Acoustic Complexity Index to assess Benthic Biodiversity of a Partially Protected Area in the Southwest of the UK.

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Highlights:

- Acoustic Complexity Index higher in fished area vs protected area
- Acoustic Complexity Index does not correlate with simultaneous biodiversity indices
- Acoustic Complexity Index covaries with mobile assemblage composition

Abstract

The soundscape of the marine environment is a relatively understudied area of ecology that has the potential to provide large amounts of information on biodiversity, reproductive behaviour, habitat selection, spawning and predator-prey interactions. Biodiversity is often visually assessed and used as a proxy for ecosystem health. Visual assessment using divers or remote video methods can be expensive, and limited to times of good weather and water visibility. Previous studies have concluded that acoustic measures, such as the Acoustic Complexity Index (ACI), correlate with visual biodiversity estimates and offer an alternative to assess ecosystem health.

Here, the ACI measured over 5 years in a Marine Protected Area (MPA) in the UK, Lyme Bay, was analysed alongside another monitoring method, Baited Remote Underwater Video Systems (BRUVs). Two treatments were sampled annually in the summer from 2014 until 2018 with sites inside the MPA, as well as Open Control sites outside of the MPA.

Year by year correlations, which have been used elsewhere to test ACI, showed significant correlations with Number of Species and ACI. However, the sign of these correlations changed almost yearly, showing that more in-depth analyses are needed.

Multivariate analysis of the benthic assemblage composition (from BRUVs) was carried out by Permutational Multivariate Analysis of Variance (PERMANOVA) using Distance Matrices. Although not consistently correlating with univariate measures, the ACI was significantly interacting with the changing benthic assemblage composition, as it changed over time and protection (Inside vs Outside the MPA).

ACI showed potential to allude to shifting benthic communities, yet with no consistency when used alongside univariate measures of diversity. Although it is not without its own disadvantages, and thus should be developed further before implementation, the ACI could potentially reflect more complex changes to the benthos than simply the overall diversity.

Keywords: Acoustic Complexity Index, Biodiversity, Marine Protected Area, Monitoring Tools

1. Introduction

Biodiversity provides a useful measure to assess ecosystem health (Worm et al., 2006), and is increasingly being used for conservation and monitoring purposes, with an observed decrease used as a proxy for a degraded or negatively impacted ecosystem (Wabnitz et al., 2018). To quantify and compare these changes in diversity, many univariate indices have been produced, which simplify an assemblage of taxa into a single value. The most commonly used indices involve integrating the number of species present with measures of how the species are distributed within the assemblages, such as Number of Species (Kaplan et al., 2015; Pieretti and Farina, 2013; Sheehan et al., 2013b), Shannon-Wiener's diversity index (De-La-Ossa-Carretero et al., 2012), Simpson's diversity index (Miralles et al., 2016; Rombouts et al., 2019) and taxonomic distinctness (Clarke and Warwick, 2001; Leonard et al., 2006), which also involves phylogenetic distance.

Historic methods for assessing marine biodiversity have often used destructive practices (Francour, 1994; Lipej et al., 2003), such as poisoning (Diamant et al., 1986) or trawling (Cappo et al., 2004). However, for the study of recovering and fragile benthic systems, such as those in Marine Protected Areas (MPAs), non-invasive, non-extractive methods such as Underwater Visual Census (UVC) or Underwater Video Survey (UVS) are considered more appropriate (Sheehan et al., 2013a, 2010). Visual methods will always have the drawback that there is no physical sample taken, although image libraries give a permanent record, and thus those species that are harder to identify visually will always be under-sampled; yet this lack of physical sample means the populations being researched are almost or completely unaffected by the survey taken. A potential addition to supplement visual survey would be the assessment of the marine soundscape (Staaterman et al., 2017). This method for sampling the marine environment is similarly non-extractive and non-invasive, while sampling components of the ecosystem potentially under-represented by visual methods alone.

The marine soundscape comprises both natural and anthropogenic elements. Assessment of the biological element (biophony) of the marine soundscape has been used to describe overall biodiversity (Bertucci et al., 2016), reproductive behaviour (de Jong et al., 2018), habitat selection (Vermeij et al., 2010), spawning (Casaretto et al., 2014; Hawkins and Amorim, 2000) and predator-prey interactions (Bernasconi et al., 2011; Giorli et al., 2016). Biophony is produced by a wide range of taxa ranging from large cetaceans producing low frequency (~20Hz) calls or songs (Samaran et al., 2013), that can be detected up to thousands of kilometres away (Rivers, 1997), to crustaceans creating loud (190 dB re 1 μPa), broadband (2kHz up to 300kHz) 'snaps' and 'pops' (Picciulin et al., 2013).

Acoustic indices have been developed and utilised in marine (Gordon et al., 2018; Harris et al., 2016; Nedelec et al., 2015; Pieretti et al., 2017; Trenkel et al., 2011) and terrestrial (Farina and Pieretti, 2014; Merchant et al., 2015; Pieretti et al., 2015, 2011; Pijanowski et al., 2011; Sueur et al., 2008b) environments to assess whole ecosystem biodiversity. The use of these acoustic indices is perceived to allow hidden or shy species, overlooked by other survey methods, to be accounted for (Staaterman et al., 2017). The ACI as set out in Pieretti *et al.* (2011) quantifies the relative change in sound intensity across all frequencies of a soundscape, while being minimally affected by constant anthropogenic noise. The ACI was developed on the assumption that with increased diversity of species, there would be an increase in the complexity of biological sound produced. So far, most analyses of ACI have shown a positive correlation with a variety of biodiversity indices (Bertucci et al., 2016; Harris et al., 2016; Meyer et al., 2018; Pieretti et al., 2015, 2011).

The two survey methods, visual and acoustic, are thought to complement each other by overlapping, as well as covering differing spatial scales and taxonomic groups (Staaterman et al., 2017). However, the majority of studies to date regarding this interaction have been based either in areas of very high biodiversity, such as coral reef systems (Bertucci et al., 2016; Kaplan et al., 2015), or only focused on fish diversity (Harris et al., 2016). As such, the transferability to other habitats and ecosystems is limited.

This study assessed the suitability of the ACI index derived from using acoustic recording as a monitoring method and to explore its relationship with seabed biodiversity. As such, a 5 year study within a recovering temperate reef seabed ecosystem was undertaken, in which were protected areas and those open to bottom fishing.

It was expected that the ACI and two visual biodiversity indices, Number of Species and Shannon's Diversity Index, derived from Baited Remote Underwater Video systems (BRUVs) data ('visual biodiversity indices' from now on), would increase over time in the MPA relative to the areas that continue to be fished. As a recovering system it would be predicted that the interaction of time and treatment would be significant. Therefore, the following hypotheses were assessed for inside vs outside the MPA:

1. The ACI would increase over time,
2. The visual biodiversity indices would increase over time,
3. The visual biodiversity indices and the ACI would correlate with each other over time,
4. Changes in the mobile benthic assemblage composition would result in similar changes to the ACI.

2. Methods

2.1 Study Location

Lyme Bay (Fig. 1), is located on the south coast of England, and contains areas of rocky reef habitat known to include nationally important fragile reef building species (Hiscock and Breckels, 2007). A Statutory Instrument (SI), a type of MPA, was established in 2008 in Lyme Bay. The SI excluded all towed demersal fishing equipment (scallop dredging and trawling) from a 206 km^2 area of the bay.

Experimental site selection was based on similar biotope classifications to negate any confounding effects of habitat heterogeneity (Claudet et al., 2008), with all sites being on either hard or 'mixed' substrate at similar depths (Sheehan et al., 2013b; Stevens et al., 2014). There were two treatments: Inside the MPA (n=12) and Outside the MPA (n=6). Geographically similar pairs of sites were grouped into 'Areas'.

In the winter of 2013/2014, the south coast of the UK experienced severe and unprecedented storm activity that was observed to have major impacts on South West England's coastal systems (Masselink et al., 2016). The effect to both the protected and non-protected ecosystems provided an opportunity to start a new monitoring strategy. This incorporated acoustic recording and assessment of the marine soundscape inside and outside of the MPA alongside visual measures of the seabed assemblage and allowed the assessment of the emerging acoustic analyses.

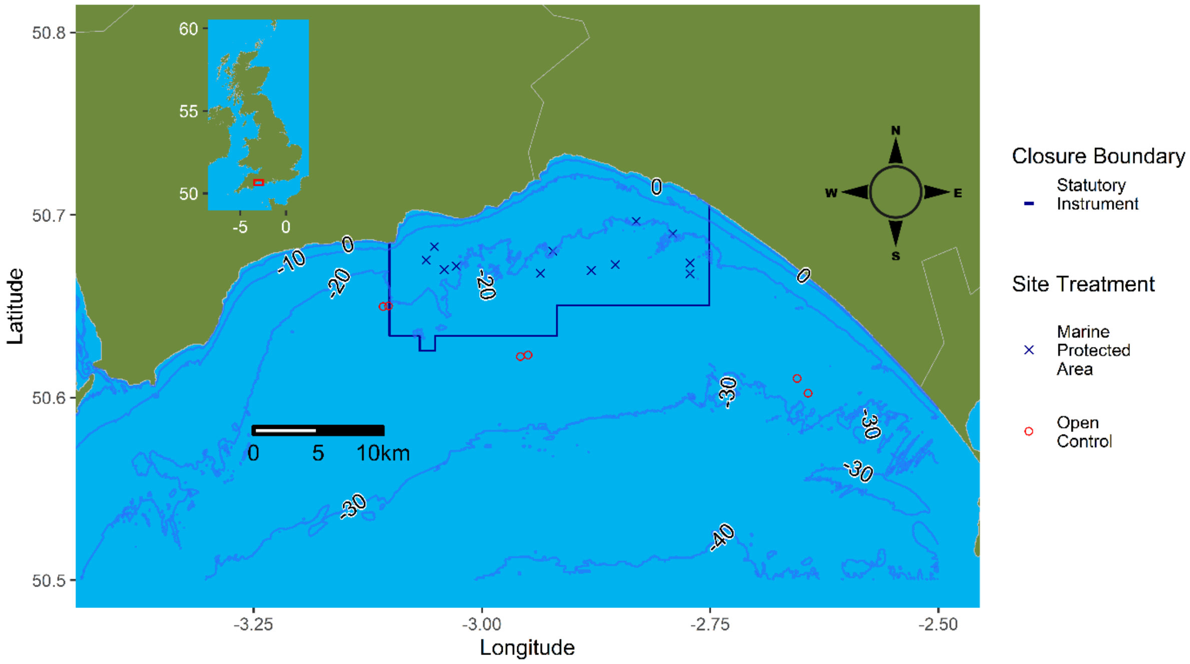

Fig. 1. Map of the UK inlaid in a map of Lyme Bay, showing site locations and their treatments (Blue Cross: Marine Protected Area, Red Circle: Open Control). Dark blue line shows the Statutory Instrument and light blue lines show 10m depth contours.

2.2 Data Collection

2.2.1 Acoustic Recorder Deployment

At each site, an acoustic recorder was attached and deployed with one of the three replicate BRUVs (Fig. 2). The acoustic recorders used were low power Digital SpectroGrams (DSG) (Hydrophone Calibration Sensitivity=-190dBV/uPa, Sample rate=50 kHz, Decimation Factor=4, System Gain=20, Effective Sample Range=0-25 kHz; Loggerhead Instruments, Sarasota, FL, USA), which were used to record DSG files on a duty cycle of 16 seconds recorded every 2 minutes to conserve battery life. The recorders were attached to one BRUVs for every site, to sample identical locations (Fig. 2), but, as

DSG acoustic recorders sample a larger area (Simard *et al.,* 2015) than the BRUVs maximum effective range of attraction (AR) (Cappo *et al.*, 2004), single acoustic recordings were used across the three BRUVs replicates (Fig. S1).

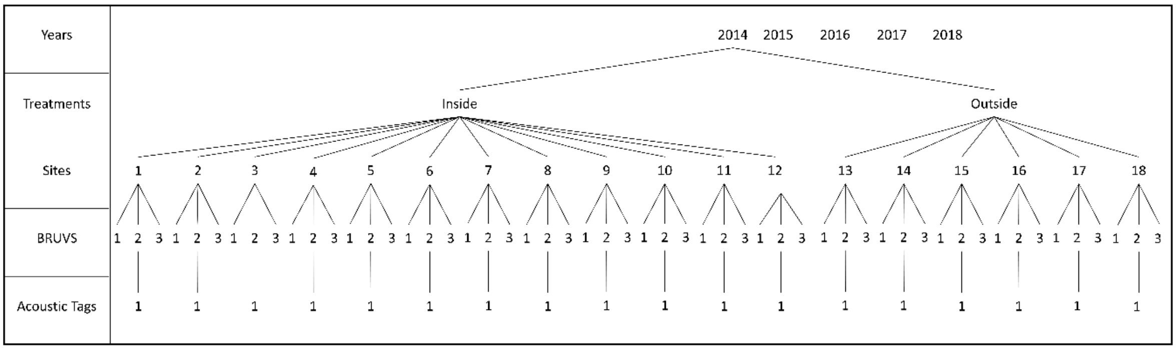

Fig. 2. Sampling design for BRUVs and Acoustic Recorders.

2.2.2 Acoustic File Extraction

For each deployment, audio recordings were prepared to ~360 second WAV format files, equating to 45 minutes, to align with the recorded BRUVs. Auditory and visual examinations were then used to remove any sporadic dominant anthropogenic interference using ‘seewave’ package in R (Sueur et al., 2008a). For each deployment Acoustic Complexity Index (ACI)(Pieretti *et al.*, 2011) was calculated, using the R packages ‘tuneR’ and ‘seewave’ (Ligges et al., 2016; Sueur et al., 2008a).

2.2.3 Acoustic Complexity Index

Originally developed to analyse terrestrial avian communities, the Acoustic Complexity Index (ACI) quantifies the change in adjacent spectrogram intensities for all temporal steps and frequency bins of a recording (Pieretti et al., 2011). Firstly, sound files were split into frequency bins and temporal steps. The change in adjacent intensities are then summed across these frequency bins and temporal steps. Thus, high ACI values are produced by large variations in sound intensity across many frequencies and times, whereas constant levels of similar intensity, such as most anthropogenic sources (e.g. boat engine), will produce low values of ACI (Bertucci et al., 2016).

ACI was chosen for the current study since the hydrophones used were encased in resin and could not be calibrated, which is a necessary requirement for calculating amplitude. As such, the acoustic files created could not be analysed with other popular acoustic measures which rely on amplitude, such as

Acoustic Entropy, Acoustic Richness, Root Mean Square or Sound Pressure Level (Picciulin et al., 2013; Sueur et al., 2008b).

2.3 Baited Remote Underwater Video Systems

Baited Remote Underwater Video systems (BRUVs) are a non-destructive method for sampling mobile communities (Babcock et al., 1999; Heagney et al., 2007). Three replicate BRUVs were deployed at each site (Fig. 2) for 45 minutes then recovered. Specifications of equipment are described in Bicknell *et al.,* (2019).

2.3.1 Video analysis

After a preliminary settling period of 5 minutes, 30 minutes of video were analysed in 1 minute segments. For each segment all mobile benthic organisms were identified and recorded. All organisms were identified to the highest taxonomic resolution possible. Abundance (MaxN) was calculated for each species from the maximum number of individuals of each species observed across all of the 30 minute segments.

2.4 Statistical Analysis

Permutational Analysis of Variance (PERMANOVA) was used to test differences in between years and treatments for the ACI, Shannon's Diversity Index, Number of Species and the assemblage composition. Year and Treatment were fixed factors with five and two levels respectively (Year: 2014, 2015, 2016, 2017 and 2018; Treatment: MPA and Open Control) using Primer v7 and PERMANOVA+ (Anderson et al., 2008; Clarke and Gorley, 2015). The assemblage composition analysis also included a random factor Area, which was nested inside Treatment (MPA=6 areas, OC=3 areas). PERMANOVA was chosen as it is robust to unbalanced designs (Sheehan et al., 2013b). For Shannon's Index, Number of Species and assemblage composition, the ACI was included as a covariate. The statistical significance of the variance components were tested using 9999 permutations under a reduced model (Anderson, 2001; Anderson and ter Braak, 2002). The analyses of the ACI and the two visual biodiversity indices were undertaken on the basis of a Euclidean distance matrix calculated from the Index values (Anderson and Millar, 2004). The assemblage composition analysis was based on a Bray-Curtis dissimilarity matrix calculated from dispersion weighted, fourth root transformed abundance data. Significant interactions (p<0.05) of fixed terms were tested using PERMANOVA pairwise tests.

To assess correlations between visual biodiversity measures and the ACI, scatter plots were created with Pearson correlations showing R values and significance (p<0.05). Assemblage composition was visualised using non-metric Multi-Dimensional Scaling (nMDS: Clarke and Gorley, 2015).

3. Results

3.1 Acoustic Complexity Index

The interaction between year and treatment was significant for the ACI (Table 1: Pseudo-F=2.6766, p=0.0351). This significant interaction shows that there is a combined effect of year and treatment. The MPA was more acoustically complex than Open Controls (OC) in 2014 and 2018 (Table 1; 2014: p=0.009; 2018: p=0.0288), whereas the OC group was more complex in 2016 (Fig. 3A, Table 1; 2016: p=0.0218). Overall across all years, mean ACI was lower inside the MPA (1.4% lower than outside: Fig. 3A).

*Table 1 Results table of PERMANOVA analysis of Euclidean distances assessing Acoustic Complexity Index with Year and Treatment as interactions and Pairwise comparisons of yearly differences between Treatments for the ACI. (Bold p values denotes significance, *: p<0.05, **: p<0.01, ***: p<0.001).*

	PERMANOVA				**Pairwise Comparisons**		
						MPA vs OC	
Source	**df**	**MS**	**Pseudo-F**	**p**	**Year**	**t**	**p**
Year	4	849.5	8.6736	**0.0001*** **	**2014**	2.7956	**0.009** **
Treatment	1	255.76	2.6113	0.1117	**2015**	0.33271	0.9741
Year x Treatment	4	262.15	2.6766	**0.0351***	**2016**	2.3627	**0.0218***
					2017	1.6034	0.1182

Residual	168	97.941		**2018**	2.2848	**0.0288***

3.2 Visual Biodiversity Indices

Both indices of diversity (Shannon's Diversity Index and Number of Species) were greater on average inside vs outside the MPA, with a mean percentage difference from outside to inside of: 20.0% for Shannon's and 8.4% for Number of Species (Fig. 3B & 3C).

When all diversity indices, both acoustic and visual biodiversity, are analysed by year within treatment, there is no significant trend with year displayed by the ACI, either inside or outside the MPA (Fig. 4, Inside: R=0.14, p>0.05; Outside R=0.18, p>0.05). However, Number of Species significantly increased with time both inside and outside the MPA (Fig. 4: Inside: R=0.36, p<0.0001, Outside: R=0.32, p=0.02). Outside the MPA, Shannon's index shows no significant trend with time and has a small significant increase with time inside the MPA (Fig. 4: Inside: R=0.21, p=0.017; Outside: $R=5.7x10^{-4}$, p>0.05).

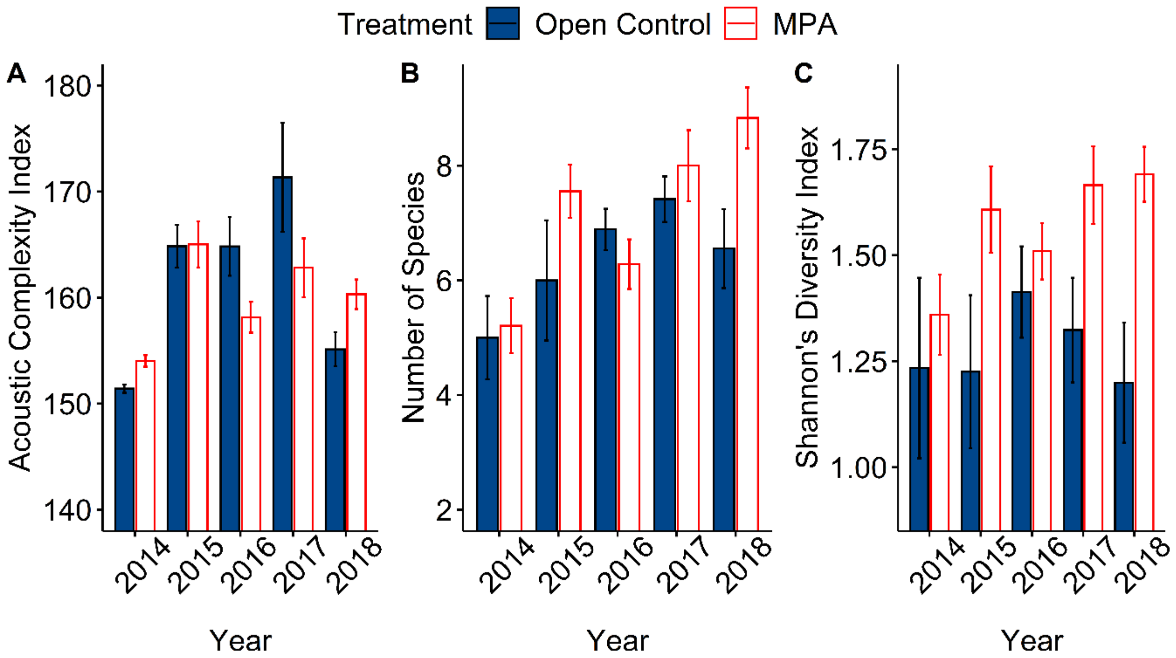

Fig. 3. Mean ± SE Acoustic Complexity (A), Number of Species (B) and Shannon's Diversity (C) Inside (Filled Blue) and Outside the MPA (Unfilled Red) across all years.

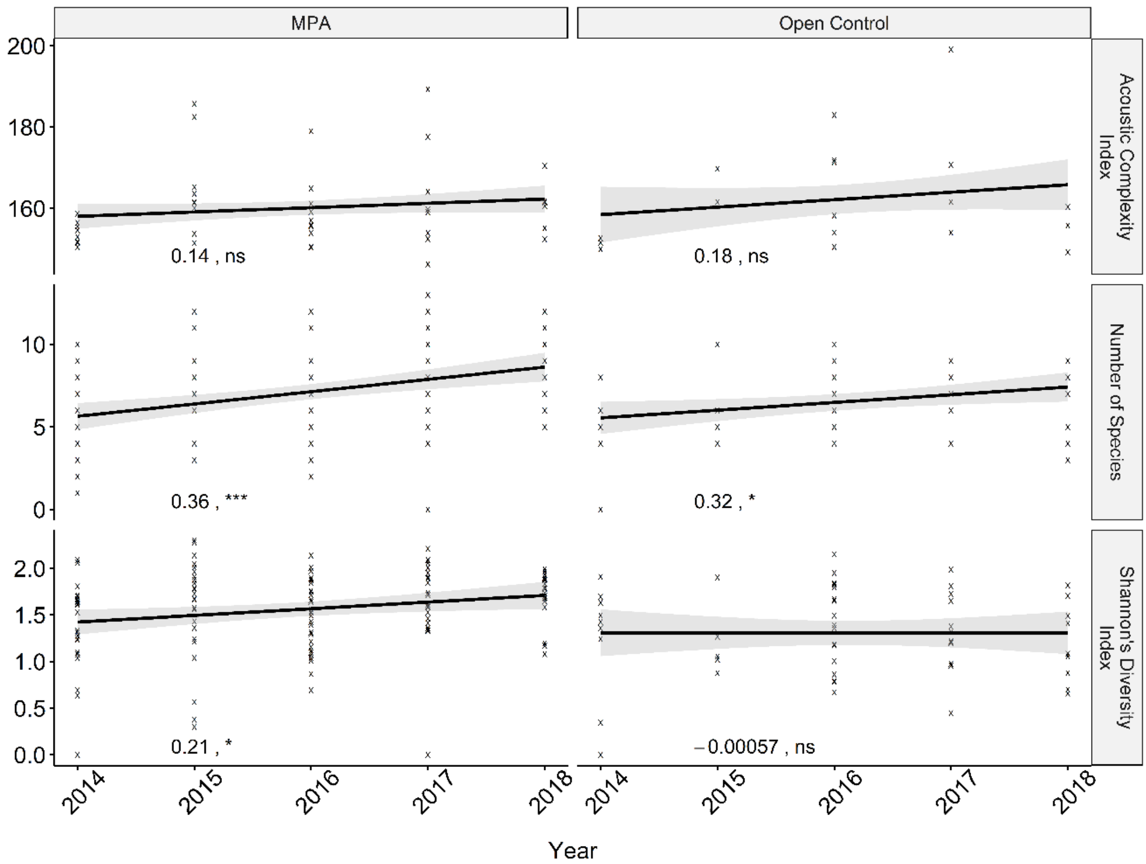

Fig. 4. Scatter plot with Pearson correlation coefficient for Year against Acoustic Complexity Index (top), Number of Species (middle) and Shannon's Diversity Index (bottom) split by treatment (Inside: blue and left, Outside: red and right). Includes R values and significance shown by ns: p>0.05,: p<0.05, **: p<0.01, ***: p<0.001. Shading around regression line shows 95% confidence interval.*

3.3 Relationship between Visual Biodiversity and Acoustic Complexity

Shannon's Index was greater in the MPA than the Open Controls but the relationship with Year was marginally non-significant (Table 2; Treatment: Pseudo-F=10.726, p=0.0013; Year: Pseudo-F=2.3123, p=0.0564). For the Number of Species there was a significant interaction between the Year and the ACI (Pseudo-F=6.4837, p=0.0002) as well as ACI and Treatment (Pseudo-F=6.1875, p=0.0157). This shows that, although not correlating with Number of Species, the ACI is interacting when the factors Year and Treatment are introduced. However, this is not significant for the Year x Treatment interaction.

*Table 2. Results table of PERMANOVA analysis on Euclidean distance assessing Shannon's Diversity Index (A) and Number of Species (B) with Year and Treatment as interactions and ACI as a covariate. (Bold p values denotes significance, *: p<0.05, **: p<0.01, ***: p<0.001).*

		Shannon's Diversity Index			Number of Species		
Source	**df**	**MS**	**Pseudo-F**	**p**	**MS**	**Pseudo-F**	**p**
ACI	1	0.17249	0.89629	0.3526	1.5537	0.34091	0.5553
Year	4	0.445	2.3123	0.0564	53.05	11.64	**0.0001*** **
Treatment	1	2.0641	10.726	**0.0013 ** **	6.3986	1.4039	0.2415
ACI x Year	4	0.30352	1.5771	0.1812	29.55	6.4837	**0.0002** **
ACI x Treatment	1	0.21135	1.0982	0.2936	28.2	6.1875	**0.0157* **
Year x Treatment	4	0.25056	1.3019	0.2712	10.249	2.2487	0.0635
ACI x Year x Treatment	4	0.27228	1.4148	0.2248	5.133	1.1262	0.3408
Residuals	158	0.19245			4.5576		

Neither Number of Species nor Shannon's Index correlated with the ACI when compared across all the years and treatments (Fig. 5; Shannon's Index: R=-0.37, p>0.05 and Number of Species: R=-0.067, p>0.05). However, within each year, Number of Species did correlate with ACI with the exception of 2016. However, the orientation of this correlation was inconsistent; it was positive for 2014 and 2018, and negative for 2015 and 2017 (Fig. 6; Positive- 2014: R=0.36, p=0.041; 2018: R=0.4, p=0.041; Negative- 2015: R=-0.42, p=0.017; 2017: R=-0.57, p<0.001). In contrast, Shannon's Index correlated with ACI in only 2017; this correlation was negative (Fig. 6; 2017: R=-0.4, p=0.017).

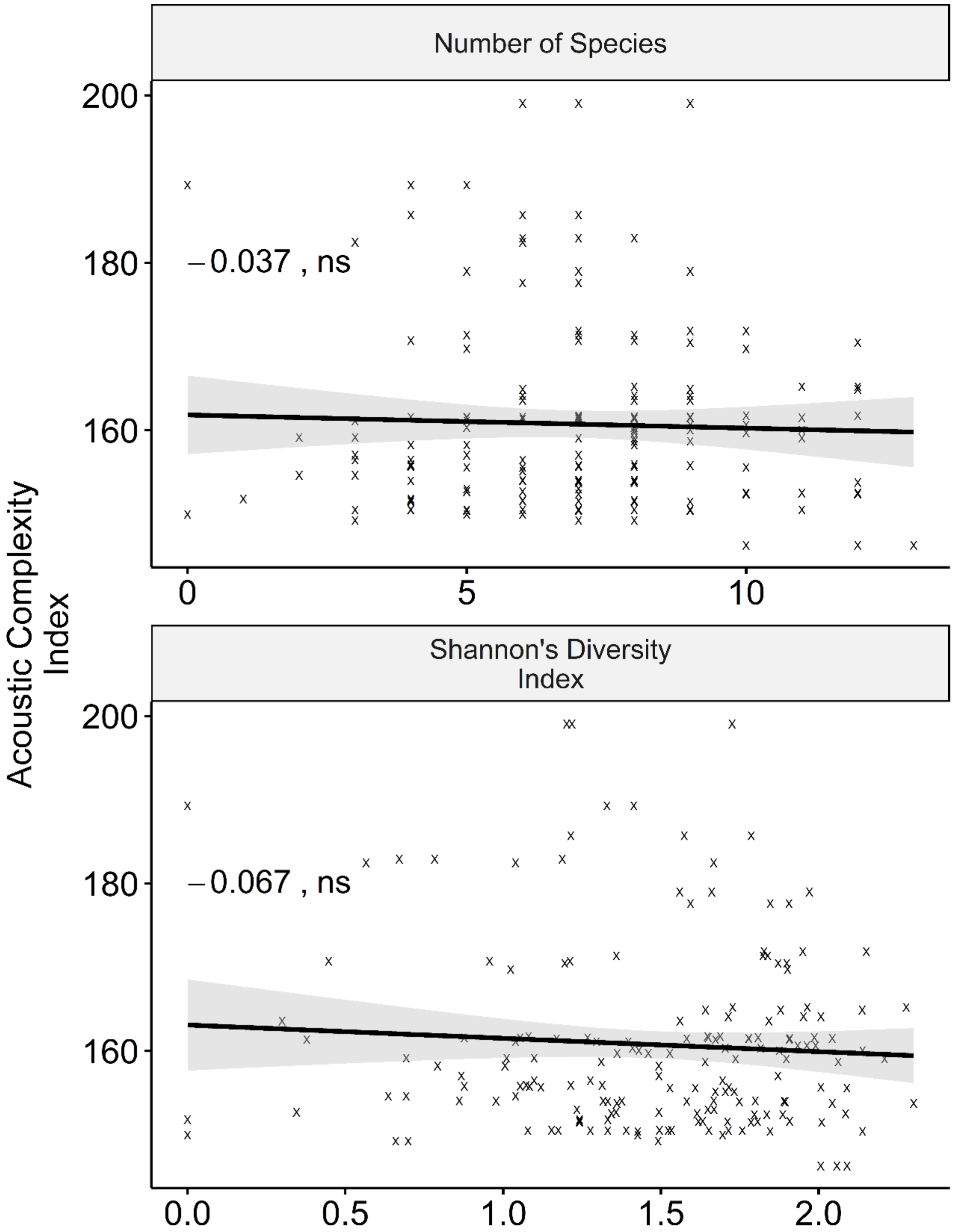

Fig. 5 Scatter plot with Pearson correlation coefficient for Acoustic Complexity against Number of Species (above) and Shannon's Diversity Index (below). R values are shown and significance shown by ns: p>0.05,: p<0.05, **: p<0.01, ***: p<0.001. Shading around regression line shows 95% confidence interval.*

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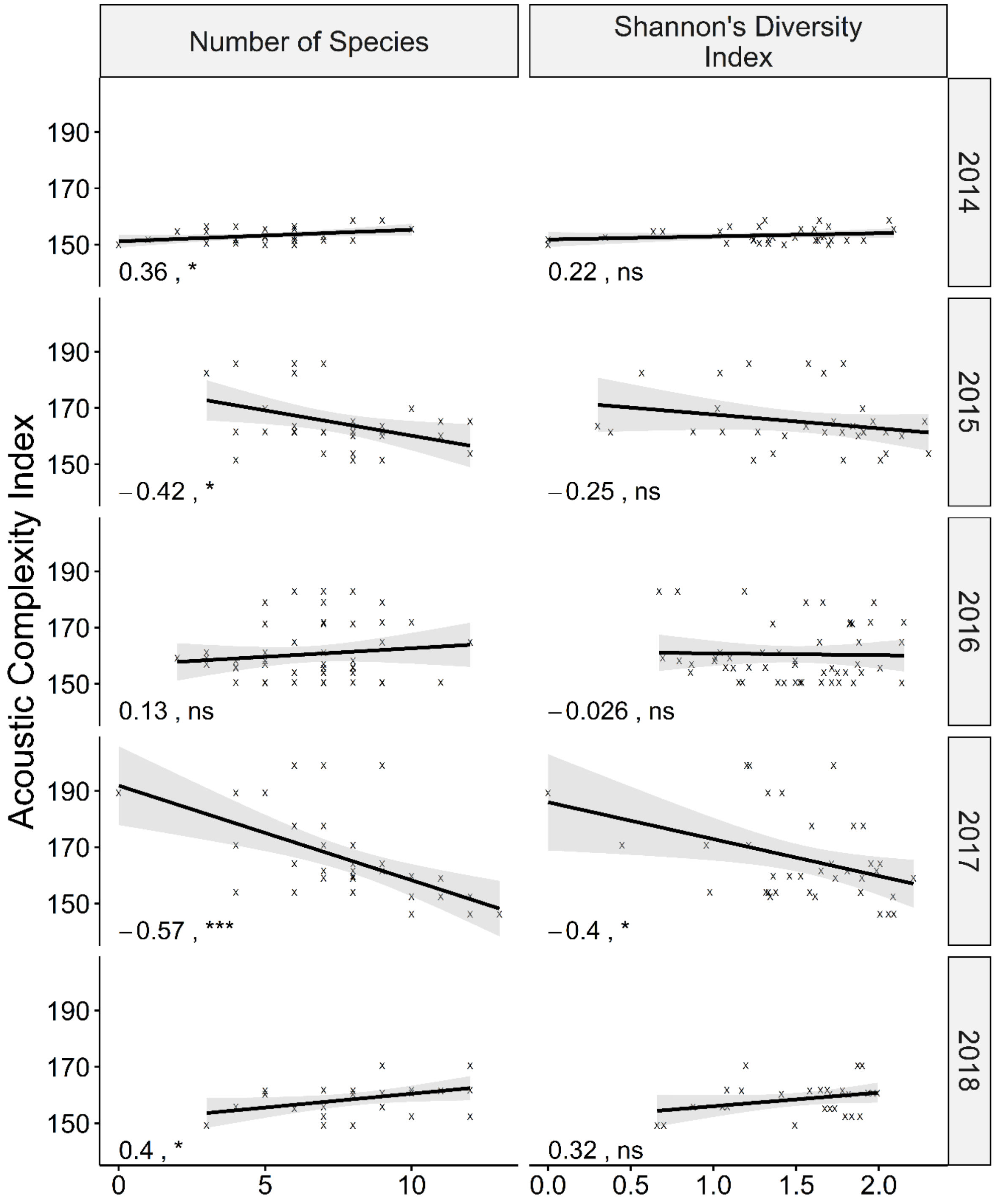

Fig. 6. Scatter plot with Pearson correlation coefficient for Number of Species (left) and Shannon's Diversity Index (right) against Acoustic Complexity Index. R values are shown and significance shown by ns: p>0.05,: p<0.05, **: p<0.01, ***: p<0.001. Shading around regression line shows 95% confidence interval.*

3.4 Mobile Benthic Assemblage Composition

The assemblage compositions of the two treatments diverged with increasing time with the two treatments changing at different rates (Fig. 7; Table 3: ACI x Year x Treatment: Pseudo-F: 1.7682, p=0.0482). Pearson correlation of more than 0.85 showed the reptant decapod crustaceans *Inachus* spp. and *Pagurus* spp. were most important to the Open Control composition (Fig.7), whereas, the species most important for the MPA assemblage were the wrasse species *Labrus mixtus, Labrus bergylta and Ctenolabrus rupestris* (Fig. 7).

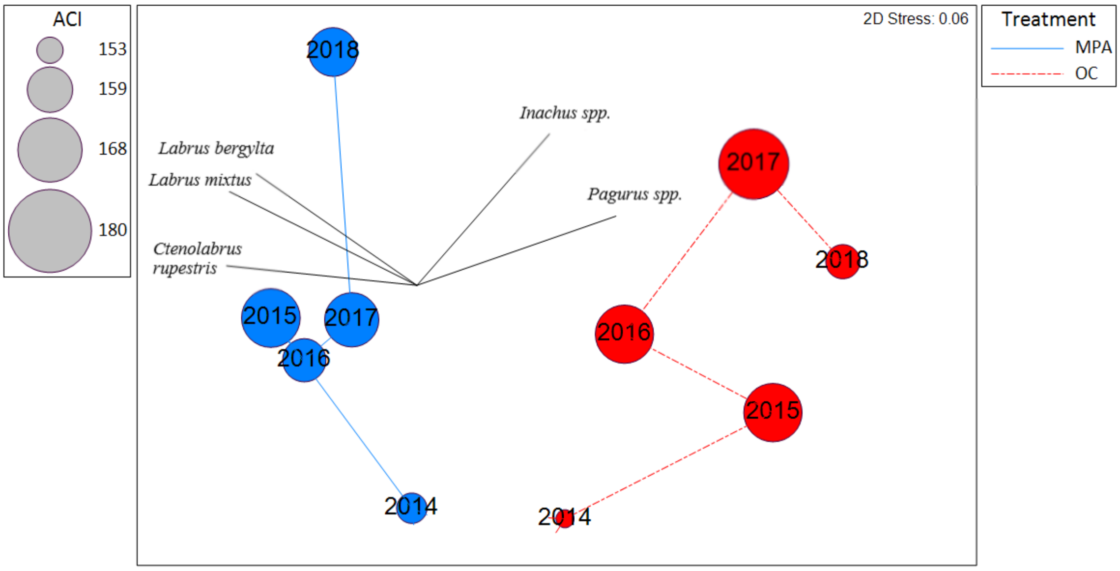

Fig. 7. Non-Metric Multidimensional Scaling plot of distance to centroids split by Year and Treatment from adjusted Bray-Curtis similarity of fourth root transformed abundance data. Points are labelled by Year and coloured by treatment (blue: Inside MPA, red: Outside MPA) and scaled according to mean ACI values. Vectors overlaid display 0.85 Pearson correlation for the species driving the difference in the assemblage composition.

*Table 3. Results table of PERMANOVA analysis on adjusted Bray-Curtis similarity assessing mobile benthic assemblage composition with Year and Treatment as fixed interactions, area as a random interaction nested within treatment and ACI as a covariate. (Bold p values denotes significance, *: p<0.05, **: p<0.01, ***: p<0.001).*

	Assemblage Composition			
Source	**df**	**MS**	**Pseudo-F**	**p**
ACI	1	4003.3	2.0845	**0.0457***
Year	4	9011.3	3.4253	**0.0001******

Treatment	1	29580	3.652	**0.0003*** **
Area(Treatment)	7	8302.8	8.974	**0.0001*** **
ACI x Year	4	2167.7	1.5326	0.0607
ACI x Treatment	1	2296.2	2.4819	**0.0177***
Year x Treatment	4	1883.9	1.1026	0.3666
ACI x Area(Treatment)	7	1338.7	1.4469	**0.0328***
Year x Area(Treatment)	23	1419.6	1.5344	**0.0001*** **
ACI x Year x Treatment	2	1635.9	1.7682	**0.0482***
Residuals	123	925.2		

4. Discussion

After high storm activity impacted the coastal systems of Lyme Bay and beyond (Masselink et al., 2016), acoustic and BRUV monitoring was implemented. It was hypothesised that the Acoustic Complexity Index would increase over time as the biodiversity of the area increased. Furthermore, the ACI was expected to be greater inside the protected area in comparison to the surrounding fished areas. Finally it was hypothesised that the ACI would change in a similar pattern to that of the mobile benthic assemblage composition recorded by BRUV systems.

4.1 Visual Biodiversity and Acoustic Indices

The Acoustic Complexity Index, as a covariate, showed a greater number of significant interactions alongside treatment and year with Number of Species than Shannon's Diversity (Fig. 4 & 6; Table 2). This relationship between ACI and Number of Species implies that the ACI is less affected by abundance and more by the number of species present. This is to be expected as the ACI was developed under the theory that many differing biological noises in an environment imply many different species (Pieretti et al., 2011). Yet, when studying fish vocal communities specifically, the ACI shows little discrimination between the abundance and the diversity of sound (Bolgan et al., 2018). Thus, the features that the ACI is enumerating from the marine soundscape may not, as hypothesised, correspond directly to the species detected by the BRUVs represented in the biodiversity indices tested. This may explain the yearly correlations between ACI and Number of Species, which changed orientation (between positive and negative), while also displaying small effect sizes throughout the study. This complete reversal, at times, could be misleading if studies using this method do not cover an appropriate temporal scale. This inconsistency may be the result of abiotic or anthropogenic noises (McWilliam and Hawkins, 2013), or specific species and behaviours dominating the soundscape, meaning the presence of specific species in the acoustically sampled area, but not recorded by the BRUVs, could be driving this inconsistent pattern and therefore may preclude it from certain applications, such as directly replacing more traditional biodiversity monitoring methods.

4.2 Shifting Benthic Composition

There was a clear divergence of assemblage compositions, between inside and outside treatment areas moving further apart year on year (Fig. 7). However, without data on the 'before fishing' assemblage, it would be very difficult to suggest whether this separation is recovery of the ecosystem. Yet, Pearson's correlations would suggest that the species most associated with the MPA are classed as reef dwelling species: *Ctenolabrus rupestris* remain in the same local area for several years, thus, maintaining their 'territory'(Darwall et al., 1992). The Open Controls were dominated by the scavenging species *Inachus* spp. and *Pagurus* spp. (Fig. 7): both have broad habitat preferences although *Inachus* spp. is more likely to be found on mixed coarse substrata (Rowley, 2008).

Although not significant, inside the MPA there was a higher Number of Species and Shannon's Diversity (Fig. 3). Both indices increasing with treatment (Shannon's Index) and year (Number of Species) show that the MPA in Lyme Bay acts as a refuge to allow biodiversity to increase (Fig. 4: Table 2). The assemblage composition does not interact with changing ACI alone, but is significant when aligned with year and treatment, which would be expected of a recovering system (Table 3). This would suggest that, although it did not correlate overall with visual biodiversity, the ACI is sensitive to some level of the non-background variation in the assemblage composition.

4.3 Applications of the Acoustic Complexity Index

Research into acoustic recordings is such a growing area due to its ability to provide information on local assemblage structure (Pijanowski et al., 2011; Sueur and Farina, 2015). Yet, as with most areas of ecology, the transition from the terrestrial to the marine poses a new range of obstacles to overcome (Giorli, 2016; Radford et al., 2011; Ricci et al., 2017). Many different indices have been produced to quantify marine biological processes, such as Acoustic Richness, Acoustic Entropy Index and Acoustic Complexity Index (Gage and Axel, 2014; Lillis et al., 2014; McWilliam and Hawkins, 2013; Staaterman et al., 2014). Their use as proxies for marine biodiversity has been assessed (Harris et al., 2016), with the Acoustic Complexity Index being the most favoured (Lindseth and Lobel, 2018) both alone and in combination with other acoustic indices (Gordon et al., 2018).

The Acoustic Complexity Index has been shown to have a number of drawbacks (Kaplan et al., 2015; McWilliam and Hawkins, 2013). These drawbacks can arise from interference by the biophony, geophony or anthrophony. For example, the ACI has shown to be increased heavily by snapping shrimp, which produce a high intensity broadband ‘snap’, meaning an increased ACI when diversity has only marginally increased (McWilliam and Hawkins, 2013). In contrast, chorusing behaviour can heavily decrease ACI (Kaplan et al., 2015). Hence, ACI in certain situations can be dominated by either few or many species, producing opposing changes in the ACI and the observed biodiversity. The assemblage composition outside of the MPA, in this case, was heavily dominated by hermit crabs of the genus *Pagurus*. It is possible that these large aggregations of *Pagurus* spp. (up to 70 in one video), which ‘rap’ on others’ shells for shell competition (Edmonds and Briffa, 2016), dominated the ACI in a similar way to snapping shrimps. Dominance of snapping shrimp in the marine soundscape affects most other acoustic indices, not just the ACI (Au and Banks, 1998; Lindseth and Lobel, 2018; Radford et al., 2008).Thus, this issue needs to be overcome for multiple different methods. The ACI can also be heavily influenced by geophony such as wind and rain (McWilliam and Hawkins, 2013). Although designed to minimise the influence of anthropogenic inputs, ACI will also be affected by any sounds which are not repetitive or consistent in intensity sounds, such as boat engines (Pieretti et al., 2011). Therefore, post sampling examination of the recordings was carried out here to minimise any sporadic dominant abiotic or anthropogenic interference, which would otherwise influence the ACI.

All recordings here were made during the day and, as such, potentially not at the highest acoustic activity times, which for most fish are dawn and dusk (Bertucci et al., 2017, 2016, 2015; Radford et al., 2014). Further investigation into this index should include diurnal recording strategies, while also taking into consideration the activity cycles based upon lunar phase (Harris et al., 2016; Staaterman et al., 2014). As shown here, correlations between the ACI and other diversity measures can occur, but can vary considerably in their orientations over years. Thus, temporal scales which include lunar and daily cycles, should be used to assess these indices. Although not possible here, the combination of multiple metrics together has been suggested to provide a more robust assessment of the marine soundscape (Gordon et al., 2018). However, the individual aspects of the soundscape which ACI quantifies need to be further understood before it can be appropriately combined with other metrics.

The use of ACI in this MPA, off the south coast of the UK, has not shown the direct relationship with the observed ecology as demonstrated elsewhere (Harris et al., 2016; Picciulin et al., 2013), yet did show significant interactions across treatments and years. As the significant interactions were found under multivariate and not univariate analysis, it is likely that the ACI is quantifying other elements of the marine soundscape and not just the diversity of species (Bolgan et al., 2018; Kaplan et al., 2015; McWilliam and Hawkins, 2013). For this, or another, acoustic index to be used as a rapid and cost-effective monitoring tool, the drawbacks mentioned here need to be addressed. Yet more importantly, the elements of the marine soundscape, which the ACI is quantifying, need to be better understood. Subsequently, thorough experimental assessments will be needed, with robust spatial and temporal coverage. This is essential, as based on a single year of this study (e.g. 2014 or 2018), ACI would have shown a positive correlation with Number of Species that has been found elsewhere. Again, this shows that temporal and geographical scales are important considerations for the development of any such index or method.

4.4 Conclusions

In conclusion, the Acoustic Complexity Index is not as yet ready to be used as a standalone marine diversity monitoring tool. In conjunction with other methods, such as BRUVs, which showed the recovery and increased diversity within the Lyme Bay MPA, this acoustic index shows potential to allude to shifting benthic assemblage compositions. Yet this was not seen with consistency when used alongside univariate measures of diversity. This implies that although it is not without its own disadvantages, the ACI is demonstrating more complex changes than overall univariate diversity. This potential as a tool for rapidly assessing a large area of the marine environment makes it highly attractive. However, for it to be used as a monitoring tool, the information it provides regarding shifting assemblage compositions and diversity needs to be fully researched and understood.

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Authors' Contributions

EVS and MJA conceived the ideas and monitoring design; MJW provided technical advice regarding acoustic analytical methods; EVS, LH, AR and BFRD collected data; BFRD and LH organized and analysed data; BFRD, EVS and LH led the writing of the manuscript. All authors contributed critically to drafts and gave final approval for publication.

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