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Safe harbours for the intruders of the sea: Greater abundance of invasive species on artificial structures and management implications

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ABSTRACT

Artificial structures, both coastal (e.g. piers, seawalls) and offshore (e.g. platforms), have been hypothesised to support greater abundance and diversity of invasive species, including introduced species that have established and spread or have the potential to do so, compared with natural reefs. Despite this widespread assumption, no systematic evaluation has been conducted, leaving a critical knowledge gap regarding the ecological consequences of expanding artificial infrastructure. To address this gap, a systematic review and meta-analysis – comparing abundance, richness and cover of invasive species between artificial structures and natural reefs – were conducted. The influence of moderators on effect sizes – including distance to ports, shores and natural reefs, artificial structure category, size, age, material and invasive species traits – was also assessed. Artificial structures, on average, harboured more abundant – but not more diverse – populations of invasive species. Structural and spatial variables further influenced outcomes, with effect sizes generally small despite being statistically significant. Artificial structures located farther from ports and shore exhibited greater differences from natural reefs, and those constructed from concrete or metal supported higher invasive species abundance. Contributing factors likely include proximity to invasion points (e.g. harbours), structural characteristics of the infrastructure, and competitive traits of invasive species. Given the rapid global expansion of artificial structures, particularly in coastal environments, risk-reduction strategies are essential. This study provides management-relevant recommendations to support risk-aware planning, ecological design and adaptive management of marine artificial infrastructure, thereby mitigating invasion risk and guiding sustainable policy and practice.

1. Introduction

Coastal population growth and intensifying urbanisation are driving rapid expansion of hard artificial structures (AS) in the marine environment (Komyakova et al., 2022a; Reimann et al., 2023; Bugnot et al., 2021). These deployments aim to support human dependence on marine ecosystem services, including energy production, fisheries, transport and coastal protection. AS come in a range of guises such as marinas, breakwaters, seawalls, pontoons and pilings, as well as accidental or purposely deployed artificial reefs (Firth et al., 2016; Merkens et al., 2016; Bugnot et al., 2021). With a growing human population, marine infrastructure is expected to expand through the construction of artificial islands and the development of offshore energy production and aquaculture, including floating installations, as well as the expansion of ports and marinas to support transport needs (Bugnot et al., 2021; Luo and Komyakova, 2025). At the same time, intensifying climate-change

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pressures are likely to drive further coastal hardening to meet increasing demands for coastal protection (e.g. breakwater walls) (Bugnot et al., 2021). The most recent estimates suggest that, although the physical footprint of artificial marine structures will grow by approximately 23% from 2018 to 2028, the surrounding seascape modified by these installations is expected to expand far more extensively, increasing by an estimated 50–70% over the same period (Bugnot et al., 2021). Many of these structures are not designed to serve as marine habitats, they introduce hard substrate into pelagic and coastal environments that facilitates the settlement of marine communities (O'Shaughnessy et al., 2023). However, artificial habitats might not replicate natural-habitat functions and therefore could host distinct communities that differ from those in natural systems – often described as '*novel ecosystems*' (Hobbs et al., 2013; O'Shaughnessy et al., 2023).

Although AS can support species diversity and biomass comparable to local natural reefs (NRs) (Lemasson et al., 2024; Paxton et al., 2020), they can also cause numerous environmental impacts. These include pollution from toxic leachates, deterioration of plastic components or structural breakdown (e.g. tyre reefs), habitat alteration and loss (e.g. replacement of natural habitats – such as seagrass beds, kelp forests and soft sediments – with AS) (Komyakova et al., 2022a; Heery et al., 2017), and the formation of ecological traps where organisms experience poor fitness (Hale and Swearer, 2016; Komyakova et al., 2021; Swearer et al., 2021). A key characteristic of novel ecosystems is the proliferation of invasive species (Airoldi et al., 2015; Bulleri and Chapman, 2010; Hilliam et al., 2024; Mineur et al., 2012). Invasive species are a primary driver of global biodiversity loss, accounting for the second highest rate of native species extinction (Bellard et al., 2016). Invasive species can disrupt ecosystems by altering community structure and functioning, with significant consequences for ecosystem services (Groeneveld et al., 2018; Katsanevakis et al., 2014). Globally, marine invasive species are estimated to cause economic losses of US$3.6 billion, impacting local industries, people's income and job security (Cuthbert et al., 2021; Groeneveld et al., 2018).

Marine areas are among the most heavily invaded ecosystems (Grosholz, 2002) with reports of invasives species in 84% of marine ecoregions (Molnar et al., 2008). As a result, the environmental and economic costs of marine invasions are likely to be severely underestimated (Cuthbert et al., 2021). Given that predictions suggest a 20-fold increase in shipping traffic by 2050, we are likely to see a 3-fold increase in marine invasion rates (Sardain et al., 2019). The future increase in invasion rates and subsequent ecological and socio-economic impacts of invasive species highlight the need for improved management strategies and further research into mechanisms that facilitate invasions, including habitat preferences for AS (Giakoumi et al., 2019; Pyšek and Richardson, 2010; Simberloff et al., 2013).

AS have been suggested as potential hubs for marine invasions, providing footholds for invasive species – unlike NRs, which may be more resilient (Airoldi et al., 2015; Komyakova et al., 2022a). This assertion has been based on structural differences between natural and artificial habitats. AS are often less structurally complex and constructed from non-natural materials (e.g. concrete, plastic), features which are commonly considered to favour invasive species. AS are also commonly located in areas of elevated pollution that can weaken the resilience of native communities, deployed as bare substrates that provide competitor-free space for colonisation, and situated near major vectors of invasive species transfer (Komyakova et al., 2022a). However, the evidence to confirm this assertion has not been systematically evaluated, and understanding its validity is crucial for informing conservation strategies and mitigating ecological risks. To address this gap, this review has three key objectives. First, we systematically evaluate existing research to determine whether AS support greater numbers or diversity of invasive species compared to NRs. Second, we explore a range of explanatory factors that may influence relative abundance of invasive species. Third, employing a narrative literature review approach, we provide evidence-based recommendations for environmental managers on mitigating the risks of artificial infrastructure acting as invasion hubs for marine invasive species.

2. Methods

2.1. Terminology

Non-native or introduced species are commonly defined as species that occur outside their native range as a result of human activity, whether intentional (e.g. deliberate introductions) or unintentional, via human-mediated pathways such as ballast water, ship hull fouling or transport of goods (Colautti and MacIsaac, 2004; Richardson et al., 2000; Blackburn et al., 2011; Soto et al., 2024). Some introduced species are further classified as invasive, generally based on their ability to establish, spread and increase in abundance, resulting in significant negative impacts on recipient ecosystems and native organisms (Wright et al., 2023; Clinton, 1999; Ricciardi, 2013; Soto et al., 2024). Not all introduced species become invasive (Blackburn et al., 2011), and some species may be invasive in certain regions but not in others. The establishment, spread and impacts of introduced species are strongly influenced by local environmental conditions, biotic interactions and the resilience of the native community (Davis et al., 2001).

There is limited consensus within invasion ecology on a standardised definition of invasive species (Colautti and MacIsaac, 2004; Blackburn et al., 2011; Soto et al., 2024). Colautti and MacIsaac (2004) proposed describing invasion in terms of population stages, focusing on biogeography rather than taxonomy, whereby a species may be 1) locally restricted and rare (non-invasive stage), 2) widespread but occurring at low abundance (invasive stage), 3) locally restricted but numerically dominant (non-invasive stage), or 4) both widespread and numerically dominant (invasive stage). However, their framework did not explicitly incorporate the magnitude of negative impacts, which is commonly considered a defining characteristic of invasive species, especially in policy and management documents (Clinton, 1999; Ricciardi, 2013). Since Colautti and MacIsaac (2004), expanded biogeography-based (e.g. Blackburn et al., 2011) and impact-based (e.g. Jeschke et al., 2014) frameworks have been proposed. Some frameworks suggest that the definition of invasion should be decoupled from impact, relying more on spread and establishment (e.g. Colautti and MacIsaac, 2004; Blackburn et al., 2011). Others demonstrate that even introduced species, not formally classified as invasive may cause environmental impacts (e.g. Jeschke et al., 2014). Complexity is further compounded by invasion lag times, whereby introduced species may only become invasive many years after introduction (Crooks, 2005; Essl et al., 2011; Ricciardi, 2013).

For the purposes of this study no attempt is made to redefine invasion ecology or evaluate the appropriateness of existing terminology. Instead, a general definition is adopted whereby invasive species are considered introduced species that have established and spread within a new ecosystem, resulting in environmental impacts. Given the study focus on potential facilitation of invasive species by artificial marine infrastructure a precautionary simplification is applied. Accordingly, introduced species are assumed to have the potential to become invasive under suitable conditions (Colautti et al., 2014; Mooney and Cleland, 2001). Therefore, all relevant taxa are hereafter referred to as "invasive species". Notably, most reviewed studies defined their study species as invasives.

2.2. Systematic literature review

A systematic literature review was conducted to quantify differences in invasive and/or introduced species presence (i.e. abundance, cover, richness, diversity) between AS and adjacent NRs. Searches were performed using Scopus on 2 June 2025 and Web of Science on 24 June 2025. Search terms included those related to invasive and/or introduced species defined by Wright et al. (2023), and to artificial reefs, NRs and marine environments (Table 1).

Additional terms were identified and incorporated using *litsearchr*, a tool that facilitates the refinement of search strategies through keyword co-occurrence networks (Grames et al., 2019). The initial searches were limited to the titles and abstracts of peer-reviewed articles published in English. The Web of Science search was conducted using the "All Databases" setting to ensure comprehensive coverage. In Scopus, the search was further restricted by subject area, with the following fields excluded: Social Sciences; Computer Science; Physics and Astronomy; Neuroscience; Medicine; Immunology and Microbiology; Mathematics; Chemical Engineering; Chemistry; Business, Management and Accounting; Decision Sciences; and Pharmacology, Toxicology and Pharmaceutics. Due to differences in subject classification between Scopus and Web of Science, these exclusions were not applied to the Web of Science search.

The initial search criteria returned 446 publications from Web of Science and 420 from Scopus. These were imported into Covidence (https://www.covidence.org) – a systematic review management software – and duplicate entries removed. After removing 327 duplicates, 539 unique articles remained. The relevance of the remaining publications was assessed through title and abstract screening by two observers. Publications were excluded if they did not focus on invasive, introduced, or non-native species occurring on AS and adjacent NRs. Studies were included if they provided a comparative analysis using quantitative measures such as abundance, cover, species richness, or diversity. The first shortlist included 44 studies that met the preliminary inclusion criteria (Fig. 1).

The systematic searches were supplemented with two additional approaches – targeted searches in Google Scholar and a snowballing technique – to further identify relevant articles and grey literature. Similar keywords to those listed in Table 1 were used in the Google Scholar search; however, due to limitations of the platform, the number and order of search terms had to be varied manually. An additional 112 articles from the Google Scholar search were screened. The snowballing technique involved screening the reference lists of studies shortlisted through the systematic literature review, resulting in 67 additional articles identified for abstract evaluation. The snowball search was expanded to include publications written in Spanish, German and French. One additional study written in Spanish was short-listed (Vega et al., 2015). After reviewing these articles, an additional 16 publications were selected for full-text examination (Fig. 1).

Table 1

Terms used in the systematic literature review on non-native species on artificial structures and natural reefs.

Topic	Search terms
Invasive	alien OR "alien speci*" OR "biolog* invas*" OR colonis* OR establish* OR introduc* OR "introduc* speci*" OR invas* OR "invas* alien" OR "invas* impact*" OR "invas* speci*" OR "non-indigen* speci*" OR "non-native speci*" OR "speci* invasive"
Artificial structure	"artificial habitat*" OR "artificial installation*" OR "artificial reef*" OR "artificial structure*" OR "artificial substrate*" OR Biohut OR breakwater* OR "bridge piling*" OR causeway* OR "constructed reef*" OR "designed habitat*" OR dock* OR "eco-block*" OR "eco-engineered structure*" OR "engineered reef*" OR groyne* OR "habitat enhancement structure*" OR "hard substrata*" OR HMR OR jetty* OR "man-made reef*" OR "human-made reef*" OR "man-made structure*" OR "human-made structure*" OR "marine infrastructure" OR marina* OR "offshore infrastructure*" OR "oil rig*" OR pier* OR "purpose-built reef*" OR pipeline* OR platform* OR "purpose built reef*" OR reefball* OR "Reef ball*" OR seawall* OR shipwreck* OR subsea OR "submerged structure*" OR "sunken structure*" OR wreck*
Marine	marine OR ocean OR reef OR sea OR seascape
Natural reef	bedrock* OR "boulder field*" OR "coral reef*" OR "hard bottom*" OR "hard-bottom*" OR hardbottom* OR "fringing reef*" OR "kelp forest*" OR "natural habitat*" OR "natural reef*" OR outcrop* OR "oyster reef*" OR "reef flat*" OR "rock reef*" OR "rocky reef*" OR "shelf reef*" OR "shore boulder*" OR "temperate reef*" OR "worm reef"

* Wildcard for any group of characters or no character.

Stringent selection criteria were applied during the full-text screening of the initial shortlists. The criteria were as follows: (1) Articles contain data on the presence of non-native species, expressed as a measure of abundance (e.g. counts, cover) or diversity (e.g. richness, diversity indices); (2) Studies compared non-native species presence on AS and adjacent NRs, including any hard habitat such as rocky, coral or oyster reefs; (3) Studies examined non-native species presence on the AS and NRs themselves, rather than on experimentally deployed settlement plates, which are not representative of stable, long-term established communities, only reflect early settlement patterns, and are unlikely to mimic the material and structural composition of AS and NRs; (4) The same sampling methods (e.g. quadrats, transects) were used on both the AS and NR within the same study to avoid sampling bias (rapid assessment methods were excluded because they do not rely on standardised sampling techniques and do not provide reliable population metric estimates); (5) The adjacent or paired AS and NR were located no more than 20 km apart (Figs. A1–A4), a pragmatic upper limit chosen to minimise confounding from broad environmental and hydrodynamic gradients while retaining sufficient studies for synthesis; and (6) comparative sampling of the adjacent or paired AS and NR occurred within three months.

Based on the criteria, 26 publications were selected for further analysis (Fig. 1). Of these, 20 provided mean and variance data: 15 reported abundance, 3 reported richness, and 2 reported both. To ensure accuracy in data extraction, quality assurance and quality control (QA/QC) was conducted by a second reviewer, who cross-checked all extracted data from 20% of the papers. Any complex articles that raised questions regarding their suitability were discussed by the full team of co-authors.

2.3. Data extraction

Extracted data on invasive species included means, totals and measures of variance (standard deviation, SD; standard error, SE; or confidence interval, CI) for both natural and artificial habitats. Where necessary, data were extracted from plots using WebPlotDigitizer 5.2 (Rohatgi, 2024). Metadata of key descriptive variables were extracted, including the number of replicates, study duration, characteristics of AS and NRs (e.g. structure size, material and age), study locations, sampling methods, distance to shore, distance to nearest pier, and taxonomic information (Table A1). Descriptor variables were selected based on previous research identifying factors that may impact artificial habitat performance and invasive species establishment (Glasby et al., 2007; Komyakova et al., 2022a; Paxton et al., 2020; Sheehy and Vik, 2010; Vivier et al., 2021).

Many studies did not provide all variables of interest, so authors were contacted to request missing data. Targeted internet searches and Google Earth Pro were then used to fill as many remaining gaps as possible. Distances to NRs were calculated using a coordinate distance calculator. When a study pooled data across multiple AS and NR, the pooled estimate was treated as a single AS–NR comparison. Corresponding explanatory variables (e.g. AS site age and size) were calculated as the mean of the values across the contributing habitat pairs. If distances were not provided for pooled data, the minimum distance between multiple pairs of natural and artificial reefs was measured using the coordinate distance calculator, allowing calculation of a mean minimum distance for that comparison. In two studies, multiple effect sizes were extracted for the same AS–NR pair across different sampling times. Non-independence among these repeated measures was explicitly accounted for by including site identity as a random effect in the meta-analyses.

AS were categorised into five broad groups based on their intended

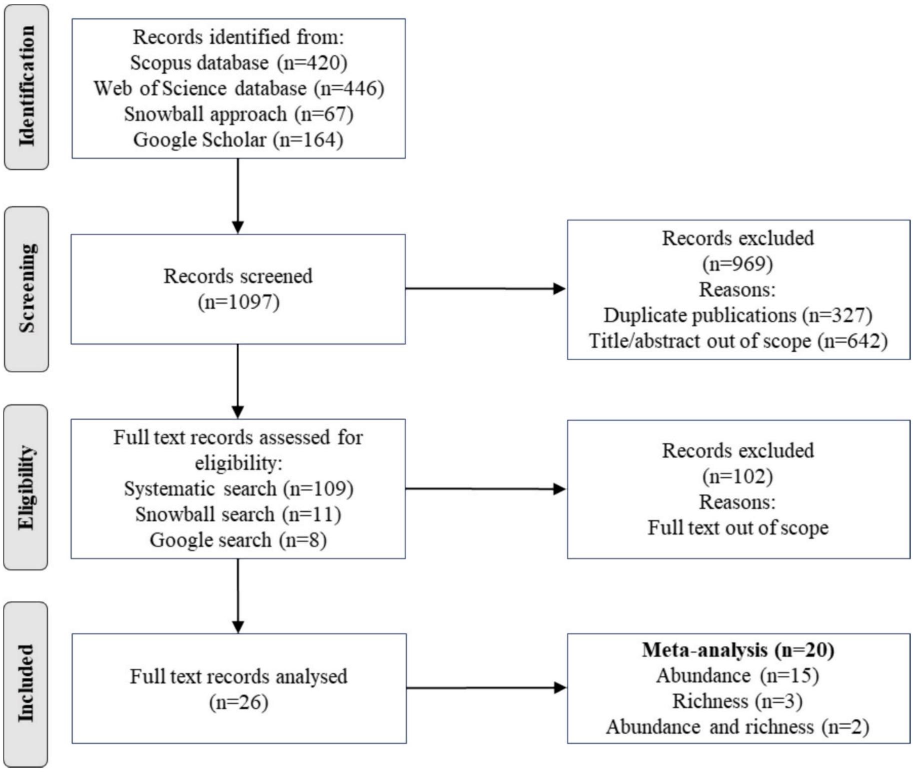


Fig. 1. Search flow of publication records – on the differences in the abundance and richness of invasive species between artificial and natural habitats – included in the systematic and snowball literature review. Based on PRISMA flow (Moher et al., 2009).

function: shoreline structures, floating structures, artificial reefs, oil and gas platforms, and derelict constructions. Shoreline hardening structures, made from hard and immovable materials, comprise dykes, breakwaters, revetments, marinas, jetties, seawalls, pontoons and pilings. Floating structures, not fixed to the seafloor but anchored via flexible connections (e.g. ropes, chains, lines), are typically fibreglass pontoons and floating docks. Artificial reefs included Reef Balls®, pyramids, tetrahedrons and cinder blocks. Derelict constructions encompassed remains of abandoned buildings originally installed for non-ecological purposes (e.g. military or industrial) that are no longer in use but remain in place.

The taxonomy of invasive species was recorded to species level where possible. Invasive species were categorised into major taxa: algae, Annelida (e.g. polychaetes), Bryozoa, Cnidaria (e.g. anemones and corals), Crustacea (e.g. amphipods and barnacles), Mollusca (e.g. bivalves and gastropods), Tunicata (e.g. ascidians), Echinodermata and fish. Species were also classified by feeding guild (filter feeders vs non-filter feeders) and movement habit (sessile vs mobile) to assess whether functional roles influenced their relative prevalence on AS.

2.4. Data analysis

The 20 publications that provided both mean and variance data – were included in the weighted meta-analysis, while the remaining 6 articles were used as additional information for management and recommendations for AS deployment (Table A2). Effect sizes were calculated for each comparison between AS and NRs. To calculate the effect sizes (i.e. the difference in invasive species abundance or richness between AS and NRs), the SD, sample size (n), and mean values (x) for each comparison were used to calculate Hedges' g (β). For cases where SD was not reported, conversions were applied. If the SE was provided, SD was calculated using the following formula:

$$SD = SE\sqrt{n}$$

When 95% CIs were reported, SD was calculated as:

$$SD = \frac{upperCI - lowerCI}{2 \times Z_{\alpha/2}}$$

Where $Z_{\alpha/2}$ equals 1.96 for a 95% CI.

Statistical analyses were performed in R version 2023.03.0 (R Core Team, 2022) using the *metafor* package (Viechtbauer, 2010) and *ggplot2* (Wickham, 2016) for data visualisation. Effect sizes were calculated using Hedges' g (β) with the rma function. A random-effects meta-analysis was conducted, with a site identifier as a random factor to account for multiple data points from the same site. Positive values of Hedges' g indicate higher invasive species abundance/richness on AS compared to NRs, whereas a value of zero indicate no difference.

A base model without moderators assessed overall differences in abundance and richness between habitats. Subsequently, the influence of covariates (distance to pier, distance to shore, distance to NR, AS size and AS age) on effect sizes was evaluated using a mixed-effects meta-regression model, with each covariate tested individually as a moderator. Because of uneven data gaps across covariates, multiple covariates could not be included in the same model. Positive moderator estimates indicate that an increase in the covariate is associated with a higher relative abundance of invasive species at AS compared to NRs. Statistical significance was inferred when $p < 0.05$ and when the 95% CI did not overlap with zero, and R^2 was used to estimate the proportion of heterogeneity explained by moderators.

3. Results

3.1. Bibliographic results

Although only 20 publications provided data suitable for the inclusion in the meta-analysis (Table A2), 26 have reported on the comparisons of invasive species abundance or diversity on AS vs NR

(Tables A2–A3). These studies are summarised subsequently. Studies evaluating abundance or diversity of invasive species on AS and NR were predominantly conducted in tropical, sub-tropical and temperate regions, including: the Aegean Sea, Greece (1 study); the Caribbean Sea, Cuba (1); the English Channel, England (3) and France (1); the Gulf of Mexico, Mexico (3); the Mediterranean Sea, Israel (2) and Spain (1); North Atlantic, USA (r1); northern Adriatic Sea, Italy (1); Santa Barbara Channel, California (2); South Atlantic Sea, Brazil (3); southeastern Vancouver, Canada (1); southern Baltic Sea, Poland (1); and Sydney Harbour (3) and Tasmania (1) in Australia (Fig. 2).

These studies reported data from 100 unique AS. The most common AS type was shoreline structures (65), which includes seawalls and wooden pilings, followed by floating structures (15) such as pontoons, then habitat enhancers (14) such as reef balls and tetrahedra. Less frequently reported structures included derelict constructions (3) such as concrete remains of offshore structures, and platforms (3) (Fig. 2). AS construction materials comprised concrete (31), fibreglass (3), metal (5), plastic (9), rock (17), wood (14) and mixed materials (21).

Of the 498 total comparisons extracted from 26 publications, 479 involved comparisons of AS to rocky reefs, 11 to coral reefs, and 8 to oyster reefs (Fig. 3). Of the 498 data points, 177 had means and variance and were thus included in the formal meta-analysis: 166 with rocky reefs and 11 with coral reefs. The remaining 321 comparisons did not report sufficient variance information for effect-size calculation and were therefore excluded from the formal meta-analysis. To avoid discarding relevant evidence, these comparisons were retained in our descriptive synthesis (Section 3.1).

3.1.1. *Monitoring program*

Nineteen studies monitored for less than one year, four between one and three years, and three exceeded three years (n = 26, mean ± SD = 1.59 ± 3.75 years). The most common survey methods were photography (8 studies) and underwater visual census (7 studies). Direct sampling techniques were used in five studies, involving scraping, airlifts, hand collection, or destructive sampling (complete removal), to quantify invasive species richness and abundance. Remotely operated vehicles (ROVs) were used in two studies to estimate invasive species density. In addition, five studies used a combination of methods, including photography, underwater visual census and video. Information regarding cleaning practices for AS was rarely reported, with details available in only two studies (Dafforn et al., 2012; Lemm, 2023).

3.1.2. *Targeted taxa*

The most common taxonomic groups monitored were molluscs (10 studies), Tunicata (9), Bryozoa (9), algae (7 studies), and Crustacea (7). Other groups included Annelida (6), fish (5), Cnidaria (4), and Echinodermata (1) (Tables A2–A3).

3.2. *Meta-analysis*

3.2.1. *Invasive species abundance and richness between artificial structures and natural reefs*

The abundance of invasive species was modestly, but significantly, higher on AS than on NR (Table 2, Fig. 4; $k = 129$, $\beta = 0.24$, $P < 0.0001$). The richness of invasive species communities did not differ significantly between AS and NR (Table 2, Fig. 4; $k = 48$, $\beta = 0.092$, $P = 0.1019$), although the low sample size (n = 5 studies) reduced statistical power.

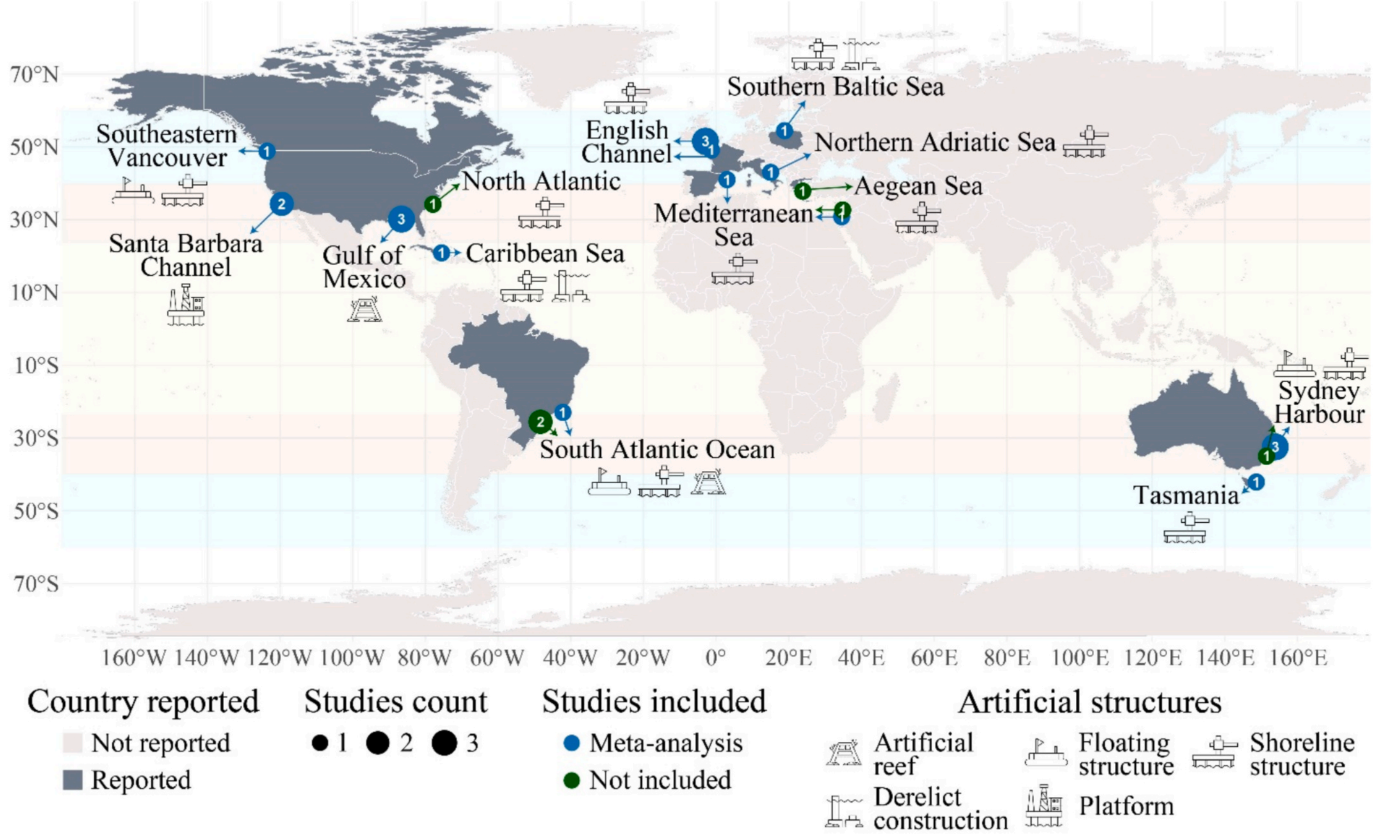


Fig. 2. General locations of artificial structures where studies reported differences in abundance and richness of invasive species between artificial structures and natural reefs. The map delineates temperate (light turquoise band: 40°–60° latitude), sub-tropical (light beige band: 23.5°–40° latitude) and tropical (light yellow band: 0°–23.5° latitude) zones. Countries with relevant studies are highlighted in blue-grey, with the size of the blue circles representing the number of studies in each area. The colours of the circles represent whether the study was included in the meta-analysis. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

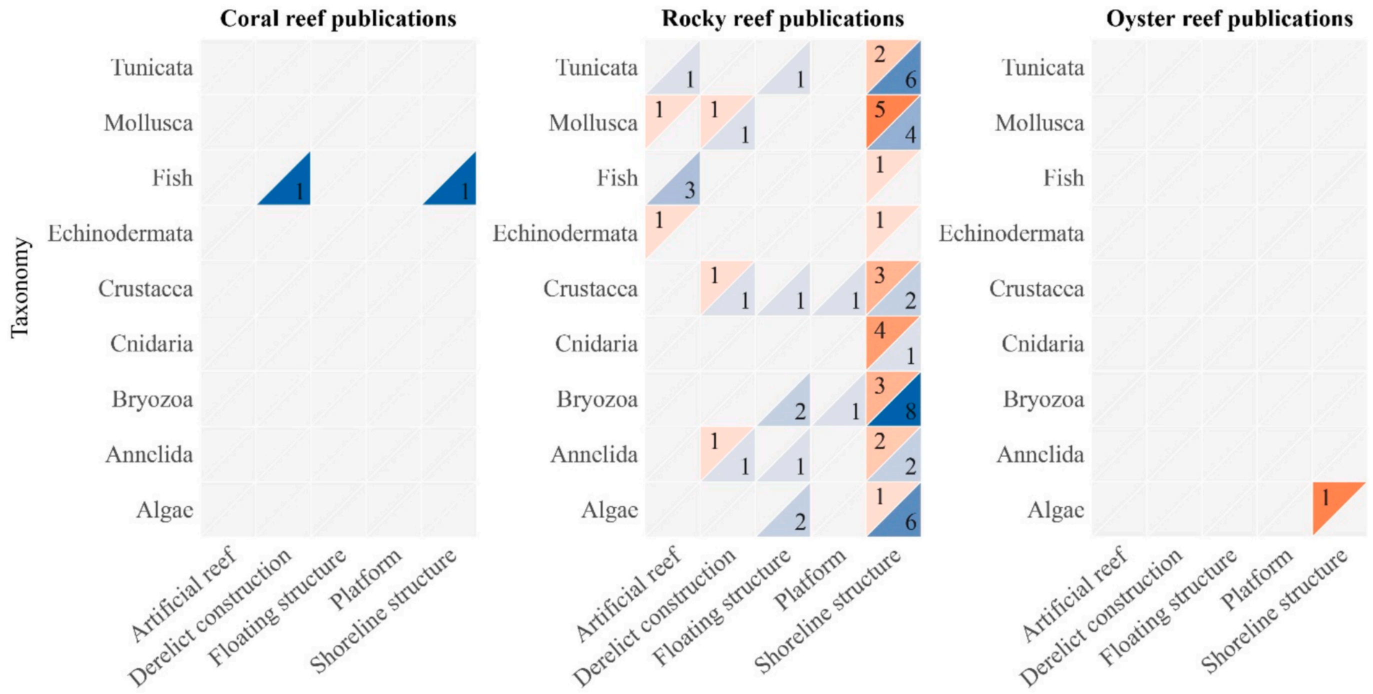


Fig. 3. Number of publications comparing invasive species abundance and richness on artificial structures and natural reefs (rocky, coral and oyster reefs). Blue triangles (lower triangle) represent publications included in the meta-analysis. Orange triangles (upper triangle) represent publications not included in the meta-analysis due to data limitations. Some publications reported multiple artificial structure types. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 2
Meta-analysis of effect sizes (Hedges' g) between artificial structures and natural reefs on invasive species abundance and richness, and the effect of covariates on the abundance differences between these habitats.

Outcome	k	Effect size β	SE	95% CI Lower	Upper	p-value	Heterogeneity R^2%
Difference between AS and NRs							
Abundance	129	0.2453	0.0391	0.1686	0.3220	**<0.0001**	
Richness	48	0.0916	0.0560	−0.0182	0.2014	0.1019	
Covariates (abundance only)							
Distance to NR	129	0.0136	0.0135	−0.0129	0.0400	0.3139	14.5
Distance to port	105	0.0189	0.0081	0.0030	0.0348	**0.0198**	49.2
Distance to shore	111	0.0284	0.0139	0.0012	0.0556	**0.0410**	24.9
Size	41	−0.0000	0.0000	−0.0000	0.0000	0.1557	0
Age	24	−0.0179	0.0149	−0.0470	0.0013	0.2306	89.68

Bold p-value indicates statistical significance.

3.2.2. *Evaluation of the effect of covariates on the difference between AS and NRs*

Covariate effects were assessed only for studies reporting abundance because richness data were too limited. The relative difference in abundance between AS and NRs was not influenced by the distance between them but was affected by distance to port and distance to shore (Table 2, Fig. 5). Both covariates showed weak positive effects, with differences between AS and NRs slightly greater when AS were located farther from ports and the shore (Table 2, Fig. 5). AS age and size were not significant moderators (Table 2, Fig. 5), although data for these variables were scarce.

All AS types exhibited significantly positive estimates (indicating higher abundance of invasives compared to NR), except for derelict constructions, which harboured lower abundance of invasives than NRs (Table 3, Fig. 6). AS made of concrete and metal were associated with significantly higher invasive species abundance on AS compared to NRs, while all other materials showed no significant effects, noting that replication for some materials was comparatively low (Table 3, Fig. 6). The greater invasive species abundance on AS compared to NR was true for both sessile and mobile species, and for filter feeders and non-filter feeders (Table 3, Fig. 6).

4. Discussion

Invasive species pose a significant risk to marine ecosystems worldwide, often restructuring communities and altering ecosystem functions, which makes understanding the contexts that facilitate their establishment critical. A key factor contributing to the establishment success and dominance of invasive species over local populations is their adaptability to diverse environmental conditions, supported by biological traits such as rapid sexual maturity, high growth rates, high fecundity, extended breeding seasons and short larval stages (Epelbaum et al., 2009; Lambert and Lambert, 1998; Li et al., 2023; Paul, 1937) (Fig. 7). Invasive species can generally tolerate a wide range of conditions, including pollution (Blanco et al., 2021; Kocak et al., 1999), high temperatures (Tepolt and Somero, 2014; Zerebecki and Sorte, 2011) and a broad range of salinities (Jud et al., 2015; Kenworthy et al., 2018a; Marchessaux et al., 2024).

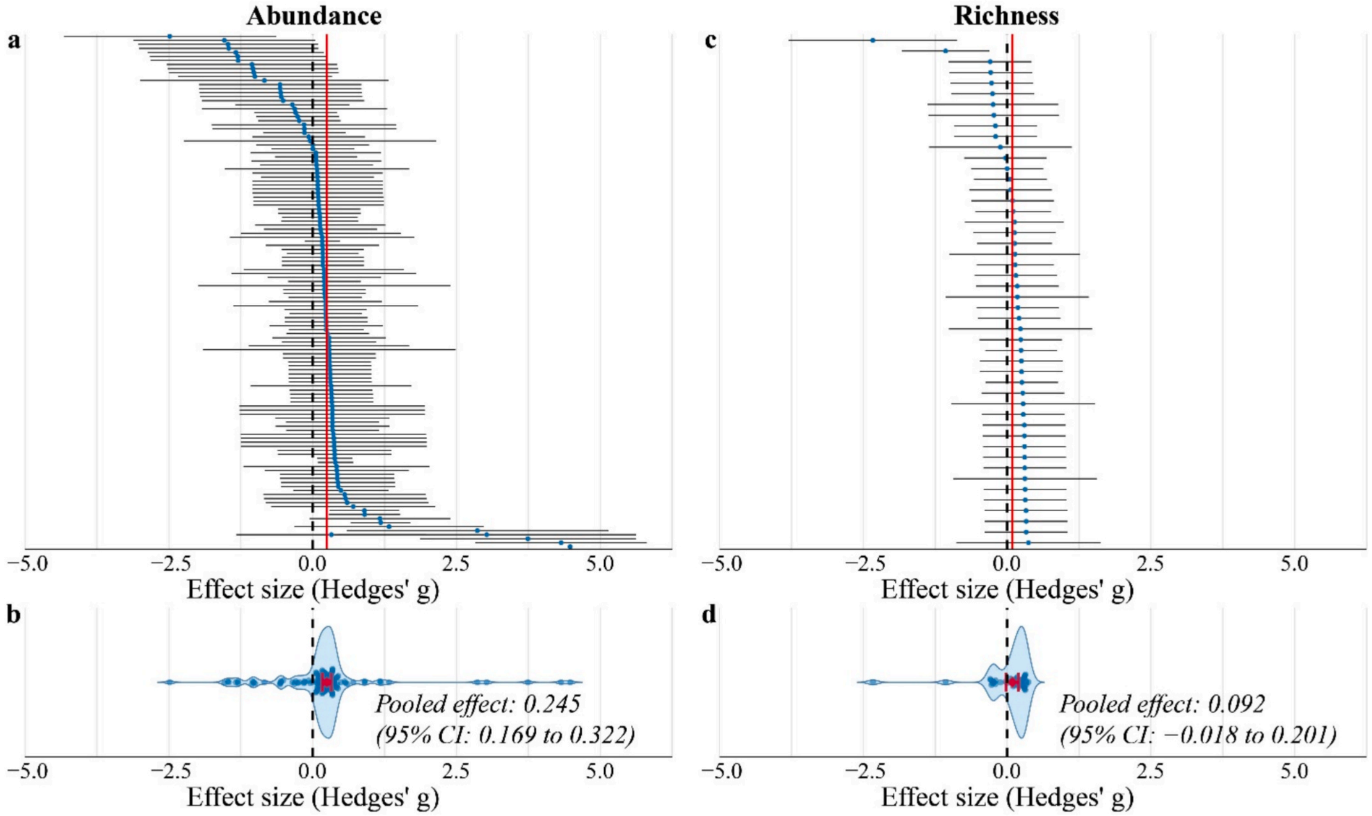


Fig. 4. Meta-analysis comparing all effect sizes for the abundance (left panel) and richness (right panel) of invasive species on artificial structures versus natural reefs. (a,c) Caterpillar plot showing all effect sizes and their corresponding 95% confidence intervals (black lines). (b,d) Orchard plot showing the distribution of effect sizes. In both plots, blue dots represent individual comparisons, and the red vertical line/points indicate the estimated mean effect size (Hedges' *g*). Positive effect sizes indicate that invasive species are more abundant on artificial structures compared to natural reefs. Abundance plots are based on 129 effect sizes extracted from 17 publications. Richness plots are based on 48 effect sizes extracted from 5 publications. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

The capacity to colonise AS can also be linked to a species' adaptive capacity and generalist diet, as in the case of lionfish (*P. volitans*) (Vega et al., 2015), reported in four studies included in the meta-analysis. Lionfish exhibit phenotypic plasticity by adjusting their feeding strategies based on prey availability (Cure et al., 2014; D'Agostino et al., 2020; Steell et al., 2019). This allows them to thrive in environments such as isolated oil platforms (Garner et al., 2019), where prey types and densities differ from those on NRs (Dahl and Patterson, 2014). Assessing the likelihood of potential invasive species in an area is therefore important when planning AS deployment because responses to new structures can be species-specific (Alvarez-Aguilar et al., 2022).

Here, we evaluated the risk of AS acting as invasion hubs using a meta-analysis, which found consistently higher invasive-species abundance on AS than on NRs. To understand and manage this risk, we integrate these quantitative results with a narrative synthesis that distils key findings, highlights knowledge gaps, identifies AS-specific risk factors, and translates them into practical recommendations to reduce establishment and spread.

4.1. *Meta-analysis of invasive species populations and communities between artificial structures and natural reefs*

4.1.1. *Proximity to transfer vectors, pollution impacts, stepping stones*

AS near ports and shore tended to have similar invasive species abundances to adjacent NRs, likely due to shared exposure to port-based transfer and reduction in native species community resilience due to exposure to various stressors (e.g. pollution) (Boon et al., 2023; Komyakova et al., 2022a; Rondeau et al., 2022) (Table 2, Fig. 5b,c). AS supported a significantly greater abundance of invasive species than NR at greater distances from shore or port, although the effect size was small. Shipping and boating are widely recognised as primary vectors, with organisms transported on hulls or in ballast water (Cariton and Geller, 1993; Drake and Lodge, 2007; Ruiz et al., 2000; Seebens et al., 2013). Most studies in this review were conducted in regions with intensive international shipping activity, including the English Channel (4 studies), Sydney Harbour (4) and the Mediterranean Sea (3) (Table A2). Ports and urbanised coastlines are associated with high pollution levels (Bulleri and Chapman, 2010; Burak et al., 2004; Burt, 2014; Komyakova et al., 2022a). Pollutants can alter ecosystem composition by weakening native assemblages and favouring invasive species dominance (Piola and Johnston, 2008; Sun et al., 2023), which may increase the likelihood of colonisation in both natural and artificial habitats. Natural habitats located farther from areas of disturbance may support more stable native communities that are resilient to invasive colonisation, resulting in lower invasive-species occupancy than on AS that may not have the advantage of having an established native community.

AS are typically deployed as bare substrates, providing competitor-free space for colonisers. Because many invasive species are opportunistic generalists and rapid colonisers and growers, they are well placed to establish on these structures ahead of native taxa. Offshore areas typically have fewer NRs, making infrastructure the only available hard substrate (Dahl and Patterson, 2014; Sammarco et al., 2004; Schulze et al., 2020). This scarcity of NR may reduce native propagule availability and colonisation rates (Epstein and Smale, 2018), weakening biotic resistance to invasion (Page et al., 2006). AS may thus serve as initial footholds for colonisation, with subsequent spread into natural habitats, especially in areas where natural community resilience was

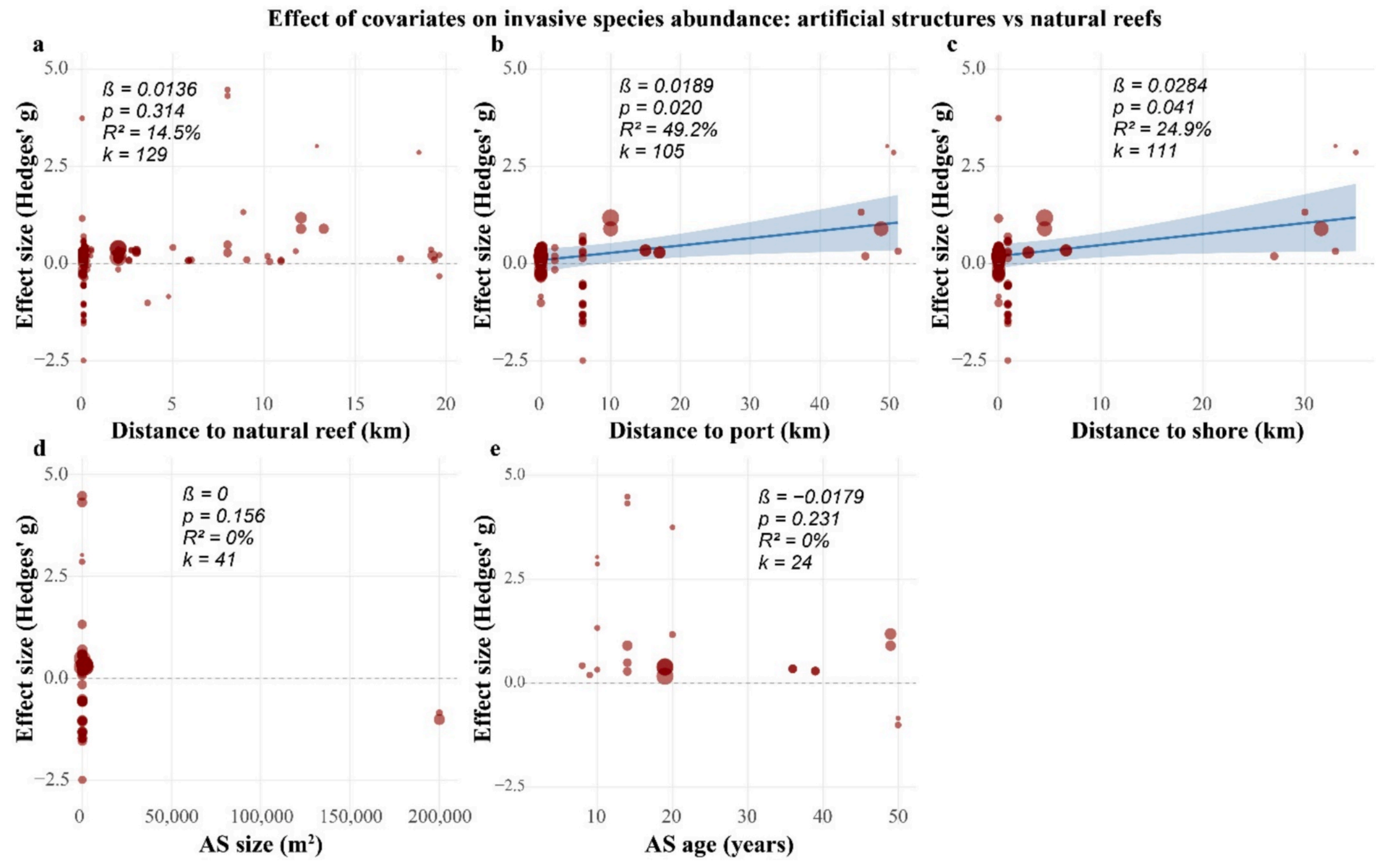


Fig. 5. Scatter plots showing effect sizes (red dots) against artificial structures' characteristics: distance to (a) natural reef, (b) ports and (c) shore, and structure properties (d) artificial structure size and (e) age. "k" denotes the number of comparisons (sample size). β represents the regression coefficient (mean effect size) with its statistical significance *p*. R^2 indicates the proportion of heterogeneity explained by the moderator. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

compromised (e.g. near ports, urbanised coastlines). As such, AS can act as invasion stepping stones. For example, the presence of the macroalgae *Undaria pinnatifida* – one of the world's worst invasive species – on rocky reefs has been linked to proximity to marinas (Epstein and Smale, 2018).

Distance from NR to AS, abundance of invaders on AS, colonisation capacity of invasive species (Epstein and Smale, 2018; Farrell and Fletcher, 2006) and local community resilience (Komyakova et al., 2022a) may influence the likelihood of spillover between the two habitats.

Although vectors of transfer and pollution likely prime AS for invasion, outcomes ultimately reflect multiple drivers, notably the propagule supply of native and non-native species, dominant invasive species in the region, their mode of transfer and time of reproduction (Bulleri and Chapman, 2010; Mayer-Pinto et al., 2015; Piola and Johnston, 2008; Sun et al., 2023).

4.1.2. *Age and space availability*

Space availability can be the main initial difference between natural and artificial habitats (Serrano et al., 2013). On natural substrata, space is a limited resource, with established native species being able to repel or compete with new arrivals (Elahi and Sebens, 2012; Peyton, 2009), making it challenging for some invasive groups including algae, Tunicata and Annelida to successfully occupy these substrata (Dafforn et al., 2012). Over time, AS may be progressively colonised by native communities that outcompete pioneer invasive taxa. Convergence between artificial and NR communities over time has been reported for fish assemblages (Komyakova et al., 2019), however not for the sessile communities at least based on one available long-term study (Relini et al., 1994). This suggests multiple mediating factors are likely involved and highlighting potential difference in the settlement cues between these habitat types.

We thus predicted that age of the structure may play an important role in terms invasive species abundance, due expected colonisation dynamics, with the recently deployed structures providing newly available space for fast colonisers and opportunistic species (Creed, 2006; Perkol-Finkel et al., 2005; Sampaio et al., 2012; Wendt et al., 1989). Contrary to our expectations, age was not a significant predictor; however, the youngest AS in our dataset was eight years old, limiting our ability to assess early colonisation dynamics. Furthermore, long-term studies are generally rare, preventing reliable conclusions on how colonisation evolves with time and highlight a vital research gap (Dean and Hurd, 1980). Lack of long-term monitoring data has been a hinderance for a range of other ecological assessments (e.g. Gonçalves et al., 2024; Sievers et al., 2024; Wang et al., 2024).

4.1.3. *Materials and design*

Unnatural material can favour the establishment of invasive species (Dodds et al., 2022; Glasby et al., 2007; Schaefer et al., 2024). Although no material type supported fewer invasive species than NRs, as expected, both concrete and metal were associated with significantly higher invasive species abundance compared to NRs. Some materials – including plastic, fibreglass and rock – did not support significantly higher abundances. However, replication across several material categories (e.g. plastic, fibreglass and rock) was low, and effect-size estimates for these materials should therefore be interpreted cautiously. A related meta-analysis of 46 studies examining the effects of natural versus artificial materials on colonising biota on built structures found that invasive species are more abundant on polymers and concrete than

Table 3
Meta-analysis results for the effect of covariates categorised on the difference between artificial structures and natural reefs. P-values reflect whether mean effect sizes significantly differ from 0 (i.e. the abundance of invasive species on artificial structures is modestly, but significantly higher or lower than natural reefs).

Covariates	k	Effect size β	SE	95% CI Lower	Upper	p-value
AS category						
Derelict constructions	26	−0.3844	0.1397	−0.6582	−0.1107	**0.0059**
Floating structures	36	0.1985	0.0781	0.0455	0.3515	**0.0110**
Habitat enhancers	10	1.1293	0.1804	0.7757	1.4828	**<0.0001**
Platforms	6	0.6981	0.1437	0.4164	0.9798	**<0.0001**
Shoreline structures	51	0.2205	0.0526	0.1174	0.3237	**<0.0001**
Material						
Concrete	56	0.2427	0.0610	0.1231	0.3623	**<0.0001**
Fiberglass	7	0.1965	0.1384	−0.0748	0.4678	0.1558
Metal	17	0.4513	0.1109	0.234	0.6686	**<0.0001**
Mix	4	0.1723	0.1647	−0.1504	0.4950	0.2953
Plastic	14	0.2335	0.1232	−0.0080	0.4750	0.0581
Rock	5	0.5804	0.3222	−0.0707	1.2314	0.0806
Wood	26	0.1543	0.0840	−0.0104	0.3189	0.0663
Invasive species						
Feeding guild						
Filter	70	0.2991	0.0574	0.1866	0.4116	**<0.0001**
Non-filter	62	0.2054	0.0695	0.0692	0.3417	**0.0031**
Movement category						
Mobile	47	0.3323	0.0786	0.1782	0.4863	**<0.0001**
Sessile	76	0.2295	0.0455	0.1403	0.3187	**<0.0001**

Bold p-value indicates statistical significance.

on wood (Dodds et al., 2022). Similarly, laboratory experiments revealed that plastic was more favourable than concrete and wood for the settlement of invasive species, including larvae of a common bryozoan *Bugula* spp. (Pinochet et al., 2020). Preference for substrate type appears to be determined during the early stages of colonisation for sessile organisms (Vaz-Pinto et al., 2014), affecting the development of the community over time (Anderson and Underwood, 1994). Further research into eco-friendly alternatives like biogenic concrete, clay, ceramics and sustainable aggregates may identify options that provide engineering benefits (i.e. strength, durability) while supporting ecological sustainability and long-term environmental health (Komyakova et al., 2022b) and reducing invasive species colonisation rates.

4.2. Data quality and gaps

Despite broad search terms and an exhaustive review, only 26 studies comparing invasive species on both AS and NRs were identified, with 20 providing mean values and variances suitable for meta-analysis, highlighting a significant research gap. These 20 studies yielded 129 and 48 effect sizes for examining differences in invasive species abundance and richness, respectively, between AS and NRs. However, the small number of available studies reduced analytical power, especially for evaluating the effects of explanatory variables.

Existing research is concentrated in tropical, sub-tropical and temperate waters, with most data originating from the USA, Australia and a few countries in Europe. Together, these regions represent only eleven countries with more than 95% of countries globally lacking empirical coverage. This major limitation in knowledge of marine invasive species distributions on AS across the globe prevents reliable generalisations and constrains capacity to reduce invasion risk. Expanding research to a broader range of regions, with a focus on comparative studies of natural and artificial habitats, is essential for improving broader understanding. Such studies provide valuable context for assessing AS performance (Carr and Hixon, 1997; Marzinelli et al., 2011) and highlight key differences between habitats, including structural factors and the life-history traits of recruited organisms

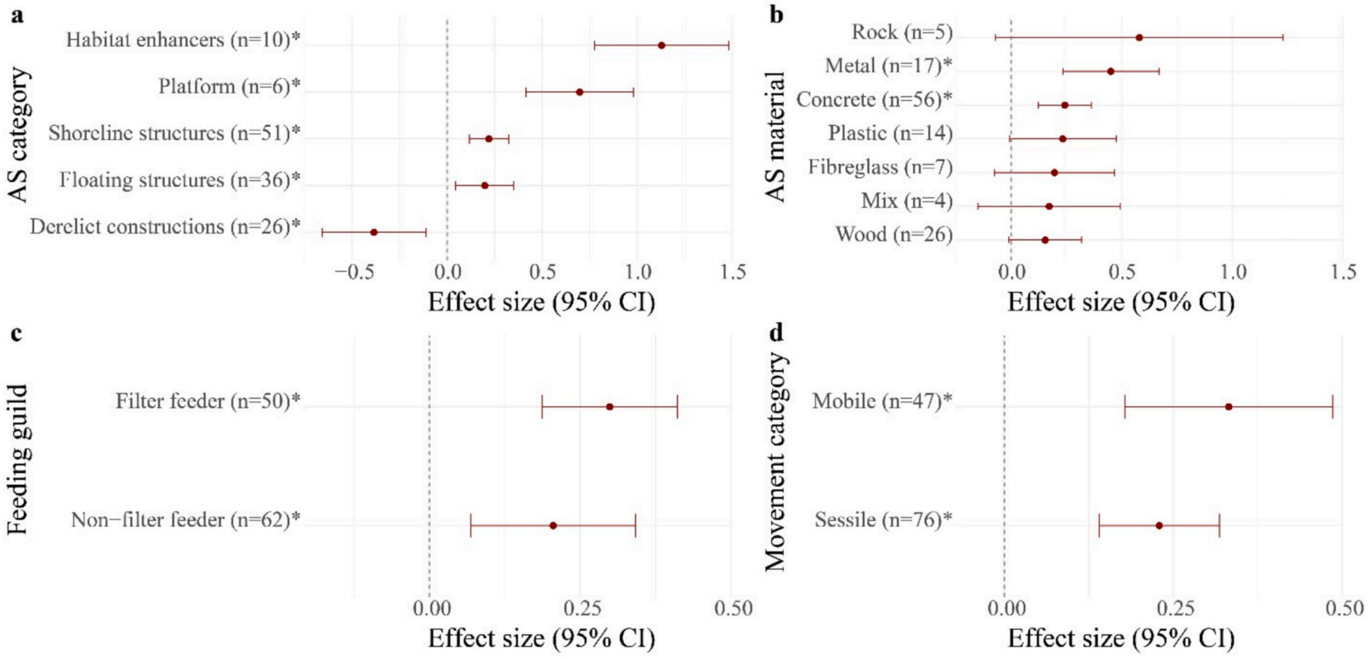


Fig. 6. Effect of covariates on the difference in invasive-species abundance between AS and NRs. Covariates include (a) AS category, (b) AS material, (c) feeding guild and (d) movement category. Red points are mean estimates with 95% CIs; values in brackets indicate sample size (number of comparisons). Asterisks (*) denote statistically significant estimates (CI does not overlap zero). Positive effect sizes indicate that the covariate is associated with higher invasive species abundance on AS relative to NRs (greater differences), whereas negative effect sizes indicate that the covariate is associated with lower invasive species abundance on AS relative to NRs (smaller differences). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

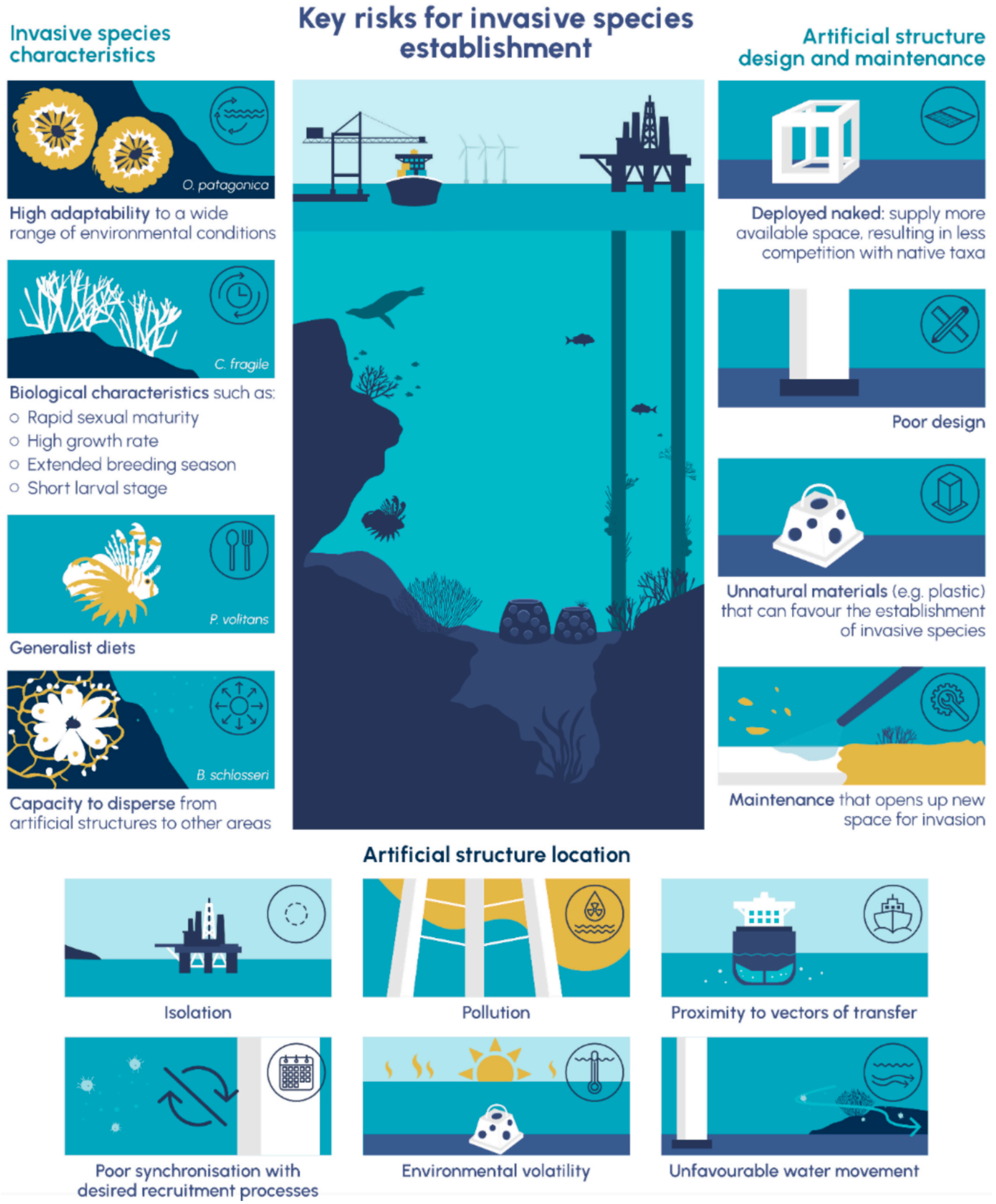


Fig. 7. Key risks for invasive species establishment. Characteristics of invasive species (left) that enable them to colonise new environments. Factors to consider in artificial structure deployment that can contribute to invasion risk include design and maintenance (right) and artificial structure location (bottom). Figure designed by Dr. Stacey McCormack (www.visualknowledge.design).

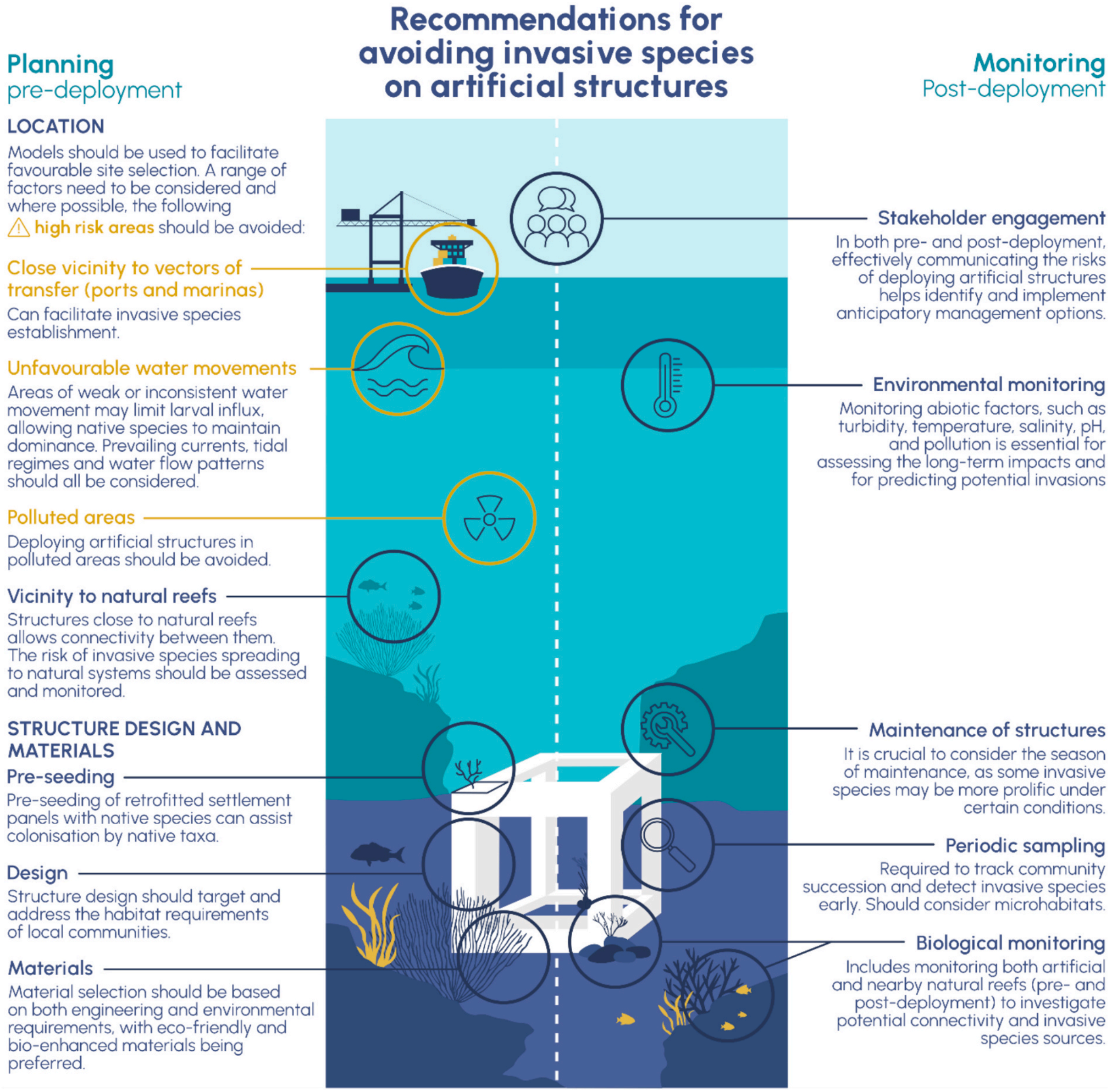
(Ignacio et al., 2010; Perkol-Finkel and Benayahu, 2004; Perkol-Finkel and Benayahu, 2007). Further, many studies provided none or only partial information on site characteristics (e.g. reef size, age and construction material), limiting analyses, including the modelling of interactions among multiple factors. Researchers are encouraged to provide comprehensive details of examined habitat characteristics to facilitate robust data analysis and improve detection of global, generalisable patterns.

In addition to the geographic bias, synthesising available studies necessarily requires pooling data across diverse systems and study designs. Such pooling data may introduce unaccounted heterogeneity including study design, spatial and temporal scales. Although such heterogeneity may influence effect size estimates, the direction of the observed effects provide confidence that the main conclusions of this meta-analysis remain consistent.

Nineteen studies collected data for less than one year, likely constrained by financial and logistical challenges (Campbell et al., 2007). Although short-term surveys are valuable for documenting static differences between AS and NRs (Byers, 2002; Valiela, 1995), they do not capture how community establishment evolves over time. Extending monitoring durations is essential to determine invasive species dynamics on individual AS over time (Bennett et al., 2019; Dafforn, 2017; Kakkonen et al., 2019). Furthermore, none of the studies provided data from the early period following initial deployment. As a result, knowledge of early colonisation remains limited. Early and frequent monitoring should therefore be incorporated into management plans to enable timely detection, track invasion patterns, assess establishment success, and evaluate potential spillover between natural and artificial habitats (Campbell et al., 2007; Castro et al., 2021) (see Section 4.3.8).

Across the literature, some AS–NR “pairs” were separated by relatively large distances (>20 km), risking confounding of invasion patterns by underlying environmental gradients and differences in propagule pressure. To strengthen inference, future studies should pair sites at ecologically meaningful scales, ideally within the same embayment or hydrodynamic cell, and match key attributes (depth, exposure, substrate), while reporting exact inter-site distances.

Systematic evaluation of the literature also highlighted substantial inconsistency in data reporting. Baseline site characteristics (e.g. depth, sediment type) and habitat descriptors (e.g. size, age, material) were frequently missing or only partially reported, often necessitating



Recommendations for avoiding invasive species on artificial structures

Planning
pre-deployment

LOCATION

Models should be used to facilitate favourable site selection. A range of factors need to be considered and where possible, the following ⚠ **high risk areas** should be avoided:

Close vicinity to vectors of transfer (ports and marinas)

Can facilitate invasive species establishment.

Unfavourable water movements

Areas of weak or inconsistent water movement may limit larval influx, allowing native species to maintain dominance. Prevailing currents, tidal regimes and water flow patterns should all be considered.

Polluted areas

Deploying artificial structures in polluted areas should be avoided.

Vicinity to natural reefs

Structures close to natural reefs allows connectivity between them. The risk of invasive species spreading to natural systems should be assessed and monitored.

STRUCTURE DESIGN AND MATERIALS

Pre-seeding

Pre-seeding of retrofitted settlement panels with native species can assist colonisation by native taxa.

Design

Structure design should target and address the habitat requirements of local communities.

Materials

Material selection should be based on both engineering and environmental requirements, with eco-friendly and bio-enhanced materials being preferred.

Monitoring
Post-deployment

Stakeholder engagement

In both pre- and post-deployment, effectively communicating the risks of deploying artificial structures helps identify and implement anticipatory management options.

Environmental monitoring

Monitoring abiotic factors, such as turbidity, temperature, salinity, pH, and pollution is essential for assessing the long-term impacts and for predicting potential invasions

Maintenance of structures

It is crucial to consider the season of maintenance, as some invasive species may be more prolific under certain conditions.

Periodic sampling

Required to track community succession and detect invasive species early. Should consider microhabitats.

Biological monitoring

Includes monitoring both artificial and nearby natural reefs (pre- and post-deployment) to investigate potential connectivity and invasive species sources.

Fig. 8. Recommendations for preventing invasions on artificial structures. Figure designed by Dr. Stacey McCormack (www.visualknowledge.design).

additional effort to obtain supplementary information and, in some cases, resulting in irretrievable data gaps. These limitations constrained the ability to assess the influence of explanatory variables. Similarly, information on cleaning regimes of artificial habitats was reported in only a single study. Such data are critical for understanding invasion potential because cleaning practices may both create available space for colonisation and increase propagule release, potentially influencing propagule spillover between artificial and adjacent natural habitats. It is therefore recommended that future research examining invasive species colonisation of artificial and natural habitats includes detailed and standardised reporting of site characteristics, habitat attributes, and use or maintenance practices (see Section 4.3.7).

4.3. Further risk reduction strategies and recommendations for deploying AS

Anticipating future biological invasions is the most cost-effective approach to risk reduction (Sheehy and Vik, 2010). Preventing invasions requires careful consideration of key factors during the planning and post-deployment stages (Fig. 8). The planning stage should prioritise site selection, assessment of environmental conditions at the structure site, and evaluation of potential substrata, whereas post-deployment should focus on monitoring.

4.3.1. Poor synchronisation with native colonisation processes

Deployment timing of AS can critically influence colonisation outcomes. Many marine species have specific recruitment periods, triggered by environmental cues such as temperature, salinity, light or chemical signals (Kingsford et al., 2002; Shima et al., 2018; Westerbom et al., 2021; Wheeler, 2016). For example, spat settlement in oysters and mussels typically occurs seasonally (Ruwa and Polk, 1994), and kelp recruitment often coincides with winter storms (Kimmel et al., 2024; Reed et al., 1988). Poor synchronisation with native recruitment can leave AS unoccupied by native taxa, creating niches for invasive species, often generalists with broader recruitment windows (Chan and Briski, 2017; Thornber et al., 2004). Deploying AS during invasive species recruitment but outside native cycles increase vulnerability to colonisation, as early establishment often hinges on timing (Benedetti-Cecchi, 2000; Bulleri and Airoldi, 2005; Crawley, 1989). To minimise invasion risk, deployment should be aligned with native recruitment cycles and informed by local ecological conditions.

4.3.2. Environmental volatility

Environmental volatility – such as marine heatwaves, cold snaps or high rainfall – may significantly influence the susceptibility of AS and NRs to biological invasions. These events create periods of environmental stress during which local species experience reduced resilience, leaving ecological niches vulnerable to colonisation by opportunistic invasive species (Gallego-Tévar et al., 2019). For example, marine heatwaves disrupt thermal thresholds, favouring thermophilic, invasive species (e.g. *Symplegma brakenhielmii* in New Zealand) that may outcompete native fauna (Smale et al., 2019; Spyksma et al., 2024).

Similarly, high rainfall events alter salinity and nutrient profiles, reducing the competitive advantage of native species adapted to stable conditions and creating opportunities for euryhaline invaders. For instance, a study on adult copepods of the genus *Acartia* in the Bay of Biscay highlighted the interactive effects of environmental variability, where a decrease in dissolved oxygen saturation led to a decline in the dominant species (*Acartia clausi*), whereas an increase in temperature facilitated the successful establishment of the invasive *A. tonsa* (Aravena et al., 2009). Conversely, Firth et al. (2011) found that cold snaps can limit the further spread of the invasive mussel *Perna viridis*. As responses to changing conditions are species-dependent, decisions on the timing of deployment of new AS should be guided by the understanding of how invasive taxa may response to environmental change and predicted environmental changes for relevant regions under future climate change conditions.

4.3.3. Water movement

Water movement can play a critical role in shaping the colonisation dynamics of marine invasive species on AS, NRs and other marine environments. Larval dispersal pathways – often referred to as "larval highways" – are influenced by prevailing currents, tidal regimes and water flow patterns. These pathways can determine whether a location is rapidly colonised by invasive species (e.g. lionfish in the Mediterranean; Bottacini et al., 2024) or more slowly settled by native organisms. AS positioned in high-flow regions may experience an overrepresentation of non-native species due to increased larval supply from distant sources. AS can also create barriers that limit water movement, restricting the dispersal of invasive propagules and allowing them to dominate within a restricted area (e.g. semi-enclosed bay) while also reducing the influx of native species (Bishop et al., 2017). Conversely, areas removed from vectors of transfer and characterised by weak or inconsistent water movement may limit larval influx, allowing native species to maintain dominance. Understanding the interaction between hydrodynamics and larval dispersal is essential for predicting invasion risks and informing the placement and design of AS to minimise their role as stepping stones for invasive species.

4.3.4. Location selection

Local abiotic and anthropogenic conditions, such as pollution, wave and current regimes, and sheltered, low-exchange spaces, strongly shape invasion risk, with harbours (including ports and marinas) and offshore infrastructure warranting special caution (Wilding and Sayer, 2002). Therefore, AS intended for conservation or fisheries should be sited away from polluted areas or major introduction vectors (Firth et al., 2014); where infrastructure must occur in such settings or is isolated offshore, stricter management and frequent monitoring (e.g. enhanced ballast-water management) are essential to minimise invasion. Siting AS near NRs may reduce invasive colonisation on the structures via native propagule supply, but it can elevate the risk of spread into adjacent natural systems (Airoldi et al., 2015; Schulze et al., 2020; Sheehy and Vik, 2010), so both AS and neighbouring NRs require close monitoring.

To optimise site selection and minimise ecological risks, hydrodynamic models of current movement and their influence on larval dispersal for both native and invasive species should be considered when deploying AS. Especially agent-based models (ABMs), being able to account for the interaction between larval behaviour and fluid dynamics (Strickland et al., 2022), can provide valuable insights into how these structures might influence the existing hydrodynamic connectivity between NRs and potentially facilitate the spread of invasive species (Komyakova et al., 2022a; Yñiguez et al., 2008). By identifying areas where currents may transport invasive propagules or facilitate the recruitment of native species, these models enable planners to strategically select deployment sites, thereby reducing the risk of biological invasions and aligning with ecological goals (Dye et al., 2022).

4.3.5. Structure design

Eco-engineering interventions that enhance the complexity of AS have been implemented globally to increase the abundance and biodiversity of native species while reducing invasive species in harbours and ports (e.g. the World Harbour Project and the Sydney Harbour Research Program; Steinberg et al., 2016). However, the success of these efforts has been variable (Strain et al., 2017; Strain et al., 2020; Toh et al., 2017; Ushiama et al., 2019). In certain locations, enhanced structural complexity has been shown to promote greater diversity of associated species (Ushiama et al., 2019) and higher survival rates of naturally recruiting coral larvae (Randall et al., 2021). However, other studies have found that eco-engineering that enhances structural complexity can also benefit invasive species (Gauff et al., 2023; Gauff et al., 2025). These discrepancies and limitations highlight the need for further research on structure design, long-term monitoring and large-scale application.

4.3.6. Seeding of the infrastructure

Eco-engineering interventions to prevent the establishment of invasive species on marine artificial infrastructure involve assisting the colonisation by native taxa or the purposeful pre-seeding of retrofitted settlement panels with native species that provide competition to newcomers, creating a barrier against the establishment of invasive species (Dafforn, 2017; Ng et al., 2015; Ohayashi et al., 2022; Toh et al., 2017). However, several studies that examined effects of pre-seeding on community establishment reported high mortality rates for pre-seeded organisms (Perkol-Finkel et al., 2012; Strain et al., 2020), resulting in concerns regarding the effectiveness of this technique. Most of these studies were conducted over relatively short time frames (up to 12 months) (Perkol-Finkel et al., 2012; Strain et al., 2020), failing to evaluate the longer-term effects of pre-seeding in terms of controlling the abundances of invasive species.

Pre-seeded taxa may act as "place holders" and "space indicators" for native communities, potentially encouraging natural recruitment. It has been hypothesised that artificial infrastructure facilitates invasion establishment partly because it is deployed "naked", providing free space for faster-colonising, rapidly growing invasive species (Komyakova et al., 2022a). Marine organisms commonly use various habitat cues, including the chemical signals of con– and hetero-specifics to detect suitable settlement sites (Kingsford et al., 2002). Pre-seeding AS could therefore support the establishment of native communities through provision of competition for space and attraction of native settlers. However, understanding the long-term effectiveness of this technique requires further monitoring and research.

4.3.7. Effects of disturbance and maintenance of structures

Human disturbances, including maintenance (Airoldi and Bulleri, 2011) and boat-anchoring (Beca-Carretero et al., 2024) activities, can affect the composition, abundance and distribution of species colonising AS, potentially creating new spaces for invasive species (Airoldi and Bulleri, 2011). Therefore, it is crucial to consider the timing of maintenance because some invasive species may proliferate under certain conditions (Airoldi and Bulleri, 2011). Planning maintenance based on species ecology can help reduce the risks of invasive species spread.(e.g. Erlandsson et al., 2006; Hutchings et al., 2023).

Another important consideration in AS management is the relocation or removal of these structures (Elden et al., 2025). When invasive species are dominant on AS, removal or relocation can contribute to their spread, as infrastructures and their associated communities may be transported during towing (Wanless et al., 2010). Conversely, when invasive species occur at low densities and AS do not act as stepping stones for invasion, removal may significantly reduce local biodiversity and biomass (Coolen et al., 2020).

It is therefore essential to assess the abundance and richness of invasive species, as well as their ecological impacts, before removing or relocating AS. Consideration should also be given to alternatives such as environmentally responsible protocols for decommissioning and structure transport (Braga et al., 2021). Likewise, more evidence-based management strategies that account for multiple spatial and temporal scales are urgently needed, as research in this area remains limited (Schläppy et al., 2021).

4.3.8. Monitoring

Once an AS is deployed, continuous monitoring is crucial to evaluate its effectiveness and prevent undesirable outcomes (Blount et al., 2021). Monitoring must include biological and environmental factors, as these can interact synergistically and influence the development of the marine community on AS (Komyakova et al., 2022a; Scarborough Bull and Kendall, 1994).

Biological monitoring should include both artificial and nearby NRs – especially in high-risk areas such as harbours or urbanised areas (O'Shaughnessy et al., 2020), as well as natural invasion "hot spots" such as oyster reefs (Rohde et al., 2017) – to investigate potential connectivity and invasive species sources (Sheehy and Vik, 2010). Periodic sampling is necessary to track community succession and detect invasive species early (Hylkema et al., 2021; Schulze et al., 2020).

Monitoring abiotic factors – such as turbidity, temperature, salinity, pH and pollution – is fundamental for assessing long-term impacts (Bortone and Kimmel, 2013) and predicting invasions (Crooks et al., 2011; Kenworthy et al., 2018b; Occhipinti-Ambrogi, 2007). Moreover, comparing post- and pre-deployment data allows for the assessment of temporal and spatial variations, helping to anticipate potential negative effects (Blount et al., 2021; Kakkonen et al., 2019). Monitoring efforts should be intensified following disturbance like storm damage or pollution events, which can create new spaces for invasive species (Airoldi and Bulleri, 2011; Schulze et al., 2020). Most studies included in this meta-analysis were of short duration. following London Convention and Protocol/UNEP (2009), we recommend a minimum of five years of monitoring with annual surveys to assess reef stability for newly deployed infrastructure.

4.3.9. Stakeholder engagement

In both the pre- and post-deployment stages, communication and interdisciplinary collaboration amongst stakeholders is critical (Naylor et al., 2012). Effectively communicating the risks helps identify and implement anticipatory management options (Sheehy and Vik, 2010). Sharing risk data and experiences with other programs and the public (Bartholomew et al., 2022) manifest the commitment of managers to encourage sustainability and assist in the success of other projects (Bartholomew et al., 2022). These communications, knowledge sharing and research efforts are particularly vital in the era of growing offshore development and associated public concerns raised around decommissioning.

4.3.10. Other considerations

AS not only attract invasive species but can also provide habitat for range-shifting species (Paxton et al., 2019). This role is particularly crucial for species affected by climate change, as they expand their range poleward in search of cooler waters (Chester and Robson, 2013). For example, the poleward expansion of the mangrove tree crab (*Aratus pisonii*) has been facilitated by the presence of docks within salt marsh habitats (Cannizzo and Griffen, 2019). Similarly, the movement of tropical fishes is supported by AS that act as habitat corridors (Paxton et al., 2019). However, evidence for the role of AS in facilitating species range shifts remains scarce. Further research is therefore required to determine whether artificial habitats may function as climate refugia, their role in range expansion and any associated ecological risks.

5. Conclusion

Existing research suggests that AS are significantly more susceptible to colonisation by invasive species than natural habitats such as oyster, coral and rocky reefs across temperate, subtropical and tropical zones. The space availability provided by these structures, their proximity to transfer vectors (e.g. shipping) and their exposure to pollution in urbanised areas are likely to create favourable conditions for colonisation and establishment. Invasive species success on AS is further facilitated by biological traits that allow them to outcompete native communities. However, significant data gaps, a lack of long-term studies and limited geographic coverage constrain the capacity to detect and track invasions over time and at a global scale, and to evaluate the influence of key explanatory variables.

Under ongoing global change, the role of AS in shaping invasion dynamics is likely to intensify. The frequency and intensity of extreme weather events such as heatwaves and severe storms are predicted to continue increasing (Fox-Kemper et al., 2023) in tropical and subtropical regions (Bacmeister et al., 2018). Similar trends are also expected in temperate regions (Ridgway, 2007). For example, in temperate Australia alone, ocean temperatures are warming at approximately four times the global average (Oliver et al., 2018). These changes pose increasing threats to coastal ecosystems and infrastructure and have already driven substantial declines in natural habitats, including up to 70% loss of coral cover in some areas (De'ath et al., 2012; Morais et al., 2020), and over 90% loss of canopy-forming kelp forests in Tasmania (Butler et al., 2020; Johnson et al., 2011). Climate change adaptation responses are therefore likely to drive further expansion of AS in coastal and marine environments, including coastal protection infrastructure (e.g. breakwater walls) and purpose-built artificial habitats supporting reef-based fisheries and marine aquaculture (Bugnot et al., 2021; Luo and Komyakova, 2025). Offshore energy production has already been widely implemented across Europe and other regions as a climate-change mitigation strategy and is expected to expand further in the near future.

Climate-driven stressors such as marine heatwaves, altered salinity regimes and increased disturbance frequency can reduce the resilience of native communities, thereby lowering biotic resistance to invasion. In combination with the rapid expansion of marine infrastructure, these factors are likely to increase invasion risk by facilitating the establishment and spread of invasive species associated with AS. Addressing key knowledge gaps and adopting strategic, climate-aware planning therefore represent the most effective means of reducing future invasion risk. Operationally, this requires embedding invasion ecology into marine spatial planning through informed siting and design of artificial infrastructure, anticipation of climate-mediated disturbances, and sustained post-deployment monitoring. AS represent both a challenge and an opportunity. If unmanaged, they may amplify invasion risk in a warming ocean. However, if strategically designed and monitored, they offer a critical intervention point for reducing ecological impacts in an increasingly human-modified seascape.

CRediT authorship contribution statement

Laura Avila-Turriago: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation. **Jasmine B.D. Jaffrés:** Writing – original draft, Writing – review & editing, Data curation, Validation, Visualization. **Michael Sievers:** Writing – review & editing, Visualization, Validation, Methodology, Formal analysis. **Brigette Wright:** Writing – review & editing, Validation, Methodology, Investigation, Data curation. **Ariane Easton:** Writing – review & editing, Validation. **Louise B. Firth:** Writing – review & editing. **Antony M. Knights:** Writing – review & editing. **Niclas Einert:** Writing – review & editing, Writing – original draft. **Tom Coughlin:** Writing – review & editing, Conceptualization. **Naomi Forrest:** Writing – review & editing. **Sven Frijlink:** Writing – review & editing. **Valeriya Komyakova:** Writing – review & editing, Writing – original draft, Project administration, Methodology, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Jasmine B. D. Jaffrés, Michael Sievers, Louise B. Firth, Antony M. Knights, Niclas Einert, and Tom Coughlin declare that they have no competing financial or personal interests. A portion of Valeriya Komyakova's salary, as well as the casual salaries of Laura Avila, Brigette Wright, Ariane Easton, and Naomi Forrest, were supported by funding awarded by the Department of Natural Resources and Environment Tasmania (NRE Tasmania) to Valeriya Komyakova and administered by the University of Tasmania.This research was funded by NRE Tasmania. Co-author Sven Frijlink, was an employee of NRE Tasmania, and contributed to the manuscript through editorial comments, review, and editing. Their involvement did not influence the analysis, interpretation, or conclusions of this work. The authors declare no other conflicts of interest.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.gloenvcha.2026.103137.

Data availability

Summary data are available in the figures and supplementary materials. Raw data are available on request.

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