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Impacts of data quality on the setting of conservation planning targets using the species–area relationship

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ABSTRACT

Aim The species–area relationship (SAR) is increasingly being used to set conservation targets for habitat types when designing protected area networks. This approach is transparent and scientifically defensible, but there has been little research on how it is affected by data quality and quantity.

Location English Channel.

Methods We used a macrobenthic dataset containing 1314 sampling points and assigned each point to its associated habitat type. We then used the SAR-based approach and tested whether this was influenced by changes in (i) the number of sampling points used to generate estimates of total species richness for each habitat type; (ii) the nonparametric estimator used to calculate species richness; and (iii) the level of habitat classification employed. We then compared our results with targets from a similar national-level study that is currently being used to identify Marine Conservation Zones in the UK.

Results We found that targets were affected by all of the tested factors. Sample size had the greatest impact, with specific habitat targets increasing by up to 45% when sample size increased from 50 to 300. We also found that results based on the Bootstrap estimator of species richness, which is the most widely used for setting targets, were more influenced by sample size than the other tested estimators. Finally, we found that targets were higher when using broader habitat classification levels or a larger study region. However, this could also be a sample size effect because these larger habitat areas generally contained more sampling points.

Main conclusions Habitat targets based on the SAR can be strongly influenced by sample size, choice of richness estimator and the level of habitat classification. Whilst setting habitat targets using best-available data should play a key role in conservation planning, further research is needed to develop methods that better account for sampling effort.

Keywords

English Channel, habitat targets, Marine Conservation Zones, marine protected areas, species–area relationship, systematic conservation planning.

INTRODUCTION

Marine and coastal ecosystems are under increasing pressure from a diverse range of threats including the over-exploitation of natural resources (particularly overfishing), pollution and climate change (Lubchenco *et al.*, 2003). One response to these threats is to develop marine protected areas (MPAs), which are seen as increasingly important spatial management tools for conserving marine biodiversity (Wood *et al.*, 2008), maintaining large-scale ecological processes (Roberts *et al.*, 2005) and supporting the sustainable use of marine resources (Spalding *et al.*, 2008). A widely used approach for helping to

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ensure that new MPAs achieve these goals is systematic conservation planning, which seeks to identify representative and viable networks of MPAs that also minimize costs (Margules & Pressey, 2000). Thus, systematic conservation planning can be used to design MPA networks that balance impacts on different stakeholders (Smith *et al.*, 2009), increase the likelihood of implementation and help ensure long-term biodiversity persistence (Knight *et al.*, 2006).

A key step in systematic conservation planning involves producing a list of important species, habitats and ecological processes, known collectively as 'conservation features', and then setting quantitative targets for the minimum amount of each feature intended for conservation (Knight *et al.*, 2006; Carwardine *et al.*, 2009). These targets can then be used by several conservation planning software packages (e.g. Marxan, C-Plan and Zonation) to help identify priority areas for protection (Ball *et al.*, 2009). Setting such targets provides a clear basis for conservation decisions, lending them accountability and defensibility, and ensures that the conservation planning process is more transparent, open to stakeholder involvement and less likely to be affected by political interference (Cowling *et al.*, 2003b). Approaches to target setting depend on the type of conservation feature of interest (Noss, 1987). Targets for species are often set using relatively well-established techniques based on population viability estimates (Rondinini *et al.*, 2006; Justus *et al.*, 2008; Rondinini & Chiozza, 2010). In contrast, target-setting approaches for coarse-filter conservation features, such as habitat and vegetation types, are frequently based on expert opinion (e.g. Cowling *et al.*, 2003a; Pressey *et al.*, 2003; Smith *et al.*, 2006) or policy-driven targets such as those specified in the Convention on Biological Diversity (CBD), which currently recommends that 10% of coastal and marine areas under national jurisdiction should be protected by 2020 (CBD, 2011). However, both expert-based and policy-driven targets have been widely criticized for a lack of ecological credibility (see review by Carwardine *et al.*, 2009), so there is a real need for data-driven and scientifically defensible approaches for setting habitat targets.

In response to this problem, researchers developed an approach based on using field survey data to model the species–area relationship (SAR) for each important habitat type, which is then used to estimate the proportion of habitat area required to represent a user-specified percentage of species, and can be multiplied by the extent of the habitat type to produce a target area (Desmet & Cowling, 2004; Reyers *et al.*, 2007). This methodology was subsequently adopted by the South Africa National Biodiversity Institute (SANBI) to calculate targets for each vegetation type listed in the national vegetation classification system (Rouget *et al.*, 2004). These targets were then used to help identify priority conservation areas (Rouget *et al.*, 2006; Smith *et al.*, 2008; Gallo *et al.*, 2009) and conduct threatened vegetation type assessments as part of South Africa's first National Spatial Biodiversity Assessment (Nel *et al.*, 2007; Reyers *et al.*, 2007), helping to ensure a level of consistency between projects and regions.

The success of this approach means that SAR-based targets are beginning to be developed elsewhere. In particular, they have been used to set national marine habitat targets as part of four regional projects funded by the UK Government, which seek to establish a network of Marine Conservation Zones (MCZs) in English territorial waters (JNCC & Natural England, 2010; Rondinini, 2011a). With increasing adoption, it is important that conservation planners and practitioners have confidence in this approach to target setting, as targets have a large influence on the final extent of any protected area (PA) network (Vimal *et al.*, 2011; Delavenne *et al.*, 2012) and any subsequent socio-economic impacts (Chittaro *et al.*, 2010; Mascia *et al.*, 2010; McCrea-Strub *et al.*, 2011). However, despite their growing use, there is still uncertainty about how this target-setting process is affected by data constraints, as the SAR is known to be influenced by biogeographical patterns, model parameters, model type and data quality (Chiarucci *et al.*, 2003; Walther & Moore, 2005; Hortal *et al.*, 2006). Here, we investigate these issues using a macrobenthic dataset from the eastern English Channel, examining how targets are affected by the number of sampling points used to model the SAR, the choice of estimator used to calculate total species richness in each habitat type and the level of habitat classification employed. We then compare these results developed at a regional level with those developed for the MCZ project at a national-level and assess how using these different sets of targets would influence the extent of any resulting MPA network in the English Channel.

METHODS

Study area

This study was carried out in the English Channel (Fig. 1), a cold-temperate epicontinental sea separating the south coast of the UK from the north coast of France (Delavenne *et al.*, 2012). The English Channel constitutes a biogeographical transition zone between the warm temperate Atlantic oceanic system, and the boreal North Sea and Baltic Sea continental systems of northern Europe, encompassing a wide range of ecological conditions (Coggan & Diesing, 2011; Delavenne *et al.*, 2012). The study region focused on the eastern English Channel (EEC), which is delimited by the Dover Strait to the east and Cotentin Peninsula to the west and is a key area for tourism, shipping, energy production and aggregate extraction (Carpentier *et al.*, 2009). In addition, it supports an important commercial fishery, as well as key nursery, spawning areas and migratory routes linked to specific environmental characteristics (Martin *et al.*, 2009).

There are several ongoing MPA designation projects in this section of the English Channel. Both France and the UK have implemented MPAs as part of their EU Birds and Habitats Directive commitments, and France is currently developing an MPA network in the 'Three Estuaries region' (Bay of Somme, Authie and Canche; Fig. 1). In addition, the EEC is

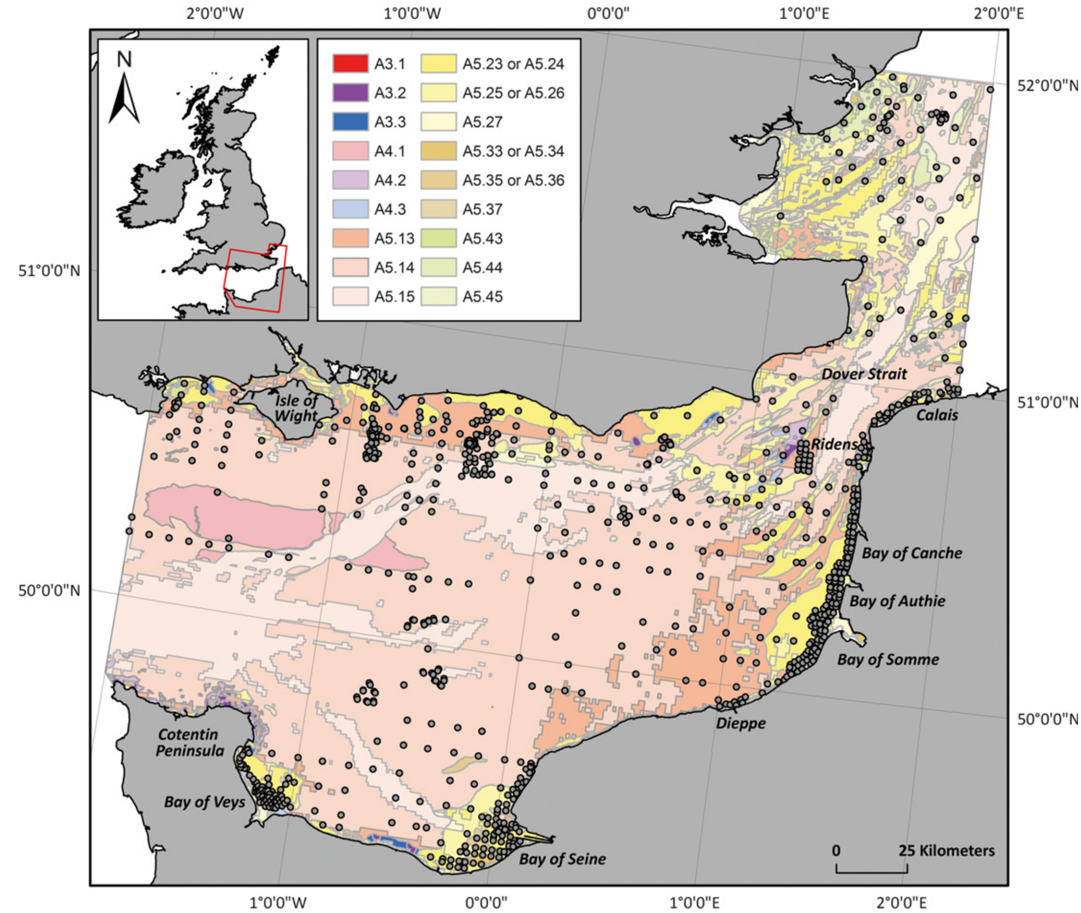

Figure 1 EUNIS levels 3 and 4 habitat map for the eastern English Channel showing the location of the 1314 sampling points. See Table S1 for a key to EUNIS habitat codes, levels and descriptions.

the focus of the Balanced Seas project (http://www.balancedseas.org/), which is one of the four regional MCZ projects, which seeks to identify and recommend MPAs for the inshore and offshore waters of south-east England (JNCC & Natural England, 2010). Balanced Seas uses habitat targets based on the SAR that were developed at a national level from biodiversity data collected in English waters (JNCC & Natural England, 2010).

Habitat map

We used a broad-scale habitat map in this analysis, which is based on the European Nature Information System (EUNIS) habitat classification hierarchy developed by the European Environment Agency (EEA, 2006; Coggan & Diesing, 2011). Figure 1 shows the distribution of each EUNIS habitat class that was modelled using physical and environmental data, including depth, substratum and energy levels. Rock habitats were modelled to level 3 in the EUNIS hierarchy, whilst sediment habitats were modelled to level 4 (Coggan & Diesing, 2011). The EUNIS level 3 habitats are broken down into three habitat types and coded as follows: infralittoral rock (A3.x), circalittoral rock (A4.x) and sublittoral coarse sediment (A5.x), which was further divided into its finer-scale EUNIS level 4 habitats (A5.xx).

Biodiversity survey data

Given the importance of macrobenthic diversity in the EEC (Vaz *et al.*, 2007; Carpentier *et al.*, 2009), the increasing emphasis on their conservation (Sanvicente-Anorve *et al.*, 2002; Vincent *et al.*, 2004) and the large amount of benthic sampling that has taken place (e.g. Desroy *et al.*, 2003; Dauvin *et al.*, 2004; Carpentier *et al.*, 2009), we developed targets using presence/absence data from macrobenthic surveys carried out between 1985 and 2007, providing data from 1314 sampling points (Fig. 1). These surveys used a range of sampling protocols and gear sizes (0.1–0.5 m^2), with samples predominantly collected using a Hamon grab, with the exception of 16 stations in the Ridens that used a van Veen grab. The sampling strategy in the study area was predominantly regularly spaced; however, there was more intensive sampling in surveys from the east of the Isle of Wight, in the Ridens and in coastal areas such as between Dieppe and Calais, the Bay of Veys and the Bay of Seine (Fig. 1).

Calculating habitat targets

We calculated habitat targets following the SAR-based approach developed by Desmet & Cowling (2004), which treats the SAR as a power function. Whilst concerns about using this particular approach in conservation planning have been expressed in the literature (see Smith, 2010 for a detailed review), we employed it in our study because (i) we specifically sought to investigate the uncertainties around this existing approach; (ii) the power function has been shown to perform well for macrobenthic datasets containing between 42 and 1300 samples (Azovsky, 2011).

This approach involves transforming the power function (equation 1) to estimate the proportion of habitat area required to represent a given percentages of species (equation 2):

$$S = cA^Z \quad (1)$$

$$\text{Log}\, A' = \text{Log}\, S'/z. \quad (2)$$

Here, S' and A' denote the proportion of species and habitat area respectively (Desmet & Cowling, 2004; Rondinini & Chiozza, 2010), and z describes the slope of the power function, which is the rate of species accumulation with increase in area (Lomolino, 2000; Tjorve & Tjorve, 2008). The constant c is a scaling factor that relates to the size (area) of an individual sampling unit and can be ignored when comparing proportions or percentages of species and area (Desmet & Cowling, 2004; Rondinini & Chiozza, 2010). Thus, it is possible to calculate habitat targets by (i) determining the z-value of the SAR for a given habitat; (ii) using the z-value to calculate the proportion of area required to represent a given percentage of species; and (iii) multiplying this proportion by the total habitat area.

We calculated habitat-specific z-values using the formula for calculating the slope of a straight line (equation 3), because a SAR modelled with a power function appears as a straight line with slope z on a log-log plot (Desmet & Cowling, 2004).

$$z = (y_2 - y_1)/(x_2 - x_1), \quad (3)$$

where y_2 = log(total number of species in a habitat class); y_1 = log(average number of species per sampling point); x_2 = log(total area of habitat class); and x_1 = log(average area of sampling points). Three of these variables (y_1, x_2, x_1) are derived from habitat-specific inventory data (Desmet & Cowling, 2004; Rondinini & Chiozza, 2010), so all that is needed to calculate z-values is to estimate the total number of species (y_2) in a given habitat type (Desmet & Cowling, 2004).

The habitat map shows the distribution of each EUNIS level 3 habitat type and subdivides the sedimentary habitat types further into finer-scale EUNIS level 4 types (Fig. 1). Thus, we assigned sampling points on rocky habitats to their associated level 3 habitat types and sampling points on sedimentary habitats to both their associated parent level 3 habitat types, and their constituent level 4 habitat types (see Fig. S1 and Table S1 in Supporting Information for more information regarding EUNIS level 3 parent habitats for level 4 habitat types in the EEC). We then calculated targets for each of these level 3 and level 4 habitats using *EstimateS* software (Colwell, 2009) to generate estimates of total species richness (y_2) and determine habitat-specific z-values for each of these habitat types.

Although there is no consensus as to which estimator provides the best predictions when estimating total species richness for a habitat type (or region) from field survey data (Brose, 2002; Herzog *et al.*, 2002; Chiarucci *et al.*, 2003; Walther & Moore, 2005), there is general agreement that the Bootstrap estimator is the most conservative (Colwell & Coddington, 1994; Chiarucci *et al.*, 2001, 2003; Hortal *et al.*, 2006). A prediction of total species richness based on this estimator should be considered as a minimum estimate (Desmet & Cowling, 2004; Rondinini, 2011a), which is why this estimator was subsequently applied by the SANBI and MCZ projects to develop national targets for both terrestrial and marine habitats.

To assess the effect that choice of species richness estimator has on the calculation of conservation targets, we compared targets derived using the Bootstrap estimator to those derived using several alternative nonparametric estimators of species richness – ICE, Chao2, Jackknife1 and Jackknife2. Whilst these alternative estimators were investigated by both Desmet & Cowling (2004) and Rondinini (2011a), these authors did not explicitly test their effect on target setting (see Colwell & Coddington, 1994; Gotelli & Colwell, 2001; Hortal *et al.*, 2006; Colwell, 2009 for more details on these estimators and their performance). Our comparison involved calculating each richness estimate based on the mean of 1000 estimates that used 1000 randomizations of sample accumulation order without replacement (Colwell, 2009). We then used these results to (i) calculate the proportion of habitat area required to represent 80% of species, hereafter referred to simply as 'targets', for each habitat type with > 5 sampling points – we chose to calculate targets based on representing 80% of species because this was used by the Balanced Seas and the other regional MCZ projects (JNCC & Natural England, 2010); (ii) estimate the number of sampling points required to produce a stable target for each habitat type, and each richness estimator, where a target was defined as stable if it exhibited a standard deviation of < 5% (as used by Desmet & Cowling, 2004); (iii) assess how the targets developed in this study compare with those from the MCZ project in the EEC; and (iv) assess how sensitive each of the estimators was to sample size effects using successively larger numbers of accumulated sampling points, which involved dividing the percentage target for each habitat type based on 100, 200 and 300 sampling points by the percentage target based on 50 sampling points (we then took the mean of each of these habitat results for each estimator to show how relative target size changed with sample size).

Finally, we investigated the effects of using different levels of habitat classification on the extent of the MPA network needed to meet the targets. This involved multiplying each habitat target by the extent of its occurrence in the planning region to provide an area target in km^2 and then summing these area targets from EUNIS level 4 habitats belonging to the same 'parent' level 3 type, so that the combined level 4 result could be compared with the level 3 result.

RESULTS

On the basis of using stable results for the Bootstrap estimator, the total number of species estimated to occur in each habitat class ranged between 240 and 1665 for the six EUNIS level 3 habitats, whilst estimates for the ten EUNIS level 4 habitats ranged between 160 and 1470 (Table 1). Habitat-specific *z*-values ranged between 0.098 for deep sea mixed sediments and 0.162 for sublittoral sand (Table 1). Percentage targets ranged from 10.27% for deep sea mixed sediments to 25.28% for sublittoral sand (Table 1), so that eight of the EUNIS level 4 habitats and four of the EUNIS level 3 habitats had targets of greater than 10% (Table 1). On the basis of the available data for each habitat investigated, this would translate into approximately 18.41% of the EEC for the finer-scale EUNIS mixed levels 3 and 4 habitat classification (Fig. 1), compared with 20.27% for the coarse-scale EUNIS level 3 habitat classification (Fig. S1).

We found that both estimates of species richness (Table S2), and resulting targets, varied between different estimators, with the difference in targets for a given habitat ranging between 1.58% for infralittoral coarse sediment, and 7.66% for low-energy circalittoral rock (Table 2). In addition, there were clear differences in the number of sampling points required to reach stable target estimates across estimators, with the Bootstrap estimator producing twelve stable target estimates, compared with five for the Jackknife1 estimator (Table 2). Moreover, the Bootstrap estimator generally required the smallest number of sampling points to reach stable estimates compared with the other estimators. For example, for a relatively well-sampled habitat such as sublittoral sand with a total of 469 sampling points, the Bootstrap estimator required 276 sampling points to reach stability compared to 409 for Chao2 (Table S3).

When we evaluated how targets calculated with the Bootstrap estimator varied with successively larger numbers of accumulated samples, we found that estimates of both species richness and targets increased with sampling effort (Table 3). For example, we found that for four relatively well-sampled habitats (sublittoral coarse sediment, infralittoral coarse sediment, circalittoral coarse sediment and sublittoral sand), targets increased by 39%, 30%, 39% and 45%, respectively, when the number of sampling points increased from 50 to 300 (Table 3), with the mean relative target increasing by 41% across all habitats (Fig. 2). In addition, the standard Bootstrap approach produced targets that were most influenced by sample size, as the mean relative increase in targets for the other estimators ranged from 26% for ICE to 33% for Jackknife1 when the number of sampling points increased from 50 to 300 (Fig. 2).

The level of habitat classification also impacted the targets, with species richness estimates, habitat-specific *z*-values and targets being higher when developed for parent EUNIS level 3 habitats than for their finer-scale EUNIS level 4 constituents (Table 1). For example, the area of each parent EUNIS level 3 habitat needed to meet targets was 8.4% higher for sublittoral coarse sediments and 41.4% higher for sublittoral mixed sediments when compared to the combined target area of their finer-scale EUNIS level 4 constituents (Fig. 3).

Finally, our regional EEC targets developed in this study were lower than the national MCZ targets developed for EUNIS level 3 habitats, with our targets ranging between 15.49% and 25.28% compared with 29.80–32.40% recommended by the MCZ Ecological Network Guidance, producing large differences in the area of habitat needed to meet these targets (Table 4).

DISCUSSION

The SAR is increasingly being used to set targets for habitat types in systematic conservation planning (Smith, 2010) and has been specifically advocated for use in marine conservation planning (Neigel, 2003; Smith *et al.*, 2009). Nonetheless, SAR-based targets have to be part of a broader set of PA design parameters because they relate only to the minimum representation of biodiversity, that is, ensuring the presence of a species regardless of its abundance, rather than ensuring its persistence (Smith, 2010). Moreover, the approach provides no information about where PAs should be located within a particular habitat type (Desmet & Cowling, 2004; Justus *et al.*, 2008; Chittaro *et al.*, 2010; Rondinini & Chiozza, 2010). However, SAR-based target setting is likely to remain an important element of terrestrial and marine PA network design. This paper is the first to investigate several key issues that may affect the robustness of targets set using this approach.

Effects of sample size, species richness estimator and habitat classification level

The value of the SAR-based approach depends entirely on producing accurate habitat-specific *z*-values which, in turn, requires accurate estimates of total species richness within each habitat type. However, species richness estimates may be sensitive to the type of estimator used (Table S2) and the amount and quality of biological survey data employed, rather than reflecting true differences in species accumulation rates (Colwell *et al.*, 2004; Walther & Moore, 2005; Hortal *et al.*, 2006; Rondinini & Chiozza, 2010). Our results show that the rate of species accumulation with increase in area (expressed as the *z*-value) for each habitat type was quite similar across estimators (Table S4) which is consistent with other studies that have investigated the behaviour of these

Table 1 Habitat-specific inventory data, total number of species estimated to occur in each habitat type (values calculated using Bootstrap estimator and rounded to nearest whole number), z-values and the proportion (%) of target habitat area for each EUNIS levels 3 and 4 habitat type.

EUNIS code	EUNIS level	EUNIS Habitat description	Area (km^2) of habitat	Number of sampling points	Average area (m^2) of samples	Average number of species per sample	Total number of observed species	Bootstrap estimator (y_2)	Number of samples to reach stable estimate	z-value	Target (%)
A3.3	3	Low-energy infralittoral rock	116	11	0.5	10	60	74	–	0.104	11.68
A4.3	3	Low-energy circalittoral rock	108	5	0.5	38	142	178	–	0.080	6.25
A5.1*	3	Sublittoral coarse sediment	29,889	725	0.26	53	1520	1665	65	0.135	19.23
A5.13	4	Infralittoral coarse sediment	4092	263	0.2	46	971	1079	67	0.133	18.65
A5.14	4	Circalittoral coarse sediment	18,934	373	0.31	59	1326	1470	53	0.129	17.84
A5.15	4	Deep circalittoral coarse sediment	6863	89	0.25	49	825	950	52	0.123	16.38
A5.2*	3	Sublittoral sand	7633	469	0.45	18	714	823	276	0.162	25.28
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	3701	288	0.45	18	590	684	208	0.159	24.65
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	3046	165	0.45	18	454	539	133	0.150	22.63
A5.27	4	Deep circalittoral sand	886	16	0.28	14	128	160	15	0.111	13.48
A5.3*	3	Sublittoral mud	335	28	0.48	21	198	240	27	0.120	15.49
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	196	17	0.49	18	139	170	–	0.113	13.97
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	134	11	0.46	26	131	158	–	0.093	8.98
A5.4*	3	Sublittoral mixed sediments	900	64	0.26	25	333	393	44	0.130	16.88
A5.44	4	Circalittoral mixed sediments	477	50	0.3	25	245	287	38	0.115	14.41
A5.45	4	Deep mixed sediments	198	14	0.11	25	164	202	13	0.098	10.27

*Species richness estimates and corresponding z-values for these EUNIS level 3 habitats are obtained from their combined EUNIS level 4 habitat and survey data; A5.1 = (A5.13, A5.14, A5.15); A5.2 = (A5.23 or A5.24, A5.25 or A5.26, A5.27); A5.3 = (A5.33 or A5.34, A5.35 or A5.36); and A5.4 = (A5.44, A5.45).

Table 2 Proportion (%) of target habitat area for each of the EUNIS levels 3 and 4 habitat types, based on five estimators of species richness. Shaded targets were determined not to be stable as the standard deviation of the richness estimate was > 5% of the estimate.

EUNIS code	EUNIS level	EUNIS habitat description	Number of sampling points	Non-parametric estimators: ICE	Chao2	Jackknife1	Jackknife2	Bootstrap	Mean target	Target range
A3.3	3	Low-energy infralittoral rock	11	17.53	14.96	14.28	16.31	11.68	14.95	5.85
A4.3	3	Low-energy circalittoral rock	5	13.91	12.07	8.89	11.17	6.25	10.46	7.66
A5.1	3	Sublittoral coarse sediment	725	19.94	20.45	20.18	21.05	19.23	20.17	1.82
A5.13	4	Infralittoral coarse sediment	263	19.34	19.16	19.66	20.23	18.65	19.41	1.58
A5.14	4	Circalittoral coarse sediment	373	18.71	18.97	18.90	19.79	17.84	18.84	1.95
A5.15	4	Deep circalittoral coarse sediment	89	17.83	17.54	17.78	18.79	16.38	17.66	2.41
A5.2	3	Sublittoral sand	469	27.04	26.97	26.65	27.83	25.28	26.75	2.55
A5.23 or A5.24	4	Infralittoral fine sand or muddy sand	288	26.57	26.09	26.10	27.22	24.65	26.13	2.57
A5.25 or A5.26	4	Circalittoral fine sand or muddy sand	165	26.22	26.45	24.54	26.39	22.63	25.25	3.82
A5.27	4	Deep circalittoral sand	16	18.56	17.20	15.90	17.99	13.48	16.63	5.08
A5.3	3	Sublittoral mud	28	20.70	20.24	17.96	20.27	15.49	18.93	5.21
A5.33 or A5.34	4	Infralittoral sandy mud or fine mud	17	19.15	19.15	16.66	19.15	13.97	17.62	5.18
A5.35 or A5.36	4	Circalittoral sandy mud or fine mud	11	13.61	14.84	11.56	13.98	8.98	12.59	5.86
A5.4	3	Sublittoral mixed sediments	64	19.87	19.87	18.86	20.63	16.88	19.22	3.75
A5.44	4	Circalittoral mixed sediments	50	17.33	18.29	16.48	18.48	14.41	17.00	4.07
A5.45	4	Deep mixed sediments	14	16.14	14.83	12.72	15.01	10.27	13.79	5.87

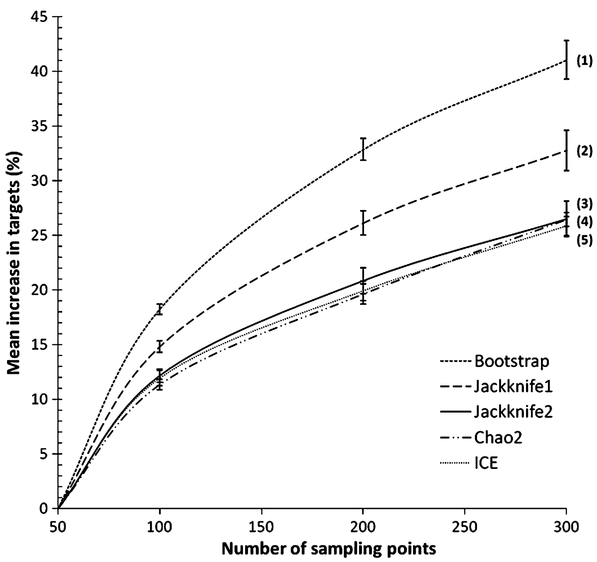

Figure 2 Mean increase in targets (including standard errors) based on increasing sample size across all habitats for the (1) Bootstrap; (2) Jackknife1; (3) Jackknife2; (4) Chao2; and (5) ICE estimators, relative to an estimate based on 50 sampling points.

estimators (Borges *et al.*, 2009). However, we show that sample size in particular can have a large influence on targets, so that increasing the number of sampling points often produced substantially higher targets (Fig. 2; Table 3). The number of sampling points needed to produce a stable result also varied with estimator type, with the Bootstrap estimator generally requiring the fewest number to reach stability (Table 2) which is consistent with the results obtained for the MCZ project (Rondinini, 2011a). This estimator is the most widely used for setting habitat targets (e.g. Desmet & Cowling, 2004; Rondinini, 2011a) and our stability results provide further support for this use (Table 2). However, we also found this estimator produced targets that were most influenced by changes in sample size (Fig. 2), which raises doubts about the robustness of the targets produced using the standard Bootstrap-based approach.

We also investigated the extent to which using different habitat classification levels affects targets because SAR-based targets provide no information about where PAs should be located within a given habitat type. Thus, it is generally better to use the most detailed habitat classification available because this ensures each finer-scale habitat type is represented. However, dividing broad-scale parent habitat types into finer-scale subclasses also results in a reduction in the number of sampling points used to calculate targets for these habitats, and so we would expect these smaller sample sizes to produce lower targets. Our results confirmed this pattern, with the area of each parent EUNIS level 3 habitat needed to meet targets calculated at this level always being higher than the combined area of the constituent EUNIS level 4 habitat targets (Fig. 3). In some cases, dividing up the data into level 4 types led to sample sizes that were too small to produce stable results (Table 2), but even results for sublittoral coarse sediment and sublittoral sand habitats, which were relatively

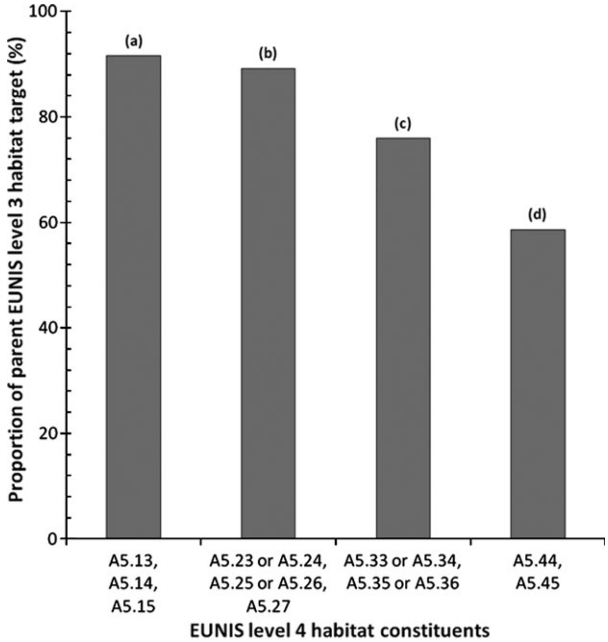

Figure 3 The proportion of target habitat area for combined fine-scale EUNIS level 4 habitat constituents compared with their coarse-scale EUNIS level 3 parent habitats: (a) A5.1; (b) A5.2; (c) A5.3; and (d) A5.4.

well sampled, showed that using the finer-scale level 4 instead of level 3 habitat classification reduced the total area needed to meet the targets (Fig. 3). However, it is possible that this result might also reflect a more direct effect of habitat classification level on the magnitude of targets. This is because habitats types that are subdivided into finer classes are more biologically homogenous, so the target area needed to represent a specified proportion of species may become lower (Whittaker *et al.*, 2001).

These results suggest that conservation planners need to be careful when calculating and interpreting SAR-based targets, yet there is currently little guidance available to users of this approach in relation to sample size requirements and choice of richness estimator. Desmet & Cowling (2004) suggested a minimum sample size of 30, to ensure stable estimates of richness. However, we found that this stability threshold is estimator-dependent and that it was possible to produce a stable result with a sample size as low as 14 (Table 2). Previous studies also implicitly recommend using the Bootstrap-based approach because it generally produces the most conservative targets (Desmet & Cowling, 2004; Rondinini, 2011a) but our results indicate that this estimator is the least likely to produce robust results. One way to overcome such problems would be to encourage conservation planners to adopt a highly standardized sampling strategy before collecting data because, as sampling becomes more exhaustive, this tends to produce more accurate estimates. This is because estimators will generally converge towards the same estimate of species richness (Colwell & Coddington, 1994; Borges *et al.*, 2009), thereby providing a more reliable basis for setting targets. However, this will not always be possible, so we also need research on how to achieve post hoc sampling parity between habitats, as simply using an equal number of samples per habitat type may over-sample habitats with a small extent of occurrence.

Applying SAR-based targets in conservation planning

There is often a near-linear relationship between habitat targets and the extent of the resulting PA networks identified (Rodrigues & Gaston, 2001; Warman *et al.*, 2004; Carpentier *et al.*, 2009; Delavenne *et al.*, 2012). Thus, setting unjustifiably high targets produces unnecessary impacts on the lives and activities of stakeholders (Chittaro *et al.*, 2010; Mascia *et al.*, 2010) and increases the costs associated with developing and managing the resulting PA systems (Naidoo *et al.*, 2006; McCrea-Strub *et al.*, 2011). We found that the national targets estimated for the MCZ projects (and applied by Balanced Seas) were between 18% and 92% higher than those estimated by this study for the four EUNIS level 3 habitats (Table 4), which implies an MPA network that would be 56.7% larger if the MCZ targets were applied to the whole EEC. This is a large discrepancy, and so it is important to understand the differences in results and the level of uncertainty associated with each, especially as both studies used the same approach and the same richness estimator. The main source of difference appears to be in the sample size because the targets developed for the Balanced Seas project were based on national-level data, and the number of sampling points for each habitat type was between 2 and 3 orders of magnitude higher than for this study (Table 4). In addition, these national MCZ targets were based on all species recorded within the Marine Recorder database (Rondinini, 2011a,b), whereas this study only used species obtained from macrobenthic surveys, and these different sets of species may show different biogeographical patterns.

This further supports the need for approaches that adjust percentage targets for sampling effort to produce results that account for total and per-habitat differences in sampling effort. It also emphasizes that systematic conservation planning has to be seen as an adaptive process that accounts for improvements in data quality over time (Margules & Pressey, 2000). The MCZ projects have followed this adaptive approach and gradually improved the quality of their ecological, socio-economic and resource-use data during the length of their project, as the UK Government recognized that this approach was the best compromise between accuracy and urgency. However, these MCZ networks are likely to be further modified, as part of a regular review process, and to form only part of marine spatial planning policy in the UK, so we would recommend that additional research on target setting is undertaken to inform these future developments. This research could also investigate the appropriateness of the current form of the SAR underpinning this approach

Table 3 Species richness estimates and targets (values calculated using the Bootstrap estimator and rounded to nearest whole number) for each EUNIS levels 3 and 4 habitat with increasing sample size.

			Number of sampling points used to generate estimates of species richness													
EUNIS code	EUNIS habitat description	Number of observed species	5	% Target	10	% Target	20	% Target	50	% Target	100	% Target	200	% Target	300	% Target
A3.3	Low-energy infralittoral rock	60	46	5.98	71	11.16	–	–	–	–	–	–	–	–	–	–
A4.3	Low-energy circalittoral rock	142	178	6.25	–	–	–	–	–	–	–	–	–	–	–	–
A5.1	Sublittoral coarse sediment	1520	252	2.61	394	5.88	563	9.03	823	12.59	1039	14.81	1257	16.61	1384	17.52
A5.13	Infralittoral coarse sediment	971	210	3.05	324	6.63	460	10.02	672	13.87	848	16.24	1019	18.08	–	–
A5.14	Circalittoral coarse sediment	1326	274	2.71	419	5.92	589	8.99	845	12.47	1052	14.61	1271	16.44	1400	17.38
A5.15	Deep circalittoral coarse sediment	825	232	3.18	365	6.92	527	10.46	787	14.49	–	–	–	–	–	–
A5.2	Sublittoral sand	714	87	3.56	138	7.57	210	11.77	334	16.54	460	19.75	611	22.51	709	23.91
A5.23 or A5.24	Infralittoral fine sand or muddy sand	590	87	3.94	139	8.27	208	12.47	335	17.51	460	20.76	604	23.46	–	–
A5.25 or A5.26	Circalittoral fine sand or muddy sand	454	88	4.15	136	8.23	200	12.27	312	17.02	430	20.36	–	–	–	–
A5.27	Deep circalittoral sand	128	73	5.20	120	10.31	–	–	–	–	–	–	–	–	–	–
A5.3	Sublittoral mud	198	91	4.51	139	9.04	202	13.44	–	–	–	–	–	–	–	–
A5.33 or A5.34	Infralittoral sandy mud or fine mud	139	82	5.42	127	10.41	–	–	–	–	–	–	–	–	–	–
A5.35 or A5.36	Circalittoral sandy mud or fine mud	131	104	4.34	151	8.44	–	–	–	–	–	–	–	–	–	–
A5.4	Sublittoral mixed sediments	333	106	3.36	162	7.26	233	11.13	354	15.74	–	–	–	–	–	–
A5.44	Circalittoral mixed sediments	245	99	3.22	143	6.65	197	10.12	287	14.41	–	–	–	–	–	–
A5.45	Deep mixed sediments	164	107	3.80	167	8.17	–	–	–	–	–	–	–	–	–	–

Table 4 Habitat-specific z-values and targets for four broad-scale EUNIS level 3 habitats developed for the eastern English Channel (EEC) in this study, and as provided by the Marine Conservation Zone (MCZ) Ecological Network Guidance in the UK (JNCC & Natural England, 2010).

EUNIS code	EUNIS habitat description	Area (km^2) of habitat in EEC	Number of EEC sampling points	EEC habitat z-values	EEC target (%)	Number of MCZ sampling points	MCZ habitat z-values*	MCZ target (%)	Difference in habitat area (km^2)
A5.1	Sublittoral coarse sediment	29,889	725	0.14	19.23	8532	0.19	32.40	3936.38
A5.2	Sublittoral sand	7633	469	0.16	25.28	9065	0.18	29.90	352.64
A5.3	Sublittoral mud	335	28	0.12	15.49	2064	0.17	29.80	47.94
A5.4	Sublittoral mixed sediments	900	64	0.13	16.88	1922	0.18	31.90	135.18

*MCZ habitat-specific z-values based on estimates of the average area of samples (x_1) being 0.5 m^2 (see Rondinini, 2011a).

(i.e. the power function) as previous work has shown that alternative functional forms, or mixes of these forms, are sometimes more appropriate (Stiles & Scheiner, 2007; Guilhaumon et al., 2008, 2010; Smith, 2010).

Policy-driven and SAR-based targets

The most widely known example of a conservation target defined by socio-political feasibility is the 10% target for world protected area coverage (IUCN, 1993). This figure was subsequently adopted by the CBD in 2004 whereby 10% of 'each of the world's ecological regions' was to be 'effectively' conserved by 2010 (CBD, 2004). However, at the 10th Conference of the Parties (COP), the proportion of terrestrial land area targeted for conservation was increased to 17%, whilst the proportion of the earth's oceans targeted for conservation remained at 10% (CBD, 2010; Harrop & Pritchard, 2011). The use of such policy-based conservation targets has been heavily criticized in recent years with some scientists suggesting that they are ecologically irrelevant, undermine the goal of biodiversity protection, foster the assumption that every habitat type needs to be equally protected and create the false expectation that such targets are sufficient for biodiversity representation and persistence (see review by Carwardine et al., 2009). Our results suggest that the application of the 10% policy-driven habitat target would fail to represent the majority of species in the EEC adequately (Table 1) and are consistent with results from other studies (Desmet & Cowling, 2004; JNCC & Natural England, 2010; Rondinini, 2011a).

However, there are two reasons why these policy-driven targets nevertheless play a valuable role. First, they are generally time bound and encourage governments to increase the extent of their MPA systems. Thus, the 10% targets should be seen in the context that only 0.05% of the total ocean area and 5.9% of territorial seas are currently designated as MPAs (CBD, 2010). Second, there are many occasions where there are insufficient data to develop SAR-based targets and so lower, policy-based targets can be used as an interim solution, pending availability of suitable data. For example, we could not set targets for four of the EUNIS level 3 and two of the EUNIS level 4 habitat types in the EEC because of a lack of data. Therefore, our results suggest that policy-based targets can play a role as long as (i) conservation practitioners are aware that they should be used as an interim measure whilst SAR-based targets are being developed and (ii) policy-based targets are low enough to ensure that no habitat type is over-represented in any eventual MPA system.

CONCLUSION

The SAR-based approach to setting habitat targets was developed to achieve two related goals. First, it provides a transparent and objective method for converting judgements of minimum species representation into a quantitative target. Second, it provides an approach for distinguishing between different habitat types and so tailors targets to account for differences in patterns of species richness and turnover. Our analysis shows that this approach can achieve these goals, but that issues relating to sample size (which are largely related to survey effort) and estimator choice have the potential to confound real differences between habitat types. Therefore, if this existing approach is to be applied to conservation decisions, there is a need for substantial research on techniques for producing target estimates that account for sample size and survey effort to address any issues of under-sampling. In the meantime, conservation practitioners should make use of best-available data and techniques to set habitat targets. They should also be aware that, where insufficient data are available to enable SAR-based target setting, time-bound policy targets offer a valid baseline whilst waiting for tailored targets to be developed.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Figure S1 Broad-scale EUNIS level 3 marine habitat map.

Table S1 Key to EUNIS codes, levels, and descriptions.

Table S2 Species richness estimates calculated using the ICE, Chao2, Jackknife1, Jackknife2 and Bootstrap estimators.

Table S3 Number of sampling points required to reach stable estimates of species richness for the ICE, Chao2, Jackknife1, Jackknife2 and Bootstrap estimators of species richness.

Table S4 Habitat-specific *z*-values calculated using the ICE, Chao2, Jackknife1, Jackknife2 and Bootstrap estimators of species richness.

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BIOSKETCHES

Kristian Metcalfe, **Juliette Delavenne**, **Sandrine Vaz**, **Stuart Harrop** and **Bob Smith** work on the marine spatial planning component of the European funded, Channel Integrated Approach for Marine Resource Management (CHARM) Phase III Project. **Clément Garcia**, **Aurélie Foveau**, **Jean-Claude Dauvin** and **Roger Coggan** work on other components of the project that deal with habitat and species distribution modelling. CHARM aims to integrate a range of biological, socio-economic, social and legal data to help develop a multidisciplinary approach for managing the English Channel.

Author contributions: K.M., S.R.H. and R.J.S. conceived the idea; J.D., C.G., A.F., J-C.D., R.C., S.V. and K.M. collected the relevant data; K.M. analysed the data; and K.M. and R.J.S. led the writing.

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