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The diversity benefits of marine artificial structures: A global meta-analysis with implications for offshore wind farms and nature-inclusive design

Katherine Richards *, Pippa J. Moore

Dove Marine Laboratory, School of Natural and Environmental Sciences, Newcastle University, Newcastle-upon-Tyne, UK

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ABSTRACT

The expansion of marine artificial structures (MAS), particularly offshore wind farms (OWFs), has raised questions about their ecological role and potential to enhance diversity. While MAS can function as artificial reefs, evidence remains fragmented, and the effectiveness of nature-inclusive design (NID) in MAS is poorly understood. This study synthesises global evidence on diversity responses to MAS, with a focus on OWFs and NID. Literature searches following PRISMA guidelines identified 104 relevant studies, of which 55 provided 367 effect sizes for a meta-analysis following outlier removal.

MAS significantly increased diversity relative to controls (SMD $= 0.899 \pm 0.122$ SE, $P < 0.001$), though responses were highly context dependent. The strongest gains were observed for oil and gas platforms, artificial reefs, and coastal infrastructure, while OWFs showed weaker, non-significant effects, although responses were context dependent. Diversity responses increased with structure age and were more pronounced for sessile than mobile taxa in OWFs. NID interventions enhanced diversity in coastal infrastructure, artificial reefs, and add-on units. However, studies assessing NID in OWFs were limited, and as direct evidence remains scarce, conclusions on their effectiveness in this context are largely inferential. A halo effect was detected, with increases in diversity extending ~50 m into surrounding soft sediments.

These findings highlight that while MAS can enhance diversity, outcomes are highly variable and context dependent. For OWFs, realising ecological benefits will require broader application of NID, long-term monitoring, designs tailored to site-specific ecological objectives, and stronger evidence to guide policy and industry adoption.

1. Introduction

The expansion of human activities in the marine environment has contributed to the proliferation of marine artificial structures (MAS) (Kamermans et al., 2025; ter Hofstede, 2024), defined as human-made installations placed in marine and coastal environments that introduce hard substrate, including energy infrastructure (e.g. offshore wind farms and oil and gas platforms), coastal defence structures (e.g. seawalls and breakwaters), and purpose-built artificial reefs (O'Shaughnessy et al., 2020; McKibbin et al., 2024; Szostek et al., 2025). This expansion, often referred to as "ocean sprawl" (Evans et al., 2019; Firth et al., 2016), is a consequence of increasing demands on marine space for activities such as resource extraction, energy production, shipping, fisheries, aquaculture, and tourism, as well as the need for coastal protection. Collectively, MAS now occupy over 32,000 km^2 of the marine environment, with continued growth of at least 23% expected by 2028 as demands on marine space increase (McKibbin et al., 2024).

A major contributor to ocean sprawl is energy production infrastructure, with a recent surge in renewable energy developments, particularly offshore wind farms (OWFs) (Bergström et al., 2014; Tougaard et al., 2020). Globally, OWFs are central to meeting climate mitigation targets by reducing dependency on fossil fuels (Lemasson et al., 2024). To meet net zero emission targets by 2050, an estimated 2000 GW of OWF capacity will need to be installed worldwide, up from just 35 GW in 2020, requiring approximately 5000 new turbines each year and occupying more than 500,000 km^2 of ocean space (Watson et al., 2024). Europe remains at the forefront of this expansion, targeting 450 GW of offshore wind capacity by 2050, 48% of which will be concentrated in the North Sea (Buyse et al., 2023; Kingma et al., 2024), where over 3500 turbines are already installed (Fowler et al., 2018). The UK is considered a global leader in offshore wind, supplying 45% of Europe's energy capacity in 2022 (Szostek et al., 2025), with targets to

* Corresponding author.
E-mail addresses: richardskatherine2002@gmail.com (K. Richards), Pip.Moore@newcastle.ac.uk (P.J. Moore).

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reach 50 GW by 2030 (GOV.UK, 2023). As spatial demands of OWFs continue to grow, competition with other marine industries is increasing, contributing to 'ocean squeeze', where available space becomes progressively limited (Pardo et al., 2023; Steins et al., 2021).

This expansion has prompted discussions on how best to balance renewable energy production with the protection of biodiversity and ecosystem services (Pardo et al., 2023; Windt et al., 2024). In England, one approach has been the introduction of Biodiversity Net Gain (BNG), a policy requirement designed to ensure all developments deliver at least a 10% net increase in biodiversity, rather than merely minimising harm (GOV.UK, 2024). At the international scale, the UN SDGs, the Convention on Biological Diversity, and the Kunming–Montreal Global Biodiversity Framework all emphasise the need to prevent biodiversity loss and promote synergies between environmental, social and economic objectives (Dannheim et al., 2020; Evans et al., 2019; Galparsoro et al., 2022; Pardo et al., 2023; Windt et al., 2024).

MAS have historically been designed to serve specific human purposes with little regard for the habitats they displace or the ecological communities they alter (Firth et al., 2016). As a result, they can exert both positive and negative impacts on the marine environment across their development stages (Berges et al., 2024), with the most negative impacts occurring during construction and decommissioning through noise, vibration, seabed disturbance and habitat loss (Abramic et al., 2022; Berges et al., 2024; Juretzek and Schmidt, 2021; Pardo et al., 2023). However, once operational, they can provide positive ecological outcomes, primarily through the artificial reef effect (Cosgrove, 2024). This process refers to the introduction of artificial hard substrates into marine environments, transforming soft-bottom habitats into hard-substrate systems (Baulaz et al., 2023; Buyse et al., 2023; ter Hofstede and van Koningsveld, 2024), facilitating the colonisation of sessile and mobile organisms, leading to changes in species distributions, community composition, biomass, and ecosystem functioning (Firth et al., 2016; Baulaz et al., 2023; Lemasson et al., 2024). Increases in biomass, diversity, species coexistence, and ecosystem productivity are often observed through increased niche availability (Firth et al., 2016; Causon and Gill, 2018; Berges et al., 2024; ter Hofstede, 2024), however, ecological outcomes differ with habitat connectivity, isolation and larval dispersal, consistent with island biogeography theory (Macarthur and Wilson, 1967). Responses can also vary with habitat type, taxonomic group, ecological metrics, and factors such as structural complexity, material type, structure age, and deployment depth of MAS Lemasson et al. (2024). The artificial reef effect can also provide additional ecosystem services including food provision, improved fisheries yield, and stock resilience (Evans et al., 2019; Firth et al., 2024), as well as greater storage of organic carbon that supports climate regulation (Causon and Gill, 2018). MAS may also create a 'reserve effect', acting as protected areas by limiting anthropogenic activities (Berges et al., 2024; Causon and Gill, 2018; Firth et al., 2016; ter Hofstede, 2024), encouraging population recovery and reduced mortality (Abramic et al., 2022; Bergström et al., 2014; Buyse et al., 2023). These refuges can support fisheries through ecological spillover, increasing fish biomass and abundance in the surrounding areas (Baulaz et al., 2023; Causon and Gill, 2018; Galparsoro et al., 2022), with diversity benefits extending beyond the physical footprint of the structures into adjacent habitats through a broader 'halo effect', defined as the spatial extension of ecological influences into surrounding areas, the extent of which is context dependent (Buyse et al., 2023; Pardo et al., 2023).

While MAS can promote diversity, they typically support less diverse and abundant communities when compared to natural reefs, often due to their limited structural complexity and the use of non-natural materials (Firth et al., 2016; ter Hofstede and van Koningsveld, 2024). These limitations can be addressed by incorporating nature-inclusive design (NID), which refer to the intentional incorporation of ecological features, knowledge, methods, and technologies into MAS, typically through the addition or modification of materials, surface complexity, or habitat-enhancing units, with the explicit aim of increasing biodiversity and ecological functioning, and supporting the restoration of degraded ecosystems (Causon and Gill, 2018; ter Hofstede and van Koningsveld, 2024; Windt et al., 2024; Pardo et al., 2023). NID aims to shift development from solely minimising ecological harm to actively enhancing environmental value (Strain et al., 2018). Enhancing structural complexity through NID has been shown to increase diversity, reflecting the habitat heterogeneity hypothesis, and can strengthen the provision of ecosystem services across the operational lifetime of marine infrastructure (Bennun et al., 2021; Causon and Gill, 2018; Firth et al., 2016; Kingma et al., 2024; Pardo et al., 2023).

Widely applied approaches include increasing surface area across multiple spatial scales, from millimetres to metres (Strain et al., 2018), and replacing traditional materials with alternatives such as enriched concrete or calcareous rocks (e.g. limestone, marble, granite) to promote shellfish settlement and improve diversity outcomes (Kingma et al., 2024; ter Hofstede et al., 2022). Where ecological considerations are not incorporated at the design phase of infrastructure, habitat enhancing units, referred to as add-ons, can be retrofitted to existing infrastructure (Strain et al., 2018). These include devices such as *Biohuts*®, reef cubes or balls, and fish hotels, which provide additional habitat complexity and niches (Hermans et al., 2020; Strain et al., 2018; Windt et al., 2024; Zupan et al., 2024). Artificial reefs, defined as man-made structures intentionally constructed or placed on the seabed to mimic ecological functions of natural reefs (Lemasson et al., 2024; Vivier et al., 2021), can also be used to offset hard substrate habitat loss, protect sites from damaging activities such as bottom trawling, and provide ecosystem services including enhanced fisheries productivity and nutrient cycling (Hackradt et al., 2011; Lemasson et al., 2024). Their performance depends heavily on design, with increased surface complexity, addition of holes, time since placement, and depth, strongly influencing ecological outcomes (Vivier et al., 2021). When placed on sedimentary habitats, the additional hard substrate provided can lead to diversity levels that exceed those of surrounding soft-bottom habitats (Lemasson et al., 2024), although this represents a shift in habitat type and associated communities rather than restoration of the natural system (Firth et al., 2016). Together, these approaches demonstrate how NID can help deliver diversity targets alongside renewable energy goals through improvements in infrastructure design (Firth et al., 2024; Kingma et al., 2024; ter Hofstede et al., 2022), however, their potential is underexplored (Hermans et al., 2020; Pardo et al., 2023; Windt et al., 2024).

Despite the growing recognition of NID, its implementation in the UK remains limited by a number of persistent barriers (Evans et al., 2019). A key limitation is the lack of confidence in the underlying evidence base, with concerns over the reliability and accessibility of evidence (Evans et al., 2019). Broader ecological benefits, including their value for conservation-priority species, are often underappreciated, and awareness among practitioners remains limited (Evans et al., 2019). Additional barriers include uncertainties in the design process (Pardo et al., 2023), and technical or ecological risks, such as the potential facilitation of non-native species (Firth et al., 2016; Pardo et al., 2023). Finally, the decommissioning phase also presents challenges, as current regulations often require the full removal of structures, which may negate long-term ecological benefits through the loss of reef habitat and associated communities (Pardo et al., 2023), reduction in biological connectivity, and increased atmospheric emissions (Fowler et al., 2018; Gorman et al., 2023).

In contrast, some North Sea countries, such as the Netherlands, have mandated NID in OWF developments to restore biogenic reefs, including the threatened European flat oyster (*Ostrea edulis*), under OSPAR and EU biodiversity objectives (Bos et al., 2023; Kamermans et al., 2018; ter Hofstede et al., 2022). With pilot projects emerging in the North Sea, the UK also has an opportunity to integrate NID features into OWF developments (Pardo et al., 2023). OWF components such as foundations, scour protection and cable protection offer potential for NID, with evidence showing that the use of diverse materials, textures and dimensions can influence diversity outcomes (Hermans et al., 2020; Kingma et al., 2024; ter Hofstede and van Koningsveld, 2024; Windt et al., 2024). Realising this potential requires improved communication between sectors, greater stakeholder engagement, and the development of a stronger evidence base to demonstrate environmental benefits (Firth et al., 2016). Aligning UK policies with global conservation frameworks and recognising the long-term ecological role of OWFs could incentivise NID adoption and ensure that diversity gains are sustained (ter Hofstede and van Koningsveld, 2024). However, evidence on the diversity outcomes of MAS and the effectiveness of NID remains fragmented, with limited synthesis of how multiple factors, including structure type, NID interventions, structure age, and distance from structures, interact to influence diversity outcomes, constraining the ability of policy and industry to make informed decisions (Evans et al., 2019). Addressing this evidence gap is therefore critical to guiding future offshore development in the UK and beyond.

This review aims to critically evaluate how MAS influence diversity, with a particular focus on OWFs, and to evaluate the potential of NID to enhance ecological outcomes. Specifically, it investigates variations in diversity responses across structure types, ecological metrics, and taxonomic groups through meta-analyses, while also considering the influence of factors such as structure age, distance from structures, and the use of NID interventions. In doing so, we determine the extent to which OWFs contribute to diversity relative to other MAS and to provide an evidence base that can inform future infrastructure design and regulatory guidance, supporting the integration of NID into UK OWF development.

We hypothesise that MAS will increase diversity relative to surrounding soft-sediment habitats, that diversity will increase with structure age, and that NID interventions will further enhance diversity through increased habitat complexity and niche availability.

2. Methodology

2.1. Data retrieval

A systematic literature search was conducted between 15th – 16th May 2025 using Scopus and Web of Science to identify studies assessing the effects of MAS on marine diversity. The search strategy followed PRISMA guidelines (Page et al., 2021) to ensure transparency and replicability. Full search strings are provided in the supplementary material (SM Table S1). Studies were eligible for inclusion if they (i) investigated ecological impacts of MAS compared to unstructured soft-bottom habitats, (ii) compared MAS with and without interventions defined as NID, where NID was restricted to intentional structural modifications or additions designed to enhance habitat complexity and diversity (SM Table S2 for classification), (iii) reported diversity metrics for both control and intervention sites, and (iv) provided sufficient data to calculate effect sizes (i.e. a standardised measure of the difference between control and intervention groups, derived from mean values, measures of variance, and sample sizes).

In addition to published studies, grey literature (e.g. industry reports, government documents) was reviewed using targeted searches of relevant amplifier organisations (e.g. The Rich North Sea (2024); Conservation Evidence (Evans et al., 2021)). Manual hand-searches of these resources were conducted until 31st May 2025. Only English-language publications were included, though studies from all global regions were considered.

All articles identified during searches were screened for duplicates, then assessed for inclusion based on the title and abstract. Full articles of potentially relevant studies were retrieved for further assessment. Articles of uncertain relevance were retained and reassessed at the next stage. A PRISMA flow diagram, detailed inclusion and exclusion criteria, and a list of studies assessed at the full-text stage (with reasons for inclusion or exclusion), are provided in the supplementary material (SM Fig. S1; Table S3; Table S4).

2.2. Data extraction

Of the 2320 studies returned through literature searches, 104 were identified as relevant to the research question. From these, 55 studies provided sufficient data to calculate an effect size, and were included in the meta-analysis. Data was extracted from text, tables, figures (using WebPlotDigitizer (Rohatgi, 2024)) and/or supplementary online materials. A range of ecological, structural, and methodological variables were extracted from each study, including continuous measures such as age of structures and distance of control sites (SM Table S2; Table S5). To facilitate statistical modelling, these were converted into categorical bins, with boundaries chosen based on the distribution and natural breaks in data.

For studies reporting multiple response variables, data was extracted for each observation. Where observations were carried out across a time series, only the final time point was used to avoid issues of temporal autocorrelation. If the first observation occurred before MAS or NID deployment, both the first and last points were taken to enable a before–after comparison. For studies that reported seasonal observations without a time-series design, data was extracted for each season. When geographical coordinates were not provided, geographical position was estimated using Google Maps. Finally, in cases where studies failed to declare what form of variability was used, standard deviation (SD) was assumed as recommended by Borenstein et al. (2021). Where variance was reported as standard error (SE) or as confidence intervals, these were converted to SD.

For studies that measured ecological responses along transects at multiple distances from MAS, effect sizes were first calculated by comparing the site closest to the structure (impact site) with the site furthest along the transect (control site). To explicitly evaluate whether diversity responses varied with distance (i.e. testing for a halo effect), data from all transect distances were compiled into a separate dataset. In this separate analysis, the impact site was consistently defined as the location closest to the MAS, and effect sizes were calculated relative to control sites at increasing distances (e.g. 25 m, 50 m, 100 m, 200 m).

2.3. Meta-analysis

All statistical analyses were performed using R 4.5.1 (R Core Team, 2025), using the *metafor* package (Viechtbauer, 2010) for the meta-analysis, alongside *orchaRd* (Nakagawa et al., 2023) and *ggplot2* (Wickham, 2016) packages for visualisation.

To account for methodological and ecological heterogeneity and the non-independence of observations within studies, a multilevel, meta-analytic model was applied, with study ID included as a random effect to model shared variance at the study level. Structure age was tested as a moderator due to its expected influence on ecological responses through successional development over time. Effect sizes were calculated as Standardised Mean Difference (SMD), which accounts for small sample bias through a correction factor (J). A positive SMD indicated an increase in diversity due to MAS or NID, while negative values indicated a decline. Effect sizes were considered significant if their 95% confidence intervals did not overlap with zero. Observational groups with fewer than five effect sizes were excluded from subgroup analyses to avoid misleading conclusions based on small sample sizes.

SMD was calculated as:

$$SMD = \frac{M_I - M_C}{SD_{pooled}} \times J$$

Where M_I and M_C are mean values of the diversity metric for the intervention and control groups, respectively.

The pooled standard deviation (SD_{pooled}) was calculated as:

$$SD_{Pooled} = \sqrt{\frac{(N_e - 1)s_e^2 + (N_c - 1)s_c^2}{N_e + N_c - 2}}$$

Where N_e and s_e are the sample size and standard deviation for the intervention, respectively, and N_c and s_c are the sample size and standard deviation from the control, respectively.

The small sample size correction, J, was defined as:

$$J = 1 - \frac{3}{4(N_e + N_c - 2) - 1}$$

To ensure robustness of meta-analysis results, statistical testing was first carried out on the entire dataset, after which outliers were identified ($|z| > 1.96$) and removed. The resulting clipped dataset was used for all subsequent analyses. Sensitivity testing involved a leave-one-out analysis, in which each study was sequentially removed to assess its influence on overall results. Where the outcomes remained consistent, the analysis was considered robust. Publication bias was assessed visually using funnel plots, with asymmetry formally evaluated using Egger's regression applied to a corresponding univariate, random, mixed-effects model. A Rosenthal's fail-safe number was also calculated to estimate the number of hypothetically missing studies with null results required to reduce the overall effect to non-significance. Where the value exceeds the threshold of 5n + 10 (where n is the number of studies included in analysis), the analysis can be considered robust to publication bias (Rothstein et al., 2005). For the full analysis, with $n = 52$, this threshold was 285; for analysis on the halo effect, with $n = 10$, the threshold was 60.

Subgroup analyses were then performed on the refined dataset to explore potential moderators, including MAS or NID type, control type, structure age, diversity metric, organism functional group, and distance between control and impact sites. Pairwise comparisons were used to assess whether effect sizes differed significantly between moderator categories, with these contrasts carried out using the 'btt' argument of the 'anova()' function in the *metafor* package. Statistical significance was evaluated at $p < 0.05$.

3. Results

3.1. Studies included

Of the 104 studies retrieved through systematic searches, 55 met the inclusion criteria and were used for quantitative analysis, generating a total of 403 effect sizes (SM Fig. S1). Study publications ranged from 1992 to 2025 (Fig. 1b), with Europe (39%) and Australasia (19%) the most represented regions, with fewer studies conducted in North America, Asia, and South America (Fig. 1a). Notably, 44.4% of all NID-focused studies were from Europe. The majority of effect sizes were derived from purpose-built artificial reefs (33%), offshore wind farms (OWF, 19%), and add-on artificial habitat units (17%; see examples in Fig. 1d), with the remaining effect sizes associated with other forms of MAS and NID types (Fig. 1c). The most frequently reported diversity metrics were species richness (39.3%) and total abundance (37.1%), with fewer effect sizes associated with other measures (Fig. 1e).

A total of 36 observations were flagged as outliers ($|z| > 1.96$) and removed, resulting in 367 effect sizes (SM Table S6, Fig. S2). A re-run of the meta-analysis on the reduced dataset returned similar estimates to the analysis on the dataset prior to outlier removal, suggesting results were robust to outlier removal. The overall multilevel meta-analysis revealed a significant positive effect of MAS on diversity outcomes (SMD = 0.899 ± 0.122 SE, $P < 0.001$; SM Fig. S2). Substantial heterogeneity across effect sizes remained (Q (df = 366) = 1233.407, P < 0.001), with variance partitioned between study-level and observation-level components (σ^2_ID = 0.596, σ^2_observation = 0.311), indicating that variation was not solely due to sampling error and may be attributable to other factors explored further in moderator analyses.

As structure age was expected to influence ecological responses through successional development, it was tested as a continuous moderator. When included in the model, structure age did not significantly explain variation in effect sizes (QM (1) = 2.145, $p = 0.143$), and the overall effect remained consistent (SMD = 0.810 ± 0.126 SE, P < 0.001). As age data were not available for all studies, reducing the dataset, this variable was not included in subsequent global models. Non-significance may suggest that any influence of age may be non-linear, which was explored further using categorical age classes (see Section 3.4).

Leave-one-out analysis revealed a stable overall effect size, with no individual study strongly influencing results (range: SMD = 0.853–0.93). Variance components remained broadly consistent throughout (σ^2_ID range = 0.515–0.636; σ^2_observation range = 0.192–0.339), indicating that no single study disproportionately influenced overall effect estimates or heterogeneity structure, reinforcing the robustness of the findings. Publication bias was also evaluated using both visual and statistical approaches, with results suggesting small-study effects or potential publication bias (SM Section 2.1.1). To further evaluate the robustness of the observed effect size to potential publication bias, a Rosenthal's fail-safe number indicated that 61,558 ($p < 0.001$) additional null-effect studies would be required to reduce the overall effect to non-significance ($P > 0.050$), suggesting robust findings, despite possible asymmetry.

3.2. Intervention type

While all MAS, with the exception OWFs, supported significantly higher diversity compared to controls, there were differences between MAS types (QM (6) = 70.801, $p < 0.001$), with pairwise comparisons revealing significant differences between all MAS types (Fig. 2a). The strongest positive effects were observed for coastal infrastructure and O&G, followed by artificial reefs and other energy infrastructure (SM Table S7), likely reflecting greater structural complexity and long-term habitat availability. Add-ons also demonstrated a significant positive effect, whereas OWFs did not differ significantly from controls, potentially due to lower structural complexity or younger infrastructure age.

When comparing MAS to soft sediment reference areas (Fig. 2b, k = 230), structure type significantly influenced diversity outcomes (QM (5) = 39.215, $p < 0.001$). Pairwise comparisons revealed significant differences among MAS types, with the exception of OWFs and coastal infrastructure (Fig. 2b). O&G, artificial reefs and other energy infrastructure all showed significantly positive effects for diversity compared to controls, while OWFs and coastal infrastructure had positive but non-significant effects (SM Table S8), suggesting that not all MAS provide equivalent habitat complexity or ecological function.

MAS with NID interventions significantly affected diversity outcomes compared to MAS without NID (k = 137; QM (7) = 35.5, $p < 0.001$; Fig. 2c). Pairwise comparisons showed that most NID MAS differed significantly from each other, with the exception of simpler artificial reef designs (C1 and C2) and energy infrastructure and OWFs (Fig. 2c). Coastal infrastructure with NID, and the addition of add-ons, significantly increased diversity relative to non-NID MAS (SM Table S9). Among artificial reefs, only the most complex design (C3) showed significant positive effects, while simpler designs did not differ significantly from MAS without NID. Notably, OWFs and other energy infrastructure showed no significant diversity gains with NID, however, the only NID applied to OWFs was an increased complexity of scour protection which may provide limited additional habitat compared to more complex interventions.

3.3. Ecological responses

The effects of MAS on diversity varied significantly depending on the ecological response measured (QM (6) = 62.763, $p < 0.001$; Fig. 3a). Pairwise tests indicated significant differences among most metrics, with the exception of evenness and taxon richness (Fig. 3a). Significant positive effects were observed for all diversity measures except for taxon richness and evenness, where no effect was shown (SM Table S10). When exploring the effects of different MAS types on the most

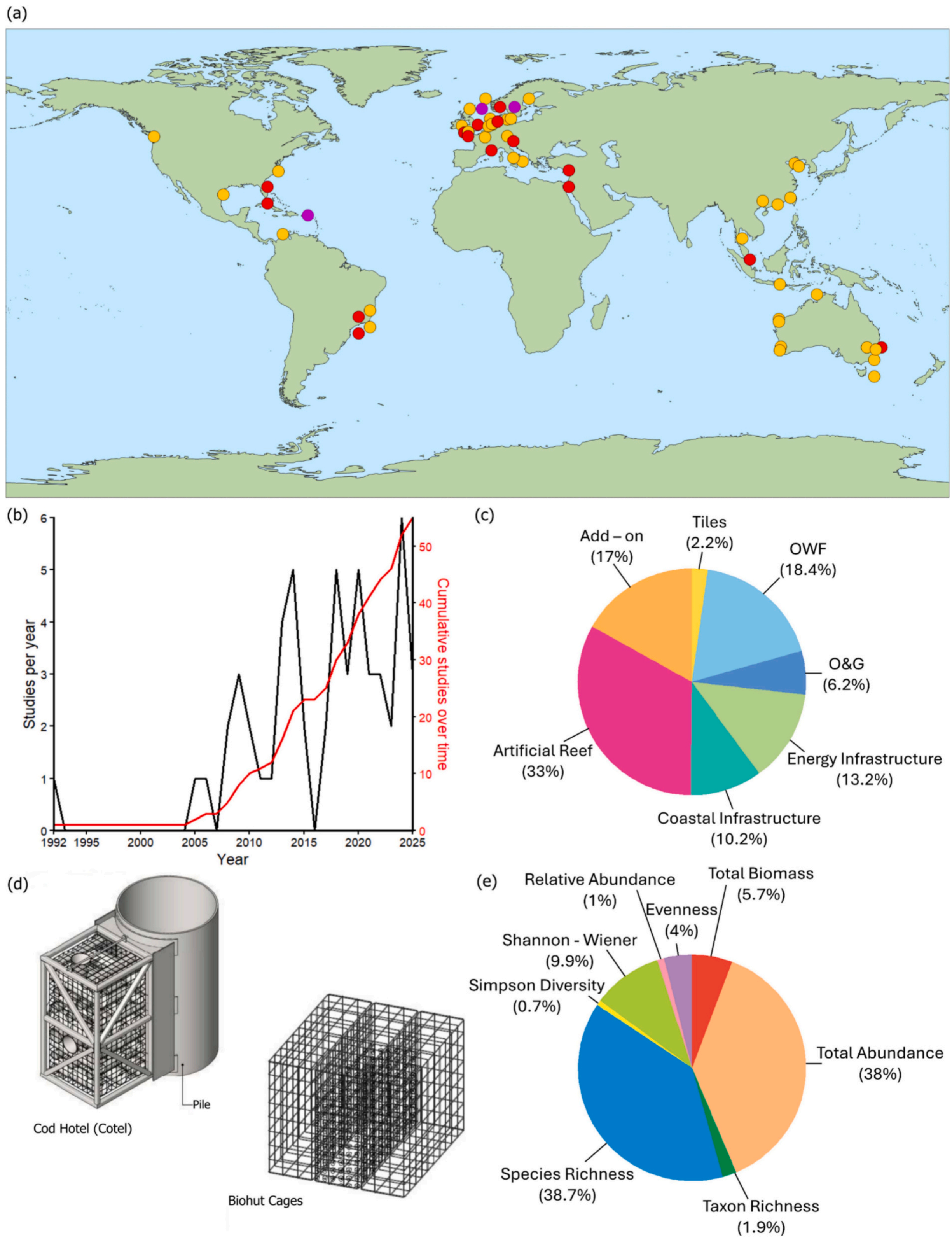

Fig. 1. (a) Global distribution of study sites included in the meta-analysis. Symbol colours indicate the type of control comparison: red - marine artificial structures (MAS) with nature-inclusive design (NID) compared to conventional MAS without NID; yellow - MAS compared to surrounding sediment reference areas; purple - studies that included both types of control comparisons. Basemap derived from Natural Earth land and ocean polygons (Free vector and raster map data @ naturalearthdata.com). (b) Number of studies included in the meta-analysis by year of publication shown as a black solid line (left axis), with cumulative total shown as a red line (right axis). (c) Proportional representation of MAS types across all observations included in the meta-analysis. Categories are offshore wind farms (OWF), oil and gas platforms (O&G), other energy infrastructure (e.g. pipelines, tidal/wave energy foundations, cables), coastal infrastructure (e.g. breakwaters, marinas), artificial reefs, and add-ons. (d) Examples of add-on options: cod hotel (cotel) and Biohut® cages based on Hermans et al. (2020). (e) Proportional representation of diversity metrics reported across all observations. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

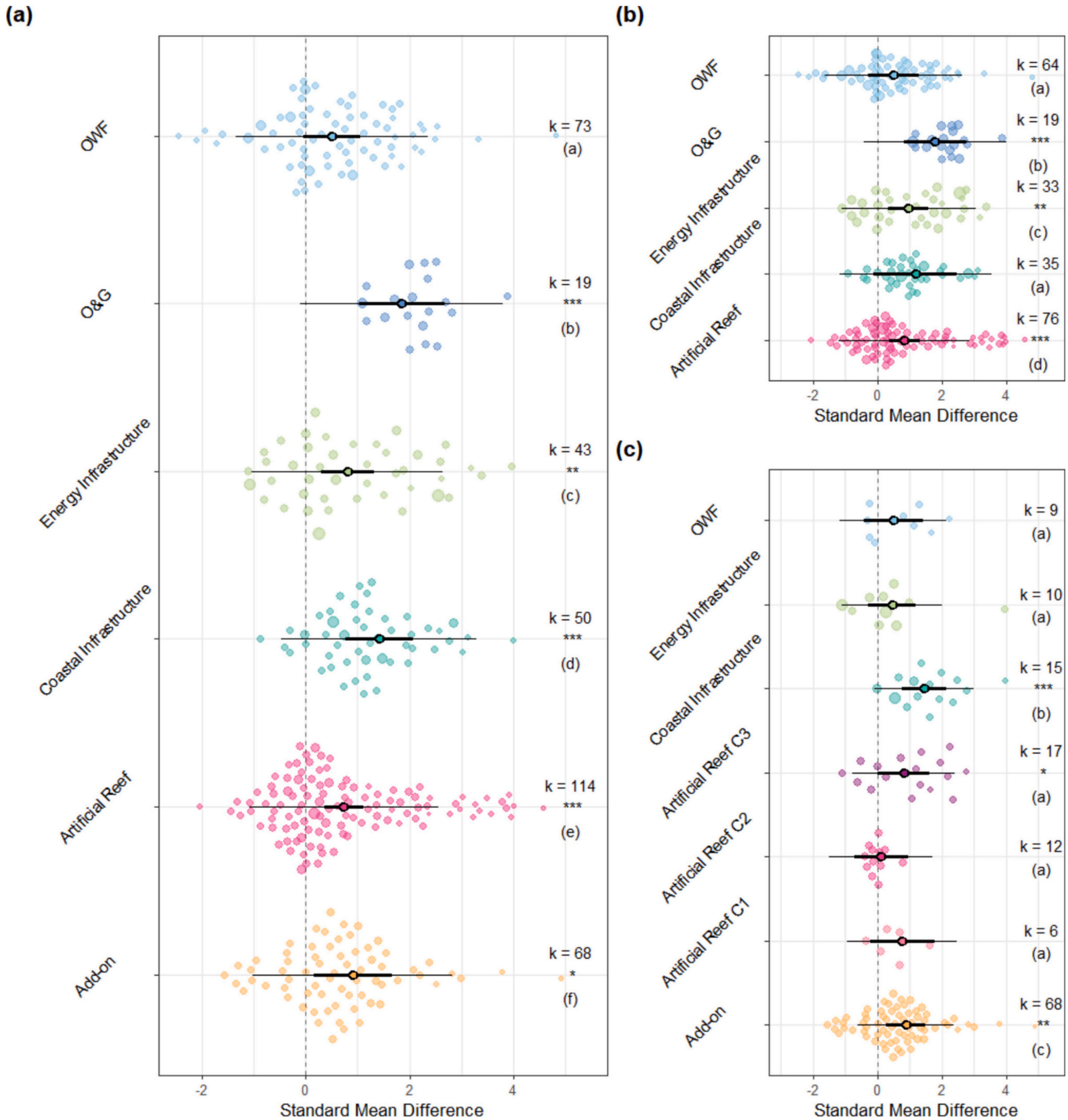

Fig. 2. Mean effect size (Standardised Mean Difference, SMD) for the impact of marine artificial structures (MAS) on diversity for (a) All effects across the full dataset. MAS categories include offshore wind farms (OWF), oil and gas structures (O&G), other energy infrastructure (e.g. pipelines, wave/tidal energy foundations, cables), coastal infrastructure (e.g. marinas, breakwaters), artificial reefs, and add-ons (retrofitted artificial habitat units). (b) Effects when MAS are compared to surrounding sediment habitats. (c) Effects when MAS with nature-inclusive design (NID) features are compared to conventional MAS. Artificial reef subcategories (C1 - least complex, C2, C3 - most complex) represent increasing levels of structural complexity. Each point represents an individual observation, with point size proportional to its precision (1/standard error, SE). Thick horizontal bars represent the 95% confidence intervals (CI) of each mean effect size, while thin error bars represent the 95% prediction interval. Effects are considered non-significant where the 95% CI cross the dashed zero line. Significance levels are indicated by asterisks: $p < 0.05$ (*), $p < 0.01$ (**), $p < 0.001$ (***). The number of observations is represented by k. Intervention types not sharing a letter differ significantly from one another based on pairwise tests (QM test, $p < 0.05$).

commonly reported diversity metrics ($k \geq 5$), there was a significant effect for species richness (QM (5) = 48.096, $p < 0.001$, Fig. 3b) and total abundance (QM (6) = 35.371, $p < 0.001$, Fig. 3c), and no effect on Shannon-Wiener diversity (QM (3) = 6.985, $p = 0.072$, Fig. 3d). For species richness, coastal infrastructure and artificial reefs were significantly different from other MAS types, having the strongest positive richness effect, along with add-on features (Fig. 3b, SM Table S11). For total abundance, O&G and artificial reefs saw the largest increases, with other intervention types showing weaker or non-significant effects (Fig. 3d, SM Table S12).

Taxa mobility significantly influenced the effects of MAS on diversity (QM (2) = 72.394, $p < 0.001$), with mobile taxa showing a strong

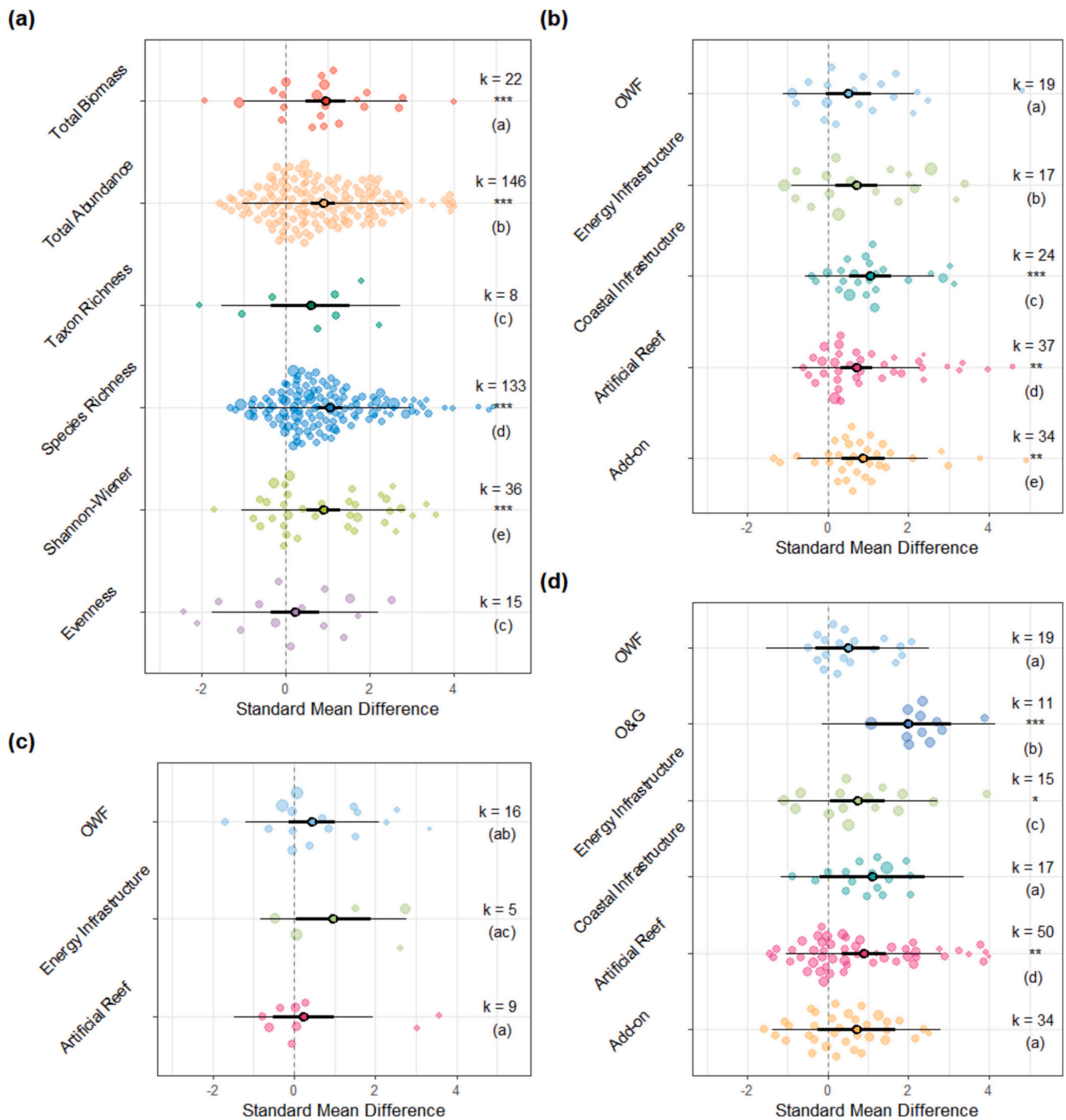

Fig. 3. Mean effect size (Standardised Mean Difference, SMD) for the impact of marine artificial structures (MAS) on diversity metrics for (a) All MAS on different diversity metrics. (b) Effects of different MAS on species richness. (c) Effects of different MAS on Shannon- Wiener diversity. (d) Effects of different MAS on total abundance. Each point represents an individual observation, with point size proportional to its precision (1/standard error, SE). Thick horizontal bars represent the 95% confidence intervals (CI) of each mean effect size, while thin error bars represent the 95% prediction interval. Effects are considered non-significant where the 95% CI cross the dashed zero line. Significance levels are indicated by asterisks: $p < 0.05$ (*), $p < 0.01$ (**), $p < 0.001$ (***). The number of observations is represented by k. Metrics and intervention types not sharing a letter differ significantly from one another based on pairwise tests (QM test, $p < 0.05$).

positive effect while sessile taxa were also significantly and positively influenced by MAS, but to a significantly lesser extent than mobile species (Fig. 4a, SM Table S13), suggesting differing responses to habitat availability and resource use. When focusing exclusively on OWFs, mobility also had a significant effect on the response of taxa (QM (2) = 9.505, $p < 0.008$). Sessile taxa showed a positive effect of OWFs, while mobile taxa showed non-significant negative effects, but the number of observations were limited ($k = 6$), and results should therefore be interpreted with caution (Fig. 4b, SM Table S14).

3.4. Age of structure

Structure age had a significant effect on diversity (QM (5) = 57.553, $p < 0.001$). Pairwise comparisons showed that all age categories differed significantly, except for the 'Old' category, which did not differ from others (Fig. 5b). 'Very Old' structures (> 180 months) had the strongest

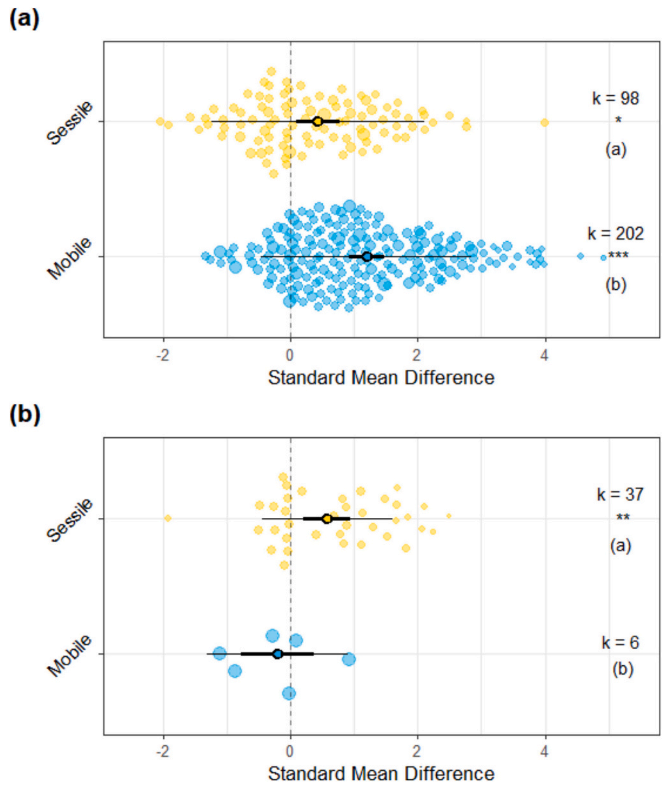

Fig. 4. Mean effect size (Standardised Mean Difference, SMD) for the influence of organismal mobility on the response of taxa to marine artificial structures (MAS) for (a) Responses of mobile versus sessile taxa to all MAS types. (b) Responses of mobile versus sessile taxa to offshore wind farms (OWFs) only. Each point represents an individual observation, with point size proportional to its precision (1/standard error, SE). Thick horizontal bars represent the 95% confidence intervals (CI) of each mean effect size, while thin error bars represent the 95% prediction interval. Effects are considered non-significant where the 95% CI cross the dashed zero line. Significance levels are indicated by asterisks: $p < 0.05$ (*), $p < 0.01$ (**), $p < 0.001$ (***). The number of observations is represented by k. The mobility types not sharing a letter differ significantly from one another based on pairwise tests (QM test, $p < 0.05$).

positive effects, followed by 'Mid' aged (19–60 months), 'New' (7–18 months), and 'Very New' (0–6 months) structures, consistent with successional development of communities over time. It is, however, worth noting that the youngest MAS types included in the analysis were artificial reefs and add-ons. The only non-significant effect was for 'Old' structures (61–180 months, Fig. 5a, SM Table S15). When focusing on OWFs, structure age also significantly influenced diversity (QM (3) = 16.135, $p < 0.001$), with older structures having a significant positive effect on diversity while 'New' and 'Very New' structures had non-significant positive and negative effects (Fig. 5b, SM Table S16), indicating that ecological benefits may take time to develop on OWFs.

3.5. Distance from MAS

The distance of control sites from MAS (SM Section 2.2) had a significant effect on diversity (QM (df = 5) = 65.718, $P < 0.001$). Pairwise comparisons revealed that there was no significant difference between diversity at the MAS and at distances up to 50 m away, suggesting a spatial gradient consistent with a potential halo effect (Fig. 6, SM Table S18), likely driven by local enhancement of surrounding habitats. In contrast, diversity was greater at the MAS and up to 50 m away compared to distances of 50-200 m away (Fig. 6).

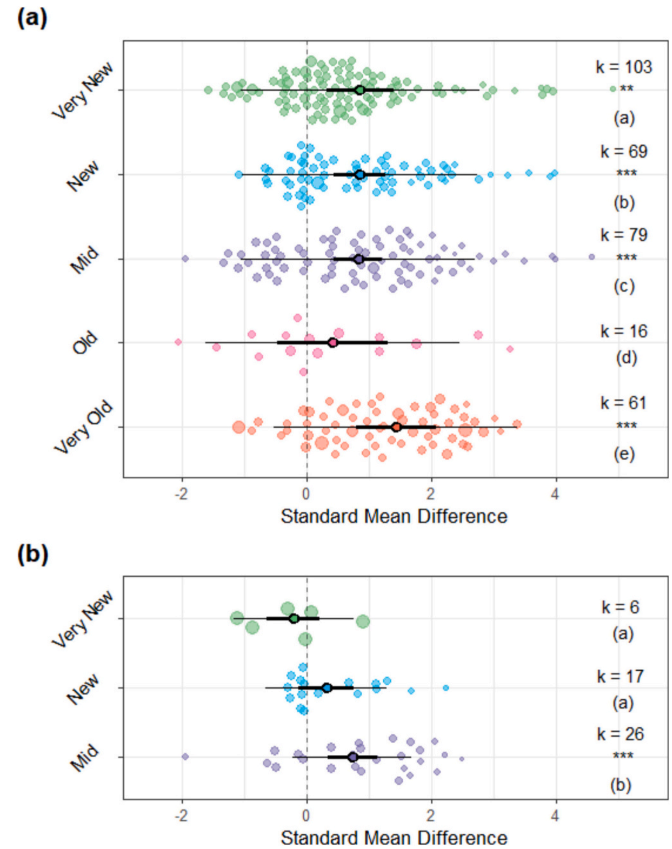

Fig. 5. Mean effect size (Standardised Mean Difference, SMD) for the influence of age of marine artificial structures (MAS) on diversity for (a) All MAS combined (b) OWFs only. Each MAS was categorised as: Very New (0–6 months), New (7–18 months), Mid (19–60 months), Old (61–180 months), and Very Old (> 180 months). Each point represents an individual observation, with point size proportional to its precision (1/standard error, SE). Thick horizontal bars represent the 95% confidence intervals (CI) of each mean effect size, while thin error bars represent the 95% prediction interval. Effects are considered non-significant where the 95% CI cross the dashed zero line. Significance levels are indicated by asterisks: $p < 0.05$ (*), $p < 0.01$ (**), $p < 0.001$ (***). The number of observations is represented by k. Age categories not sharing a letter differ significantly from one another based on pairwise tests (QM test, $p < 0.05$).

4. Discussion

This study demonstrates that MAS significantly alter diversity patterns, however, responses are strongly context dependent. While the meta-analysis showed that MAS generally enhance diversity relative to controls, the strength and consistency of this effect varied across structure type, ecological metric, functional group, structure age, and the distance of control sites, with these interacting influences synthesised conceptually in Fig. 7. This variability reflects that diversity enhancement is not a universal outcome of all MAS deployments, consistent with literature showing that the 'artificial reef effect' is influenced by structural, temporal, and ecological factors (Lemasson et al., 2024; Pardo et al., 2023; Taormina et al., 2022; Vivier et al., 2021). Abiotic factors affecting community composition around MAS include structural properties of size, shape, material, slope and degree of complexity, all contributing to the time needed for ecological succession to reach a climax state (Taormina et al., 2022). Analysis revealed positive diversity responses for oil and gas infrastructure (O&G), other energy infrastructure, coastal infrastructure, artificial reefs and add-on habitat units, aligning with previous work showing that the provision of hard substrates enhances settlement opportunities and provides refugia from

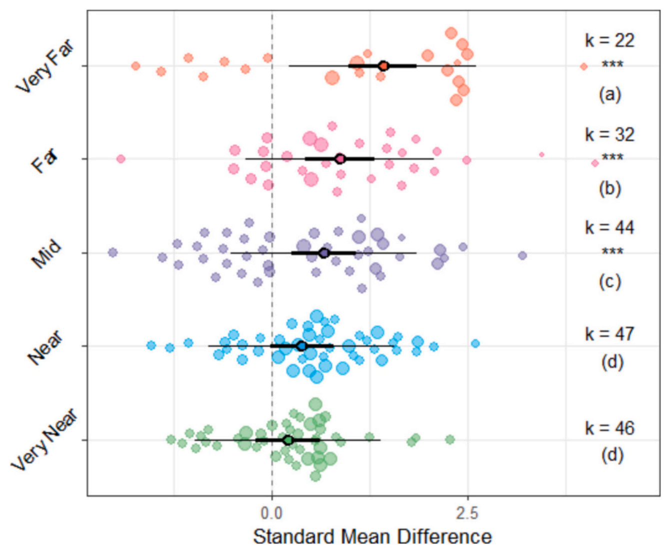

Fig. 6. Mean effect size (Standardised Mean Difference, SMD) for the impact of distance between control sites and marine artificial structures (MAS) on diversity differences. Distances were categorised as: Very Near (0–25 m), Near (26–50 m), Mid (51–100 m), Far (101–200 m), and Very Far (> 200 m). Each point represents an individual observation, with point size proportional to its precision (1/standard error, SE). Thick horizontal bars represent the 95% confidence intervals (CI) of each mean effect size, while thin error bars represent the 95% prediction interval. Effects are considered non-significant where the 95% CI cross the dashed zero line. Significance levels are indicated by asterisks: $p < 0.05$ (*), $p < 0.01$ (**), $p < 0.001$ (***). The number of observations is represented by k. Distance categories not sharing a letter differ significantly from one another based on pairwise tests (QM test, $p < 0.05$).

both abiotic and biotic stressors (Bishop et al., 2017), leading to increased taxonomic richness and biomass (Firth et al., 2016; Fortune et al., 2024; Perkol-Finkel et al., 2018). The only MAS category that did not significantly increase diversity was OWFs, the primary focus of this study, which showed weaker and less consistent evidence of diversity gains in comparison to other MAS, even with the addition of NID (Fig. 7). By comparing diversity responses, the findings provide new evidence to the debate on whether OWFs can act as effective artificial reefs and contribute to diversity relative to other MAS, as literature has shown highly variable and site-specific outcomes (Pardo et al., 2023).

When compared to soft-sediment environments, MAS overall supported significantly greater diversity, consistent with findings that artificial substrates promote communities otherwise absent from sandy or muddy bottoms through changes in the structure and functioning of the ecosystem with additional ecological benefits of increased settlement surfaces, increased food availability and refugia (Buyse et al., 2023; Firth et al., 2014; Fortune et al., 2024; Knorrn et al., 2024; Lemasson et al., 2024; Taormina et al., 2022). While these increases suggest enhanced local diversity, conclusions should be based on the intended restoration objective and target state of the environment, as they reflect a shift in habitat type and community composition rather than restoration of pre-existing conditions and may also include the establishment of non-native or opportunistic species (Bulleri and Chapman, 2010). Artificial reefs and O&G platforms in particular have repeatedly been shown to add to local diversity, increasing taxonomic richness and biomass through the artificial reef effect (Fortune et al., 2024; Methratta and Dardick, 2019; Perkol-Finkel et al., 2018; Taormina et al., 2022; Vivier et al., 2021). The stronger effects observed for O&G platforms may reflect a combination of structural characteristics, long-term stability, and environmental context, which influence habitat availability, refuge from predation, and colonisation opportunities. As artificial reefs are usually designed and deployed with aims to mimic natural reef functions, predominantly for ecosystem conservation/restoration or fisheries enhancement, their positive impact on diversity is expected (Lemasson et al., 2024; Taormina et al., 2022; Vivier et al., 2021).

While increases in diversity associated with MAS are often interpreted as ecological enhancement, it remains unclear whether these patterns reflect true increases in production or the attraction of individuals from surrounding habitats. This distinction is critical, as attraction represents a redistribution of existing biomass, whereas production implies an increase in regional diversity and net ecological gain (Methratta and Dardick, 2019; Knorrn et al., 2024; Stenberg et al., 2015). Evidence for OWFs remains inconclusive, with studies suggesting both aggregation and potential production mechanisms (Methratta and Dardick, 2019; Knorrn et al., 2024). To further investigate these processes, future studies should incorporate approaches beyond abundance metrics, including demographic analyses, recruitment studies, and trophic assessments (Bohnsack, 1989).

OWFs and coastal infrastructure, however, showed limited evidence of diversity enhancement. Historically, coastal defences have had low structural diversity, though when eco-engineering and NID are applied, their outcomes can shift positively (O'Shaughnessy et al., 2020). The lack of an increase in diversity seen with OWF deployments may result from turbine design and material type, where monopiles without scour protection provide smooth, vertical surfaces with limited microhabitats, resulting in fewer niches and less diverse and abundant associated communities (Pardo et al., 2023; ter Hofstede and van Koningsveld, 2024). This contradicts previous literature where the installation of OWFs has been expected to modify marine habitats and drive changes in local and regional diversity through the introduction of large-scale hard substrates (Causon and Gill, 2018). However, ecological effects of OWF differ depending on several factors, including sampling methodology, ecosystem type, and wind farm characteristics (Methratta and Dardick, 2019), where jacket structures offer much greater surface area than monopiles and therefore greater potential for diversity benefits (ICF, 2021).

Additionally, OWFs in this dataset were relatively young (<8 years old), falling within categories of 'Very New' to 'Mid', where study observations may have been done on communities in the early stages of succession. Studies focused on the colonisation of MAS have suggested that climax communities can take a decade or more to fully establish (Vivier et al., 2021), highlighting the importance of long-term monitoring (Pardo et al., 2023). When focusing on the age of all MAS, the oldest structures had the largest positive effect on diversity. The oldest MAS were mostly O&G and coastal infrastructure, where diversity benefits may have accumulated over time (Vivier et al., 2021), consistent with the conceptual successional trajectory shown in Fig. 7. However, community development is not solely time-dependent and is also shaped by ecological connectivity and environmental conditions, where dispersal processes, hydrodynamic regimes, and habitat context influence colonisation rates and community development (Coolen et al., 2020).

Not all NID interventions have consistent outcomes, and monitoring is often too short-term for benefits to be realised (Kingma et al., 2024; Pardo et al., 2023). The highest diversity gains were observed when NID was incorporated into coastal infrastructure, whereas coastal infrastructure overall showed no effect relative to soft sediments. This may be due to eco-engineering practices being more established for coastal infrastructure with an existing peer-reviewed design catalogue (O'Shaughnessy et al., 2020) and multiple studies comparing and synthesising common eco-engineering options (Evans et al., 2021; Strain et al., 2018). Add-ons and the most complex artificial reefs also significantly enhanced diversity when applied to MAS, demonstrating that purposeful structural modifications can lead to increases in diversity, as illustrated in Fig. 7, though their impacts are highly context dependent and vary across spatial scales (Firth et al., 2024). Their design strongly influences their ecological effectiveness, with material, shape, size, and surface heterogeneity influencing settlement rates and colonisation of

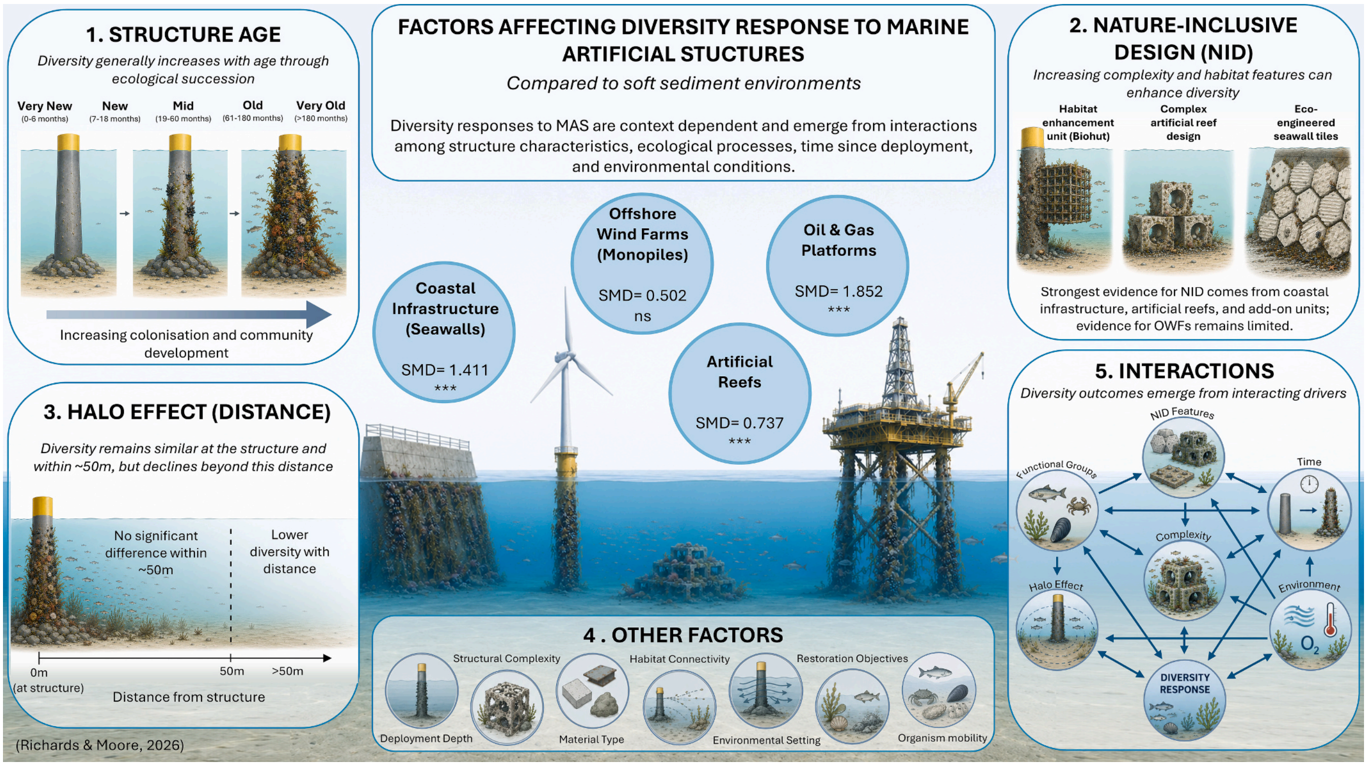

Fig. 7. Conceptual synthesis of the factors influencing diversity responses to marine artificial structures (MAS). The central illustration contrasts diversity responses among major MAS types, including offshore wind farms, oil and gas platforms, artificial reefs, and coastal infrastructure. Values are Standardised Mean Difference. Significance: *** = $p < 0.001$, ns = not significant. Surrounding panels summarise the principal factors investigated in this study: (1) ecological succession associated with increasing structure age; (2) nature-inclusive design (NID) interventions that increase structural complexity and habitat heterogeneity; (3) the halo effect, showing that diversity remained comparable between structures and sites within 50 m before declining with increasing distance; (4) additional factors influencing diversity responses, including ecological response metrics, organism mobility, and broader environmental processes discussed in the literature; and (5) the interaction of factors influencing diversity outcomes.

sessile organisms, as well as increasing refugia and reef holding capacity influencing mature fish (Firth et al., 2024; Vivier et al., 2021). With the addition of NID to OWFs, the effect on diversity remained insignificant. This likely reflects the limited scope of NID application in the included studies, which focused solely on increasing the complexity of scour protection. This is a common NID approach in OWFs, where an increase in available surface area has been shown to result in increased species richness (Kingma et al., 2024), however, the extent of impact depends on the relative increase in complexity compared to the original substrate (Bergström et al., 2014). In the study by Kingma et al. (2024), included in this meta-analysis, responses to scour protection treatments varied with material and the ecological metric measured, potentially influencing the non-significant effect of OWF NID in the current study. The limited availability of studies assessing NID in OWFs may also reflect a lack of standardised monitoring frameworks and reporting requirements, reducing the comparability and accessibility of data for scientific synthesis and limiting the ability to optimise design and implementation. As OWFs already provide substantial hard substrates, additional interventions may not affect sessile organisms, suggesting that NID efforts targeting mobile species may be more impactful.

When looking at the effect of MAS on different ecological metrics, abundance, richness, biomass, and Shannon-Weiner diversity all returned positive responses. However, only species richness and total abundance remained statistically significant when exploring the effects of different MAS on these response variables. This may be due to the methodologies of studies, where richness and abundance are the most frequently measured ecological response, or their strength as ecological metrics, which is due to their ability to reduce complex information into easily interpretable measures (Taormina et al., 2022). Standardised assessment protocols have not yet been defined to measure the ecological effects of MAS (Taormina et al., 2022), limiting comparability between studies.

The response of different functional groups varied when focussing on MAS collectively and OWFs only, where both sessile and mobile species responded positively to all MAS, and only sessile species had a positive response to OWFs. The stronger response of mobile species for all MAS likely reflects the diversity of MAS types assessed, with many providing heterogeneity and refuge attractive to fish and crustaceans, as well as increased settlement surfaces (Taormina et al., 2022). Differences in responses can also be driven by MAS material type and the time since deployment, with sessile invertebrates often showing stronger or contrasting responses compared to mobile invertebrates (Grasselli et al., 2024).

In contrast, the lack of a response of mobile species to OWFs likely reflects both methodological and ecological factors. The low sample size used in this study, combined with high variability in fish distribution and movement, may limit the reliability of monitoring and the robustness of observed patterns (Methratta and Dardick, 2019). Ecologically, the limited structural heterogeneity of turbines, particularly smooth monopile surfaces, may provide settlement substrate for sessile organisms but offer limited refuge or foraging habitat for mobile species. This, together with behavioural responses, predation risk, and broader spatial dynamics, may reduce the association of mobile species with OWFs, particularly in younger systems where communities are still developing (Methratta and Dardick, 2019).

An additional methodological finding influencing the response of diversity to MAS was the distance of control sites relative to the MAS, revealing a halo effect of extended ecological influence into surrounding soft habitats (Pardo et al., 2023; Reeds et al., 2018). Controls within 50 m ('Near' and 'Very Near' categories) showed no significant difference from MAS sites, with differences becoming larger and more pronounced as distance increased (Fig. 7). For OWFs, this reef effect has been observed to extend into surrounding sandy habitats between turbines, described as a spillover effect (Knorrn et al., 2024), with a highly localised effect particularly for fish abundance (Methratta and Dardick, 2019), benefitting surrounding fisheries (Causon and Gill, 2018; Werner et al., 2024), though this contradicts results from this review for mobile species. Nevertheless, these findings reinforce that MAS influence diversity beyond their physical footprint (Knorrn et al., 2024), with the spatial extent of these effects also shaped by local hydrodynamic and biogeochemical processes, where turbulence and changes in water flow can influence nutrient distribution, primary productivity, and the surrounding community structure (Daewel et al., 2022; Christiansen et al., 2022). As control sites placed within 50 m of MAS may underestimate ecological effects due to localised halo influences, it is recommended that future studies should position control sites at distances greater than 50 m to ensure independence and improve comparability across studies.

Overall, while MAS can enhance diversity, outcomes should be interpreted relative to environmental conditions, and ecological objectives (Coolen et al., 2020; Dannheim et al., 2020; Daewel et al., 2022). Observed increases in diversity do not necessarily translate to positive ecological change, as shifts in habitat type and community composition may include non-native species colonisation, functional homogenisation, or shifts in trophic structure (Firth et al., 2016; Bulleri and Chapman, 2010; Lemasson et al., 2024). Assessing functional diversity and ecosystem functioning alongside species richness is therefore critical for evaluating true ecological benefits (Bulleri and Chapman, 2010; Firth et al., 2016).

While MAS generally supported higher diversity, key knowledge gaps remain for OWFs, particularly on the influence of structural complexity, material type, age, and the integration of NID beyond scour protection. Although the concept of OWF NID is growing through pilot projects trialling add-ons (artificial habitat units), these are in the early stages of development (Pardo et al., 2023), and robust, long-term evidence across appropriate timescales is lacking. As a result, the effectiveness of NID in OWFs remains uncertain, with current conclusions largely based on indirect evidence from other MAS types.

Non-tested NID may lead to uncertainties in design processes, technical and ecological risks, and may also increase project costs (Pardo et al., 2023). Ensuring studies use appropriate control sites will be crucial to provide reliable outcomes to inform future developments, as pilot or observational studies that are largely descriptive without suitable comparators remain, limiting the ability to assess true ecological effects (Lemasson et al., 2024). Furthermore, variation in monitoring frequency, methodology, and reporting reduces comparability across studies, and the absence of internationally standardised metrics and shared databases limits the ability to synthesise and compare ecological outcomes across MAS (Dannheim et al., 2025). This highlights the need for stronger guidance, standardised protocols, and coordinated data sharing frameworks to improve consistency, accessibility, and the translation of evidence into policy and practice (Dannheim et al., 2025). Confidence and awareness in the evidence base therefore remains low, particularly in the UK where stakeholders have identified a lack of reliable data as a key barrier to NID implementation (Evans et al., 2019). Policy and industry uptake of NID, particularly in the UK, remains constrained by uncertainties, technical risks, and regulatory gaps (Hermans et al., 2020; Pardo et al., 2023; ter Hofstede and van Koningsveld, 2024).

Lessons from eco-engineered coastal infrastructure provide valuable insights for evaluating, adapting and applying ecological design into OWF development (Pardo et al., 2023). By manipulating structural components of NID, such as structural complexity, materials and microhabitat features, designs could more efficiently target desired functional groups and ecosystem services (Kingma et al., 2024; Pardo et al., 2023). However, the success of these approaches will depend on site-specific conditions, including connectivity, environmental energy, and ecological context, reinforcing the need for adaptive, location-specific design strategies (Firth et al., 2016; Pardo et al., 2023; Dannheim et al., 2020).

Moving forward, long-term, standardised monitoring across ecological metrics, functional groups, spatial scales, and control distances larger than 50 m is needed to account for successional dynamics and halo effects (Reeds et al., 2018; Taormina et al., 2022), which is particularly important for OWFs as they continue to expand and dominate increasing areas of marine space (Kingma et al., 2024; Watson et al., 2024).

Together, these findings indicate that while MAS offer clear potential to enhance marine diversity, achieving meaningful ecological benefits in OWFs will require a shift from incidental habitat provision to intentional, evidence-based, and context-specific design guided by clearly defined ecological objectives (Firth et al., 2016; Pardo et al., 2023; Kingma et al., 2024).

Declaration of generative AI and AI-assisted technologies

During the preparation of this work the authors used ChatGPT Go in order to generate illustrations for Fig. 7. After using this tool, the authors reviewed and edited the content as needed and take full responsibility for the content of the published article.

CRediT authorship contribution statement

Katherine Richards: Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Pippa J. Moore:** Writing – review & editing, Supervision, Project administration, Funding acquisition, Conceptualization.

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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.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoleng.2026.108103.

Data availability

Data will be made available on request.

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