.77	0.70	0.46	0.17	0.18	0.98	1.00			
	Sum R	0.72	0.63	0.40	0.19	0.19	1.00	0.98	1.00		
	$\bar{R}$	0.71	0.61	0.40	0.22	0.22	0.97	0.95	0.98	1.00	
	$\bar{R}_w$	0.71	0.62	0.42	0.25	0.23	0.96	0.95	0.98	0.99	1.00

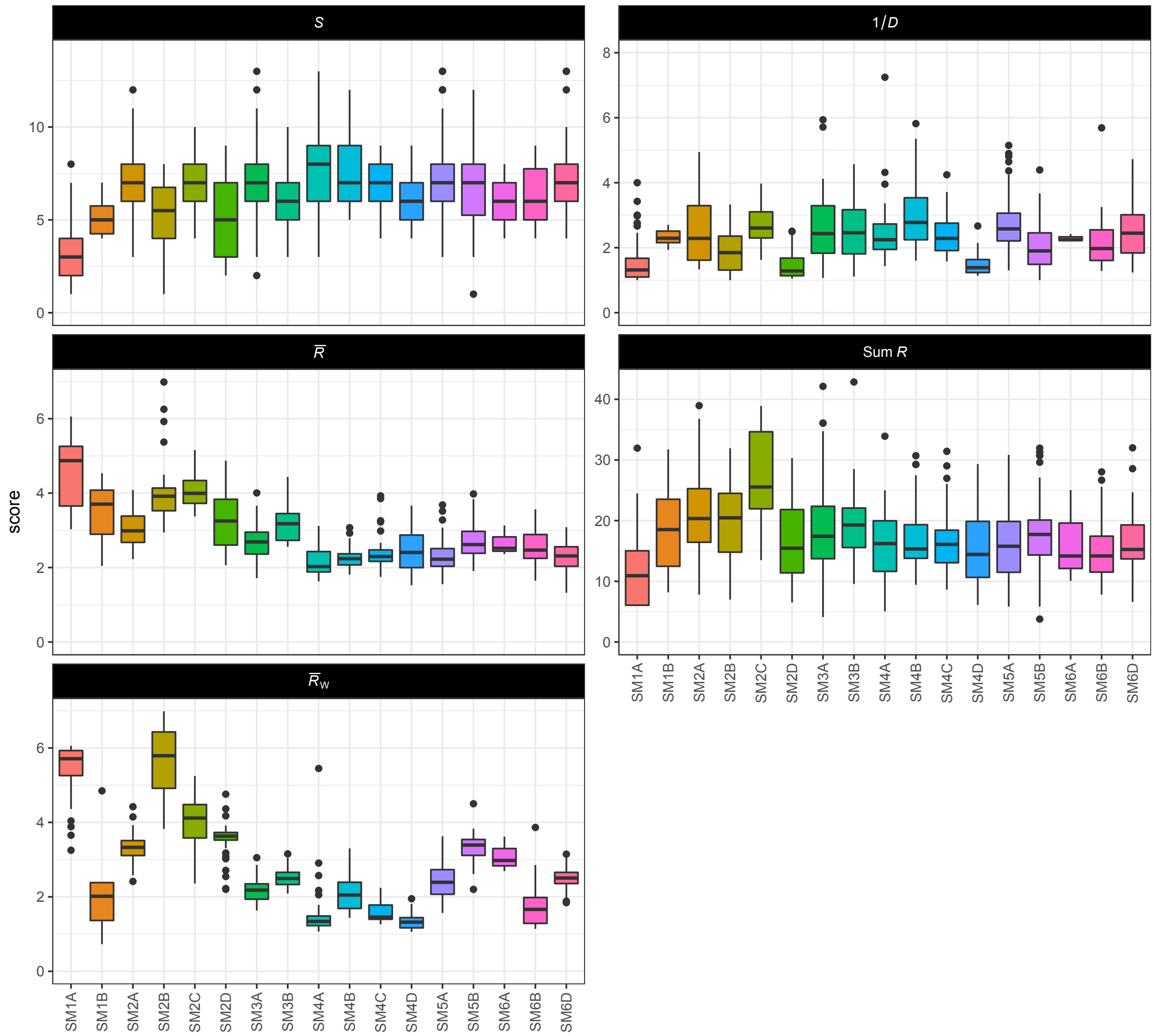

Fig. 5. Boxplot of five indices applied to vascular plant data from seventeen saltmarsh communities defined by the Irish Vegetation Classification. For names of communities, see Table 5.

Table 5
Names of Irish Vegetation Classification communities to which saltmarsh data were classified by the ERICA application.

Code	Irish Vegetation Classification community
SM1A	*Salicornia* agg. saltmarsh
SM1B	*Spartina* agg. saltmarsh
SM2A	*Puccinellia maritima – Glaux maritima* saltmarsh
SM2B	*Atriplex portulacoides – Puccinellia maritima* saltmarsh
SM2C	*Puccinellia maritima – Limonium humile* saltmarsh
SM2D	*Puccinellia maritima – Aster tripolium* saltmarsh
SM3A	*Plantago maritima – Armeria maritima* saltmarsh
SM3B	*Plantago maritima – Puccinellia maritima* saltmarsh
SM4A	*Festuca rubra – Agrostis stolonifera* saltmarsh
SM4B	*Festuca rubra – Juncus gerardii* saltmarsh
SM4C	*Festuca rubra – Armeria maritima* saltmarsh
SM4D	*Festuca rubra – Plantago maritima* saltmarsh
SM5A	*Juncus maritimus – Festuca rubra* saltmarsh
SM5B	*Juncus maritimus – Plantago maritima* saltmarsh
SM6A	*Bolboschoenus maritimus – Agrostis stolonifera* saltmarsh-swamp
SM6B	*Agrostis stolonifera – Triglochin maritimum* saltmarsh
SM6D	*Agrostis stolonifera – Juncus gerardii* saltmarsh

scale, rather than presence or abundance at a plot scale; thus they could not evaluate conservation value of specific communities, habitats or ecosystems.

Our study demonstrates that the conservation valuation of habitats may differ depending upon whether one employs measures reflecting species richness only, species rarity only, or both species richness and species rarity. Habitats 1420 Halophilous scrub and 6510 Lowland hay meadows make notable examples. Habitat 1420 is defined in Ireland by the presence of the very rare *Sarcocornia perennis* (Mill.) A.J. Scott (occurs in 4 hectads, $R = 9.79$), thus whilst it had only an intermediate rank based on vascular plant S, it was the second top-ranking habitat based on vascular plant $\bar{R}$ and on vascular plant Sum R. Somewhat conversely, for vascular plants, habitat 6510 was the third most species-rich habitat but it came second last in terms of $\bar{R}$ as it is chiefly composed of generalist species; niche specialists such as *Fritillaria meleagris* L., a characteristic species of this habitat in the UK, are absent from the Irish flora (Preston et al., 2002). Similarly, rush pastures are reasonably species-rich but contain few rare species, thereby underpinning their lack of conservation value.

Application of these indices at ecosystem, habitat and community levels revealed informative patterns of results, particularly where categories represented successional gradients. The dune ecosystem displayed a larger degree of variation in $\bar{R}$ and $\bar{R}_w$ than other ecosystems. This was due to this grouping containing both early successional dune habitats of type A, characterised by a few specialist species, and late successional dune habitats of type B, with lower average specialism.

Community-level results for saltmarsh demonstrated that the application of abundance-weighting can have a significant effect on indices. Furthermore, since habitat 1330 Atlantic salt meadows comprises vegetation from IVC groups SM2, SM3, SM4 and SM6 with *Spartina anglica*-invaded variants being classified as community SM1B, the results also evince that significant variation in richness and rarity can occur within a single Annex I habitat.

We found a general lack of concordance between rankings based on vascular plant data and those based on bryophyte data. Contrasting trends in the species richness of these taxonomic groups have been found by several other studies (e.g. Chiarucci et al., 2007; Grytnes et al., 2006; Santi et al., 2010; Virtanen and Crawley, 2010). In the habitats we analysed, these taxonomic groups are thus not suitable as surrogates for each other and this should be reflected in monitoring procedures. Delaney and Stout (2018) demonstrated the problems of using a single taxonomic group for monitoring habitat condition in the case of Irish dune slacks, finding that a plant-based monitoring system did not predict the diversity or composition of snail and water beetle communities.

We also found a clear pattern of results at the habitat level. Based on our categorisation of habitat conservation value, we propose that assessment criteria should include an appraisal of the presence of vascular plant specialists (e.g. halophytes, sand binders) for type A habitats, vascular plant species richness for type B habitats and the bryophytic flora for type C habitats.

A fourth category, type D, could be surmised, comprising *low conservation value habitats* with both low species richness and low species rarity; improved pasturage and silage fields, rank ruderal communities, habitats heavily invaded by non-native species and some forms of scrub could be included here and should be the focus of further research.

Our categorisation evidently only encapsulates certain aspects of conservation value. Firstly, there may be an inherent value in some habitats due to their threatened status, regardless of the rarity or richness of the vegetation constituents. Secondly, it does not account for value represented by other taxa (e.g. invertebrates, macrolichens or birds). For example, habitats 4010 Wet heaths, 4030 Dry heaths and 7130* Blanket bogs (active), which were all ranked very poorly on the basis of Sum R for vascular plants but highly on the basis of Sum R for bryophytes, are also important habitats in Ireland for red grouse *Lagopus lagopus scotica* (Latham, 1787) and hen harrier *Circus cyaneus* (Linnaeus, 1766), the latter a species listed on Annex I of the Birds Directive (2009/147/EEC) (Ruddock et al., 2016). Such other taxa could be assessed in a similar fashion if distribution records are sufficient to provide a reasonable estimate of true R for each species, although in the case of birds, one would need to account for the potential use of different ranges of habitats in different seasons. Thirdly, this categorisation does account for ecosystem services other than the provision of suitable habitats. For example, through the functions of wave attenuation, sedimentation and biological productivity, saltmarshes provide coastal protection and carbon sequestration (Barbier et al., 2011). Note also that habitats have been selected for inclusion under Annex I based on an assessment at the level of the European Union (European Commission, 2013) whereas our rarity indices are based on national distribution of plant species. The approach could, however, be applied at a variety of spatial scales (e.g. regional, national or continental) if adequate data were available.

5. Conclusion

This approach provides for the integration of species rarity into biodiversity assessments and could be readily applied in other countries or regions based on localised datasets. Once R values have been derived for the list of relevant species the related indices are straightforward to calculate and allow the conservation value of vegetation samples to be objectively assessed in the context of national rarity, although it should be noted that measures of abundance at the level of the samples are required to calculate $\bar{R}_w$. These rarity indices also have the advantage of being conceptually simple to interpret. Each of the five indices tested provides different information, which may be more or less useful, depending upon the aims of a particular study. Site conservation value investigations at the plot or habitat scale should, however, incorporate an assessment of species rarity, such as $\bar{R}$ or $\bar{R}_w$, as, based on our results, these may elucidate aspects of conservation value which contrast with species richness.

CRediT authorship contribution statement

Philip M. Perrin: Conceptualization, Methodology, Investigation, Formal analysis, Visualization, Writing - original draft, Writing - review & editing, Funding acquisition, Project administration. **Stephen Waldren:** Conceptualization, Writing - original draft, Writing - review & editing, Funding acquisition.

Declaration of Competing Interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2020.106387.

References

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. https://doi.org/10.1890/10-1510.1.

Blockeel, T.L., Bosanquet, S.D.S., Hill, M.O., Preston, C.D., Society, B.B., 2014. Atlas of British & Irish Bryophytes: The Distribution and Habitat of Mosses and Liverworts in Britain and Ireland. Pisces Publications, Newbury.

Chiarucci, A., D'auria, F., Bonini, I., 2007. Is vascular plant species diversity a predictor of bryophyte species diversity in Mediterranean forests? Biodivers. Conserv. 16, 525–545. https://doi.org/10.1007/s10531-006-6733-1.

Chiarucci, A., Maccherini, S., De Dominicis, V., 2001. Evaluation and monitoring of the flora in a nature reserve by estimation methods. Biol. Conserv. 101, 305–314. https://doi.org/10.1016/S0006-3207(01)00073-8.

Costanza, R., Fisher, B., Mulder, K., Liu, S., Christopher, T., 2007. Biodiversity and ecosystem services: a multi-scale empirical study of the relationship between species richness and net primary production. Ecol. Econ. 61, 478–491. https://doi.org/10.1016/j.ecolecon.2006.03.021.

Daan, N., 2001. A spatial and temporal diversity index taking into account species rarity, with an application to the North Sea fish community. Presented at the 2001 Annual Conference Meeting, Oslo, International Council for the Exploration of the Sea, Copenhagen, Denmark.

Davies, B., Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Sear, D., Bray, S., Maund, S., 2008. Comparative biodiversity of aquatic habitats in the European agricultural landscape. Agric. Ecosyst. Environ. 125, 1–8. https://doi.org/10.1016/j.agee.2007.10.006.

De Cáceres, M., Font, X., Oliva, F., 2010. The management of vegetation classifications with fuzzy clustering. J. Veg. Sci. 21, 1138–1151. https://doi.org/10.1111/j.1654-1103.2010.01211.x.

Delaney, A., Devaney, F.M., Martin, J.R., Barron, S.J., 2013. Monitoring survey of Annex I sand dune habitats in Ireland, Irish Wildlife Manuals (No. 75). National Parks & Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin, Ireland.

Delaney, A., Stout, J.C., 2018. Principles of cross congruence do not apply in naturally disturbed dune slack habitats: Implications for conservation monitoring. Ecol. Ind. 93, 358–364. https://doi.org/10.1016/j.ecolind.2018.04.075.

European Commission, 2013. Interpretation manual of European Union habitats EUR 28. DG Environment, Brussels.

Fleishman, E., Noss, R.F., Noon, B.R., 2006. Utility and limitations of species richness metrics for conservation planning. Ecol. Ind. 6, 543–553. https://doi.org/10.1016/j.ecolind.2005.07.005.

Freeman, M.C., Hagler, M.M., Bumpers, P.M., Wheeler, K., Wenger, S.J., Freeman, B.J., 2017. Long-term monitoring data provide evidence of declining species richness in a river valued for biodiversity conservation. J. Fish Wildlife Manage. 8, 418–434. https://doi.org/10.3996/122016-JFWM-090.

Freyman, W.A., Masters, L.A., Packard, S., 2016. The Universal Floristic Quality Assessment (FQA) Calculator: an online tool for ecological assessment and monitoring. Methods Ecol. Evol. 7, 380–383. https://doi.org/10.1111/2041-210X.12491.

García, D., Martínez, D., 2012. Species richness matters for the quality of ecosystem services: a test using seed dispersal by frugivorous birds. Proc. R. Soc. London B 279, 3106–3113. https://doi.org/10.1098/rspb.2012.0175.

Giraudoux, P., 2017. pgirmess: Data Analysis in Ecology, R package. https://CRAN.R-project.org/package=pgirmess.

Godefroid, S., Koedam, N., 2007. Urban plant species patterns are highly driven by density and function of built-up areas. Landscape Ecol. 22, 1227–1239. https://doi.org/10.1007/s10980-007-9102-x.

Gouhier, T.C., Guichard, F., 2014. Synchrony: quantifying variability in space and time. Methods Ecol. Evol. 5, 524–533. https://doi.org/10.1111/2041-210X.12188.

Grytnes, J.A., Heegaard, E., Ihlen, P.G., 2006. Species richness of vascular plants, bryophytes, and lichens along an altitudinal gradient in western Norway. Acta Oecologica 29, 241–246. https://doi.org/10.1016/j.actao.2005.10.007.

Hill, M.O., Preston, C.D., Roy, D.B., 2004. PLANTATT. Attributes of British and Irish Plants: Status, Size, Life History, Geography and Habitats. Centre for Ecology and Hydrology, Huntingdon.

Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Monogr. 54, 187–211. https://doi.org/10.2307/1942661.

Mace, G.M., Norris, K., Fitter, A.H., 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends Ecol. Evol. 27, 19–26. https://doi.org/10.1016/j.tree.2011.08.006.

Maestre, F.T., Quero, J.L., Gotelli, N.J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M.A., Soliveres, S., Escolar, C., García-Palacios, P., Berdugo, M., Valencia, E., Gozalo, B., Gallardo, A., Aguilera, L., Arredondo, T., Blones, J., Boeken, B., Bran, D., Conceição, A.A., Cabrera, O., Chaieb, M., Derak, M., Eldridge, D.J., Espinosa, C.I., Florentino, A., Gaitán, J., Gatica, M.G., Ghiloufi, W., Gómez-González, S., Gutiérrez, J.R., Hernández, R.M., Huang, X., Huber-Sannwald, E., Jankju, M., Miriti, M., Monerris, J., Mau, R.L., Morici, E., Naseri, K., Ospina, A., Polo, V., Prina, A., Pucheta, E., Ramírez-Collantes, D.A., Romão, R., Tighe, M., Torres-Díaz, C., Val, J., Veiga, J.P., Wang, D., Zaady, E., 2012. Plant species richness and ecosystem multifunctionality in global drylands. Science 335, 214–218. https://doi.org/10.1126/science.1215442.

McCorry, M., 2007. Saltmarsh Monitoring Project 2006. Unpublished report. National Parks & Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.

McCorry, M., Ryle, T., 2009. Saltmarsh Monitoring Project 2007-2008, Volume I (Final). Unpublished report. National Parks & Wildlife Service, Department of Environment, Heritage and Local Government, Dublin, Ireland.

Moran, M.D., 2003. Arguments for rejecting the sequential Bonferroni in ecological studies. Oikos 100, 403–405. https://doi.org/10.1034/j.1600-0706.2003.12010.x.

Nakagawa, S., 2004. A farewell to Bonferroni: the problems of low statistical power and publication bias. Behav. Ecol. 15, 1044–1045. https://doi.org/10.1093/beheco/arh107.

NPWS, 2019. The status of EU protected habitats and species in Ireland. Volume 1: Summary overview. National Parks & Wildlife Service, Department of Culture, Heritage and the Gaeltacht, Dublin.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., Wagner, H., 2017. vegan: community ecology package. R package. https://CRAN.R-project.org/package=vegan.

O'Neill, F.H., Martin, J.R., Devaney, F.M., Perrin, P.M., 2013. The Irish Semi-natural Grasslands Survey 2007-2012, Irish Wildlife Manuals (No. 78). National Parks & Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin, Ireland.

Orme, C.D.L., Davies, R.G., Burgess, M., Eigenbrod, F., Pickup, N., Olson, V.A., Webster, A.J., Ding, T.-S., Rasmussen, P.C., Ridgely, R.S., 2005. Global hotspots of species richness are not congruent with endemism or threat. Nature 436, 1016–1019. https://doi.org/10.1038/nature03850.

Paillet, Y., Bergès, L., Hjältén, J., Ódor, P., Avon, C., Bernhardt-Römmerman, M., Bijlsma, R.-J., De Bruyn, L., Fuhr, M., Grandin, U., Kanka, R., Lundin, L., Luque, S., Magura, T., Matesanz, S., Mészáros, I., Sebastià, M.-T., Schmidt, W., Standovár, T., Tóthmérész, B., Uotila, A., Valladares, F., Vellak, K., Virtanen, R., 2010. Biodiversity differences between managed and unmanaged forests: meta-analysis of species richness in Europe. Conserv. Biol. 24, 101–112. https://doi.org/10.1111/j.1523-1739.2009.01399.x.

Penk, M.R., Perrin, P.M., Kelly, R., O'Neill, F., Waldren, S., 2020. Plant diversity and community composition in temperate northeast Atlantic salt marshes are linked to nutrient concentrations. Appl. Veg. Sci. 23, 3–13. https://doi.org/10.1111/avsc.12459.

Perrin, P.M., Roche, J.R., Barron, S.J., Daly, O.H., Hodd, R.L., Devaney, F.M., 2014. National Survey of Upland Habitats (Phase 4, 2013-2014), Site Report No. 16: Caha Mountains cSAC (000093), Cos. Cork and Kerry. Unpublished Report. National Parks & Wildlife Service, Department of Arts, Heritage and the Gaeltacht, Dublin, Ireland.

Perrin, P.M., FitzPatrick, Ú., Lynn, D., 2018. The Irish Vegetation Classification – an overview of concepts, structure and tools. Bulletin of the Chartered Institute of Ecology and Environmental Management – In Practice 102, 14–19.

Preston, C.D., Pearman, D.A., Dines, D.T., 2002. New Atlas of the British and Irish Flora. Oxford University Press, Oxford.

R Core Team, 2018. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Rabinowitz, D., 1981. Seven forms of rarity, in: Synge, H. (Ed.), The Biological Aspects of Rare Plant Conservation. John Wiley & Sons, Chichester.

Rabinowitz, D., Cairns, S., Dillon, T., 1986. Seven forms of rarity and their frequency in the flora of the British Isles, in: Soule, M. (Ed.), Conservation Biology: The Science of Scarcity and Diversity. Sinauer Associates, Sunderland, pp. 1882–1204.

Ruddock, M., Mee, A., Lusby, J., Nagle, T., O'Neill, S., O'Toole, L., 2016. The 2015 national survey of breeding hen harrier in Ireland (No. 93), Irish Wildlife Manuals. National Parks & Wildlife Service, Department of the Arts, Heritage and the Gaeltacht, Dublin, Ireland.

Samu, F., Csontos, P., Szinetár, C., 2008. From multi-criteria approach to simple protocol: assessing habitat patches for conservation value using species rarity. Biol. Conserv. 141, 1310–1320. https://doi.org/10.1016/j.biocon.2008.03.015.

Santi, E., Maccherini, S., Rocchini, D., Bonini, I., Brunialti, G., Favilli, L., Perini, C., Pezzo, F., Piazzini, S., Rota, E., 2010. Simple to sample: vascular plants as surrogate group in a nature reserve. J. Nat. Conserv. 18, 2–11. https://doi.org/10.1016/j.jnc.2009.02.003.

Virtanen, R., Crawley, M.J., 2010. Contrasting patterns in bryophyte and vascular plant species richness in relation to elevation, biomass and Soay sheep on St Kilda, Scotland. Plant Ecol. Divers 3, 77–85. https://doi.org/10.1080/17550874.2010.492403.

Walsh, A., Sullivan, C.A., Waldren, S., Finn, J.A., 2019. Development of a scoring method to identify important areas of plant diversity in Ireland. J. Nat. Conserv. 47, 1–11. https://doi.org/10.1016/j.jnc.2018.10.002.

Wheeler, B.D., 1988. Species richness, species rarity and conservation evaluation of rich-fen vegetation in lowland England and Wales. J. Appl. Ecol. 331–352.

Williams, P., 2000. Some properties of rarity scores used in site quality assessment. Br. J. Entomol. Nat. History 13, 73–86.

Williams, P., Whitfield, M., Biggs, J., Bray, S., Fox, G., Nicolet, P., Sear, D., 2004. Comparative biodiversity of rivers, streams, ditches and ponds in an agricultural landscape in Southern England. Biol. Conserv. 115, 329–341. https://doi.org/10.1016/S0006-3207(03)00153-8.