British Kelp Forest Restoration: Feasibility Report

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1. Background on marine forests

1.1. Kelp forest ecosystems

Kelps are large brown macroalgae (seaweeds) belonging to the Order Laminariales, which form forest-like habitats that constitute some of the most widespread and extensive coastal ecosystems globally (Steneck et al., 2002, Teagle et al., 2017). Kelp forests are distributed from subtropical through to polar regions, covering between one-quarter to one-third of the world's coastline (Wernberg et al., 2019a, Jayathilake and Costello, 2020). In total, more than 110 species of Laminarian or 'true' kelp have been described (Bolton, 2010), although relatively few genera comprise the majority of kelp diversity and dominate in terms of distribution and abundance (Teagle et al., 2017). In addition, some other large brown macroalgal species (mostly belonging to the Orders Fucales and Tilopteridales) can from extensive habitats and fulfil similar functions within coastal ecosystems and, as such, are of comparable ecological importance. Kelp species have been recorded on all continents, extending from low intertidal shores to depths of 150 m or more, while kelp-derived detritus can be transported substantial distances from its origins, and has been recorded at depths in excess of 400 m) (Filbee-Dexter et al., 2018). The distribution of kelp species is primarily constrained by temperature, wave exposure, light and nutrient availability, although factors such as grazing pressure, competition and sedimentation can also be important.

1.2. Kelp associated biodiversity

Kelps are habitat-forming foundation species that, like seagrass, mangroves and corals, modify environmental conditions, species interactions, community structure and ecosystem functioning, supporting diverse assemblages of understorey algae and macrofauna (Steneck et al., 2002, Teagle et al., 2017, Bué et al., 2020).

The main characteristics that make kelp foundation species are their large fronds that absorb light and remove nutrients from the water column, as well as altering local hydrodynamics (Eckman et al., 1989, Schiel and Foster, 2015, Wernberg et al., 2019a). Some kelp species reduce sedimentation rates on the underlying reef through frond whiplash and abrasion (Kennelly, 1989, Toohey et al., 2004), and dense kelp forests may dampen wave forces, resulting in calmer microhabitats within the beds (Mork, 1996). Kelp canopies typically reduce light availability on the seafloor by >90%, creating darker stratified subcanopy habitats (Reed and Foster, 1984, Pedersen et al., 2014). Kelp also assimilate inorganic nitrogen and carbon, deplete local nutrient concentrations, increase pH and alter water chemistry (Krause-Jensen et al., 2016).

Kelps support elevated biodiversity by increasing habitat volume, heterogeneity, and complexity, and through the direct provision of food and shelter (Teagle et al., 2017). Most kelp species form large, complex biogenic structures which offer a substantial area for colonisation by numerous species of other macroalgae and invertebrates. Moreover, some kelp species, such as *Laminaria hyperborea,* are long lived and can survive for up to 20 years (Rinde and Sjøtun, 2005), providing a stable habitat. Different parts of the adult kelp 'plant', called a sporophyte, offer micro-habitats that support distinct communities. The holdfast, which anchors kelp to the substrate, is typically complex and intricate, where cavities and interstitial spaces between the root-like haptera create a living space for mobile invertebrates and even fish (Anderson et al., 2005, Teagle et al., 2018). For example in New Zealand, more than 350 species were found to be associated *Ecklonia radiata* holdfasts (Anderson et al., 2005), and in the UK >260 species have been identified from *Laminaria hyperborea* holdfasts (Teagle et al., 2018). Larger fauna, such as crabs, pipefish and rockfish have also been found to overwinter and take refuge in holdfasts (Salland and Smale, 2021). The kelp stipe is structurally simpler than the holdfast but still offers surface area for attachment of sessile invertebrates and macroalgae. The diversity and abundance of stipe-associated communities is extremely variable between kelp species, with some species' stipes devoid of epibionts, while others host lush plant and animal communities. For example, stipes of *Laminaria hyperborea* support abundant epiphytic red algae and sponges (King et al., 2021), which in turn offer additional food and living space for a wide variety of mobile invertebrates (Christie et al., 2003), whereas the stipes of *Laminaria digitata* are devoid of epiphytes (Teagle and Smale, 2018). Stipe-associated mobile invertebrates are prey items for fish and large crustaceans and form an important link in the local food web. Finally, kelp fronds (or blades) are typically simple leaf-like structures with a large surface area for photosynthesis, and they generally support relatively low-diversity communities. At the wider seascape scale, kelp canopies also offer extensive nursery, shelter and foraging areas for ecologically and socioeconomically important and iconic marine wildlife, including seals, sea otters, sea birds, sharks and large predatory fish (Estes et al., 2004).

1.3. Ecosystem services

Kelp forests support a range of ecosystem services (ES) that have recently been evaluated at both regional (Blamey and Bolton, 2018, Bennett et al., 2016) and global (Eger et al., 2023) scales. Perhaps the most widely recognised kelp associated ES relate to (i) carbon uptake, storage and transfer, (ii) nutrient cycling, and fisheries habitat provision (Lutz, 2023). Kelp habitats are among the most productive ecosystems on Earth, exhibit high rates of carbon uptake and release (Mann, 1973), and include one of the fastest-growing primary producers on the planet: *Macrocystis pyrifera*. The standing stock biomass of some kelp populations (e.g. *Macrocystis pyrifera, Ecklonia maxima*) can be very high and holds significant quantities of carbon. Vast amounts of kelp-derived organic matter is released into the marine environment as detritus, fuelling inshore food webs (Smale et al., 2021). This material can serve as a trophic subsidy to other habitats which in turn support rich and abundant faunal communities. Moreover, some of the carbon exported from kelp forests may accumulate in carbon storage habitats (e.g. seagrass meadows, offshore sediments, deep sea), increasing the potential for natural carbon sequestration (i.e. 'blue carbon'). The fate of kelp-derived carbon is poorly understood, but given the vast amount of organic matter released by kelp forests (Pessarrodona et al., 2018), if even a small fraction of kelp-carbon is retained in the marine environment for meaningful timescales this represents an important ES (Krause-Jensen and Duarte, 2016).

Owing to their high rates of primary productivity and turnover, kelp forests also cycle significant nutrient loads. By drawing excess nutrients out of the water, primarily nitrogen and phosphorus, kelps provide a valuable ES, particularly in areas with elevated nutrient concentrations. Kelp genera such as *Macrocystis*, *Nereocystis*, *Laminaria*, and *Ecklonia* are estimated to remove between 148 to 1900 kg of nitrogen per hectare per year and 8 to 216 kg of phosphorus (Eger et al., 2023). Because kelps also compete with phytoplankton (microalgae) for nutrients and light resources, they have the potential to mitigate the impacts of harmful microalgal blooms, which can be detrimental to ecosystems or humans (Jiang et al., 2020).

Figure 1. Laminaria hyperborea *kelp forest with diverse understorey community. Credit DS.*

Kelp forests offer a complex three-dimensional habitat and provide food and shelter to coastal fish, shellfish and other invertebrates. Commercial, recreational and subsistence fishery species may use kelp habitats as spawning, nursery, transitional, or foraging grounds, and some species rely on kelp forests throughout their life cycle. Several studies have shown positive links between kelp forest extent and condition and the productivity and yield of coastal fisheries (Bertocci et al., 2015). In some regions, coastal species that strongly depend on kelp forest habitat (e.g. rock lobster in Australia, lobster along the East coast of America, abalone in northeast Asia) are among the most economically valuable fisheries supporting regional economies (Eger et al., 2023). Recreational fisheries that target kelp-associated species are hugely important in several regions, including northwest Europe, southern Australia and the USA (Lutz, 2023).

Kelp forests also support other, less well recognised ES. Most notably, kelp and other seaweeds have been gathered or harvested for natural products by coastal communities for millennia. Harvested kelp is used for human food, livestock feed, fertilisers, and a wide range of industrial products. Kelp is also the primary source of alginates which are used in over 600 products such as thickening and gelling agents in the food and feed processing industry, as bonders, stabilizers, and emulsifiers in the pharmaceutical industry, for wound care in medicine, waterproofing in the textile industry, and in wastewater treatment. According to the FAO (2021), 14 countries harvest wild kelp, with 48% of harvested biomass coming from Chile, and 22% from Norway. There has been increasing interest in developing kelp harvesting practises in serval countries including Peru and Scotland, although in general, kelp cultivation (i.e. aquaculture) is the preferred approach to meeting increasing demands.

Kelp forests may also offer ES relating to biogenic coastal defence, through wave dampening, and local ocean acidification alleviation (Hirsh et al., 2020, Morris et al., 2020a, Xiao et al., 2021). However, the evidence base for these ES remains limited, and further empirical investigations are required. Finally kelp forests have considerable socioeconomic and cultural value. Recreation and tourism activities featuring or relying on kelp forests are significant (e.g. diving, wildlife watching, fishing), whilst kelp species feature prominently in many cultures, including indigenous peoples of the Northwest Pacific and Australia.

1.4. The UK context

Quantitative research on UK kelp forests began nearly 80 years ago, following a demand from the Ministry of Supply to produce camouflage textiles and other goods from kelp-derived alginates during the Second World War (Parke, 1948, Woodward, 1951). In the early 1950s, attempts were made to quantify the total standing stock of kelp as a potential exploitable resource. The total biomass of subtidal kelp around Scotland (mostly *Laminaria hyperborea*) was estimated as 10 million tons over an area of 8000 km^2 (Walker, 1953). This figure was a map-based estimate derived from detailed surveys of the coastline between 1946–1955, which included aerial photography and quadrat sampling over an area of 270 km^2 (Walker and Richardson, 1956). Technological advances in scuba diving in the 1960s and 1970s facilitated stepwise progress in our understanding of the distribution and ecology of UK kelp forests. Perhaps most notable were the seminal body of work by J. Kain on the ecology of *Laminaria* on the Isle of Man (summarised in summarised in Kain, 1979) and P.G. Moore's work on faunal assemblages within kelp holdfasts in NE England (Moore, 1971, Moore, 1973). From the 1980s onwards, changes in attitudes and regulations concerning scientific scuba diving, coupled with shifts in research priorities, and relatively little commercial interest in kelp, led to a dearth of primary research on kelp forests in UK waters. Renewed activity in the 2010s has addressed some key knowledge gaps, although overall understanding of these systems remains poor compared to several other countries (e.g. USA, Australia). Recent research has focussed primarily on *Laminaria hyperborea,* relating to structure and environmental drivers (Smale et al., 2016, Smale and Moore, 2017), productivity (Smale et al., 2016, Pessarrodona et al., 2018), stipe and holdfast communities (Teagle et al., 2017, Teagle et al., 2018, Teagle and Smale, 2018, King et al., 2021), understorey flora and fauna (Bué et al., 2020, Earp et al., in prep., Smale et al., 2020) (Figure 1), microbiome (King et al., 2023), fish and commercially important species (Smale et al., 2011, Smale et al., 2022).

Kelp and canopy-forming fucoids are found along >20,000 km of the UK's complex and convoluted coastline (Smale et al., 2013). In terms of areal extent, estimates vary considerably from around 8,000 km^2 (Pessarrodona et al., 2018) to >20,000 km^2 (Yesson et al., 2015), with limitations in data availability (e.g. area of subtidal reef habitat) hindering predictive modelling approaches. Even so, marine forests formed by kelp and fucoids unequivocally represent one of the most widespread and ecologically important habitat types in the UK.

Key life history and distribution information for UK species is summarised in Table 1. In general, the large stipitate kelp *Laminaria hyperborea* dominates along open wave-exposed coastlines (Smale and Moore, 2017), stretching from the low intertidal to depths of 40 m or more in exceptionally clear waters (Smale et al., 2013). The sugar kelp, *Saccharina latissima*, tends to dominate in more sheltered environments such as embayments and lochs. Other kelp species present in the UK are *Laminaria digitata*, *Laminaria ochroleuca*, *Alaria esculenta*, the non-native Asian kelp *Undaria pinnatifida* and the pseudo-kelp *Saccorhiza polyschides* (technically a Tilopteridale). Important canopy-forming fucoids include *Fucus serratus*, *Fucus vesiculosus*, *Ascophyllum nodosum* and *Himanthalia elongata*. Fucoid species tend to be restricted to intertidal habitats but can form extremely dense and extensive canopies. These marine forest species each have distinct environmental requirements and different life history characteristics, which influence their ecological functions and predicted responses to environmental variability (Table 1).

1.5. Status of UK kelp forests

Kelp forests globally are under pressure from threats including climate change, pollution and over grazing (Harley et al., 2012, Smale et al., 2013). In the UK, there is evidence to suggest that the distribution and relative abundance of some kelp species has shifted in recent years (Smale and Vance, 2015, Smale et al., 2015, Pessarrodona et al., 2019), whilst others have seemingly remained stable (Burrows et al., 2018). Relatively limited reports of widespread kelp loss or local extinctions exist for the UK, in contrast to other regions including western Australia, the Iberian Peninsula, southern Norway and northern California (Moy and Christie, 2012, Voerman et al., 2013, Wernberg et al., 2016, Layton et al., 2020b, McPherson et al., 2021). That said, there is compelling evidence to suggest that the warm-adapted kelp *Laminaria ochroleuca* has proliferated at the leading range edge and extended northwards in recent decades, with potential consequences for local biodiversity and productivity (Teagle and Smale, 2018, Pessarrodona et al., 2019). More anecdotal evidence indicates that the cold-water kelp *Alaria esculenta* has declined in abundance towards its southern trailing edge in the southwest of the UK, and that *Saccorhiza polyschides* has increased in abundance, perhaps due to ocean warming and increased physical disturbance (Hiscock et al., 2004). Localised declines in abundance, biomass, and depth penetration of kelp in response to decreasing water quality have also been suggested. However, the evidence base to support these shifts remains limited. With projected ocean warming, the distribution and abundance of kelp species is expected to shift in line with individual species' thermal niches, with both climate 'winners' (e.g., *Laminaria ochroleuca*, *Saccorhiza polyschides*) and 'losers' (e.g., *Alaria esculenta*, *Laminaria digitata*) expected (Müller et al., 2009, Smale et al., 2013). Responses of intertidal fucoids may be more complex, owing to interactions between sea temperature, air temperature and grazing pressure (Hawkins et al., 2008). See Table 1 for a summary of predicted range changes for UK species.

There is an interest in restoration in some areas of the UK where kelp declines have been observed, however large-scale kelp losses have not been widely reported so there is not an obvious need for kelp restoration effort across much of the UK coastline. A recent survey of UK experts (Appendix 1) did not identify any locations of significant loss, although regional shifts in kelp forest structure were reported. Two regions where kelp restoration has been suggested in the UK are Durham and West Sussex. In Durham, coal mining waste was historically dumped on the coastline and is anecdotally reported to have impacted kelp forests. Following an extensive pollution clean-up operation there is local interest in kelp restoration measures (Moore, pers. comm., 2021). In Sussex, kelp forests are estimated to have declined by 95% since the 1980's (Mallinson, 2020). This is thought to be driven by multiple factors, including storm damage which thinned the kelp, allowing access by trawlers. The subsequent changes in fisheries practices, along with dumping of dredge spoil, are thought to have reduced water quality and increased sedimentation impacts (IFCA, 2022) (see Background of kelp restoration in the UK section, below, for further detail). Finally, following the storms of winter 2021/22, losses of *Laminaria hyperborea* forest have been reported from shallow waters around the Farne Islands and St Abbs, where small, localised urchin barrens have also been anecdotally observed (Moore pers comm., 2022, Figure 2). As intact healthy kelp forest populations remain nearby these sites, natural unassisted recovery is anticipated.

Figure 2. Localised urchin barrens in North East England. Credit DS.

2. Restoration Techniques

2.1. Global kelp forest restoration background and history

Ameliorate drivers of loss

The Society for Ecological Restoration (SER) defines restoration as "*the process of initiating or accelerating the recovery of an ecosystem that has been degraded, damaged or destroyed*". Management and restoration of marine systems is underdeveloped in comparison to terrestrial systems (Saunders et al., 2020, Wood et al., 2019), and until recently kelp forests had received little restoration attention relative to other coastal and marine ecosystems (e.g., mangroves, seagrass, saltmarsh, shellfish and coral reefs) (Saunders et al., 2020, Eger et al., 2021, Morris et al., 2020a, Smith et al., in press). Despite this, the emerging field of kelp forest restoration is growing rapidly. Recent research reviews have focussed on lessons learned, status and goals (Eger et al., 2021), key principles and best practices (Morris et al., 2020b), evaluating success (Earp et al., 2022) and future trajectories (Wood et al., 2019, Coleman et al., 2020). There is now a guidebook for kelp restoration, aiming to share lessons learned from global efforts (Eger et al., 2022b). This section of the report summarises information on a variety of restoration techniques compiled primarily from information within the aforementioned reviews.

A discussion of kelp restoration should first acknowledge that prevention is better than cure, and priority should be given to monitoring, managing and conserving existing kelp habitats. Despite this, or perhaps due to increasing local and regional stressors, kelp forest restoration is becoming increasingly commonplace, with the goal of reinstating self-sustaining populations in areas where declines and/or losses have occurred. Critical to restoration success is prior consideration of the original driver of decline, with actions to eliminate or mitigate the stressor(s) likely to be a prerequisite to further restoration efforts. Restoration methods should be appropriate for the scale of degradation, the target restoration species, and local environmental conditions. They should also be accompanied by robust monitoring programmes to evaluate success and inform ongoing action. The overarching goals of restoration efforts are to restore biodiversity and ecological functioning, offset the impacts of coastal development, address scientific questions, or to improve ecosystem service provision (Hagger et al., 2017).

Active and passive techniques

Passive restoration, also referred to as assisted recovery, involves action to ameliorate the stressor(s) that caused population decline (Morrison & Lindell, 2011, Boström-Einarsson et al., 2020; Layton, 2020), or the installation of suitable substrate for colonisation (Earp et al., 2022 and references therein). In the case of marine forests, passive restoration refers to actions taken on the surrounding environment, rather than afforestation by direct manipulation of individual seaweeds themselves. Examples include improving environmental conditions, creation of artificial reef habitat for colonisation, and removal of herbivores or competitors (the latter two argued by some to be active interventions due to the level of effort required e.g. Eger et al., 2021). Passive methods either rely on natural processes to supply propagules from nearby healthy marine forest populations, or must be used in combination with active techniques where natural sources of propagules are absent.

Recovery using passive means is highly variable, site-dependent, may take place over extensive time periods, and is not guaranteed (see Campbell et al. 2014 for an example). Layton et al. (2020b) highlight only one example globally - Californian giant kelp (Reed et al., 2006) - in which long-term restoration success was achieved from passive restoration alone. Lack of recovery may be due to changes in environmental conditions (i.e., abiotic – ocean warming, or biotic - herbivore abundance), the absence of nearby mature populations to supply propagules combined with low dispersal capability of many forest forming species (Johnson and Brawley, 1998, Parada et al., 2016, Filbee-Dexter and Wernberg, 2018), and the stabilisation of alternative states such as urchin s or turf algae communities by positive feedback loops (Filbee-Dexter and Wernberg, 2018, O'Brien and Scheibling, 2018).

In cases were mitigating the driver of decline is insufficient to promote natural recovery, greater interventions may be required. Active kelp restoration techniques aim to increase the number of individuals, either by transplanting adults or juveniles from healthy donor populations or laboratory cultures, or by seeding areas with spores. These approaches are considered to be more successful by some authors (Layton et al., 2020b), and can be effective more quickly than passive techniques (Zahawi et al., 2014). Regardless of technique, the ultimate aim of restoration activities is that kelp populations become self-sustaining rather than being maintained through ongoing restoration efforts, which would be costly, impractical, and potentially detrimental to donor populations.

There is no ‘one size fits all’ solution for kelp restoration, and no single method will be optimal for all situations, meaning any proposed restoration work will need to be assessed on a case-by-case basis within its environmental context. Many restoration programmes to date have utilised multiple techniques simultaneously in the hope of increasing success, and restoration method flow charts or decision-making frameworks now exist (Eger et al., 2022b, Cebrian et al., 2021, Smith et al., in press).

History

The earliest recorded kelp restoration projects were conducted in the 1700’s in Japan, utilising passive techniques such as controlling kelp harvesting, improving water quality, or protecting predatory species which limit grazing activity and facilitate forest formation (Eger et al., 2021). Over the past several decades, increasing attention has been given to active restoration techniques, for example the mass culturing and release of juvenile kelp in California (North, 1976). In the past 20 years, the frequency of marine forest restoration studies has continued to increase and has been undertaken in at least 15 countries (Figure 3), most notably Japan, the USA (specifically in California), and Australia (Eger et al., 2021). A recent meta-analysis of 63 restoration studies (Earp et al., 2022) found that restoration efforts were concentrated in the North Pacific (20% in the north-east and 19% in the north-

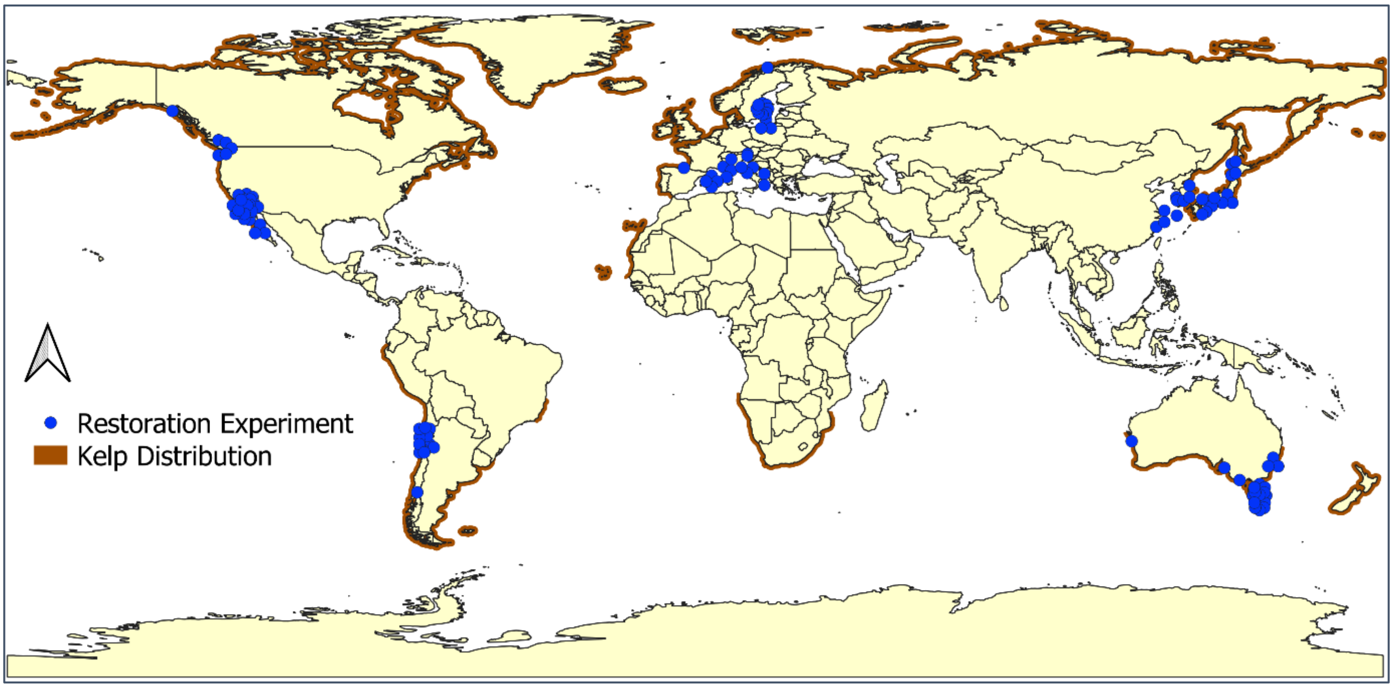

Figure 3. Global distribution of kelp forests and marine forest restoration projects. Adapted from Earp et al. (2022).

west), the South Pacific (12% and 29% respectively in the south-east and south-west), the Mediterranean Sea (13%) and the Baltic Sea (7%). Further information on the history of restoration is detailed by country in Eger et al. (2021).

Despite supporting extensive kelp dominated systems, few to no restoration studies have been conducted from the UK, South Africa and New Zealand (Earp et al., 2022, Eger et al., 2021). This may represent a historic lack of research interest from which to detect declines, limited degradation and thus need in these regions, a lack of motivation to fund or engage in restoration projects, a lack of access to restoration information or a lack of reporting of unsuccessful restoration efforts (Eger et al., 2021, Earp et al., 2022). Restoration may not always be feasible, for example due to species life history (slow growth rates e.g. *Lessonia trabeculata*) inaccessible distribution (e.g. deep water *Laminaria rodriguezii*), while successful management strategies such as commercial harvest limits and protected areas may eliminate the need for restoration (e.g. South Africa (Anderson et al., 2007).

In terms of taxonomic focus, to date the majority of restoration activities have investigated *Macrocystis pyrifera*, reflecting attention on Californian kelp habitats since the late 1950's, followed by the southern hemisphere kelp genus *Ecklonia* (Morris et al., 2020b, Eger et al., 2021, Earp et al., 2022). Given the habitat preference of these two species, the majority of restoration efforts have been focussed on subtidal habitats, with relatively few attempts made to develop techniques for intertidal populations, probably owing to the complex and dynamic variations in physical conditions of this habitat (Yu et al., 2012). Generally, restoration efforts to date have been restricted to relatively small, experimental scales both spatially (i.e. <100m^2) and in duration, with Earp et al. (2022) finding that 85% were conducted for ≤12 months (Morris et al., 2020b, Earp et al., 2022, Eger et al., 2021). The majority of these efforts have been research studies focussing on one species within a particular site, region, or environmental context. Monitoring is often limited in scope and duration, focussing on abundance or morphological traits, rather than self-sustainability and biodiversity of the associated community, which are more indicative of success at the ecosystem level (Ruiz-Jaen and Mitchell Aide, 2005) (Eger et al., 2020b, Earp et al., 2022). As such, efforts have been primarily led by academics (with the exception of projects in Japan), which perhaps reflects the emerging nature of marine forest restoration techniques, and the challenges of scale (Eger et al., 2021).

While restoration success is variable spatially and taxonomically, effective kelp forest restoration has been demonstrated at experimental scales (based on both authors inference and an objective meta-analysis), with success rates often ≥50% (Earp et al., 2022).

2.2. Kelp restoration in UK waters

No restoration experiments conducted in the UK were found in the published literature. This may be due to limited kelp declines by comparison with other regions and therefore limited restoration effort, or lack of reporting of unsuccessful restoration attempts (see History above).

To date, limited kelp losses have been documented in the UK. Instead, shifts in species composition have been recorded, often resulting in subtle ecological impacts, but such shifts are less conspicuous than widespread losses. Two notable exceptions are the Durham coast and the Sussex coast (see Section 1 - UK context, and Appendix 1).

In 2021, trials were conducted using the "green gravel" technique to seed the Sugar Kelp, *Saccharina latissima,* by Newcastle University. The Durham coastline was considered suitable for restoration efforts because kelp losses here were driven by historic dumping of coal mining waste which has since been the subject of clean-up operations. Kelp was successfully seeded onto rocks of two sizes (gravel and cobbles), reared in the aquarium, and deployed on the low intertidal at wave exposed sites. These were monitored over a period of 8-months in the field, with cobbles generally retained better than gravel, however the research was significantly impacted by storms and sedimentation (Earp et al. unpublished). Similar follow-up research is currently underway to repeat this work along a wave exposure gradient in Plymouth (Wilding pers. comm., 2022).

The only other known site of kelp restoration interest in the UK is the Sussex coastline. To exclude the physical abrasion caused by towed fishing gears in areas of sensitive habitat and known historical kelp beds, as well as reduce sedimentation and improve water quality, the Nearshore Trawling Byelaw was developed and implemented by Sussex Inshore Fisheries Conservation Authority (IFCA) in 2021, which restricts trawling activity from ~200 km^2 of nearshore habitat. The Sussex Kelp Recovery Project are now leading on a range of monitoring activities, including towed video surveys, potting surveys, eDNA and BRUV surveys, and support several researchers and groups studying related topics, to examine whether amelioration of these stressors will facilitate natural recovery of habitat forming kelp species (primarily *Saccharina latissima* but also some *Laminaria hyperborea* and *Laminaria digitata*), or if in the longer term active restoration may be required.

2.3. Kelp restoration techniques

Restoration techniques can be broadly grouped into four approaches: transplants, seeding, grazer management, and artificial reefs (Eger et al., 2022b). These are summarised and evaluated in Table 2 and discussed in turn here.

Transplants

Translating involves deployment of adults or juveniles from donor populations, lab cultivated specimens, or opportunistically collected beach cast individuals into denuded areas with the aim that they will attach to new or existing substrate by holdfast growth. This technique may be applied in either intertidal (Correa et al., 2006) or subtidal (Campbell, 2014) habitats. There are various methods for securing transplants to the benthos, including the use of glue, epoxy putty, cable ties, rubber bands, chains, bolts or ropes, to secure transplants onto hard substrates such as concrete blocks, plastic mesh matts, shells, ceramic tiles, longlines, string, gravel, artificial substrates or existing holdfasts (Earp et al., 2022, Eger et al., 2021) (Figure 4). Recently, this method was used successfully to install the kelp *Ecklonia radiata* onto artificial substrates in Australia (Layton et al., 2020b). While various materials for attachment were tested, recycled rubber bands were found to be most effective. *E. radiata* is a stalked, or "stipitate" kelp, which is morphologically similar to several UK species (*Laminaria hyperborea, L. ochroleuca, L. digitata*), so this method may be transferable into a UK context, but has yet to be trialled.

A key advantage of this method is that the presence of adult transplants immediately begins to alter environmental conditions, creating a suitable habitat for growth of new recruits or settlement of propagules (Layton et al., 2019, Morris et al., 2020b). This has led to suggestions that it may be an important "first step" towards self-sustaining restored populations (Eger et al., 2021). Transplants also allow a high degree of precision in placement and control of canopy density (Graham et al., 2021), in contrast to seeding methods (discussed below). Limitations of this method include impacts on donor populations, limited re-attachment of individuals to the seabed, and scalability. Moreover, the process of transplanting is labour intensive, often involving SCUBA divers, and may be prohibitively expensive to employ at large scales (Eger et al., 2021).

In recent reviews, both Earp et al. (2022) and Eger et al. (2021) identified transplanting as the most commonly used restoration technique globally, with a long history of use. While considered to be one of the most successful methods by Morris et al. (2020b), Earp et al. (2022) found lower survival rates of transplanted individuals compared to those in natural forests, suggesting that re-attachment may be limited (Correa et al., 2006). Although poor survival rates are clearly undesirable, if a sufficient number of transplants live long enough to reproduce and seed the restoration area, then the outcome of restoration can still be considered to be positive (e.g.(Campbell et al., 2014).

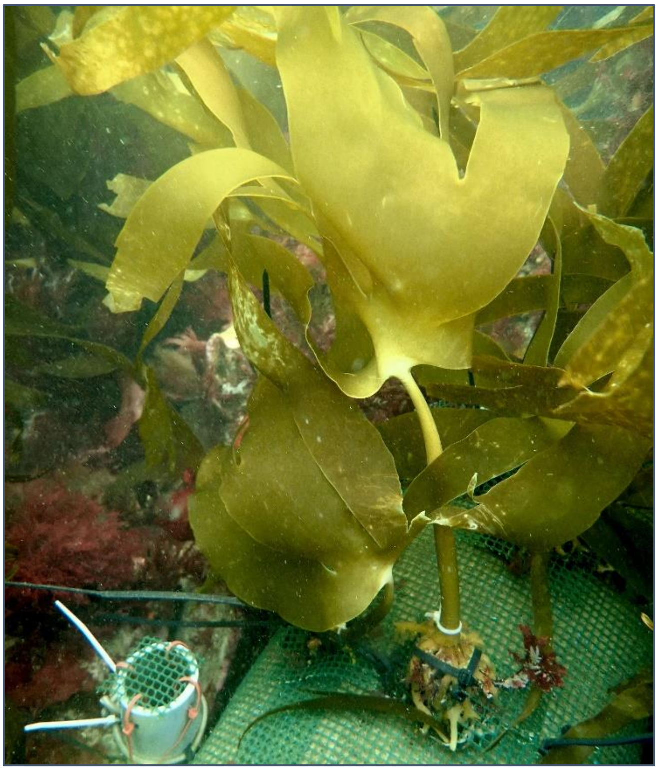

Figure 4. Juvenile Laminaria ochroleuca *and* Laminaria hyperborea *transplanted onto sand bags. Credit DS.*

Green gravel

In essence a type of transplant (Earp et al., 2022, Eger et al., 2021), a promising new restoration techniques termed "green gravel" has recently been developed (Fredriksen et al., 2020). Small rocks are seeded with kelp and cultured under optimal conditions in aquaria prior to out-planting or deployment at sea (Figure 5). The technique has been successfully trialled with *Saccharina latissima* in Norway, and is considered highly scalable as, although lab costs are high, gravel can be deployed in large numbers from boats avoiding labour intensive and costly SCUBA diver installations. Boat based deployment also allows potentially inaccessible underwater habitats to be restored. Smaller substrates are also easier to handle and transport than mature transplants or large artificial substrates, and the technique appears to be fairly robust (Fredriksen et al., 2020). The Green Gravel Action Group (www.greengravel.org) is currently testing this technique across an array of environmental contexts, kelp species and substrates to further understand its feasibility and limitations.

Furthermore, green gravel can be seeded with selected or engineered genotypes that are resilient to particular stressors (see Emerging and additional techniques section for more detail), then deployed into extant kelp forests across large scales. Such methods could feasibly be applied in the UK context.

Figure 5. Green gravel. Juvenile Laminaria ochroleuca *growing on white marble gravel in the lab. Credit CW.*

Seeding

Recruitment can be enhanced at restoration sites by releasing reproductive bodies, or by dispersal of laboratory grown early life stages. Mesh "spore bags", filled with reproductive material (called sorus tissue), are weighted and placed on the sea floor, to release propagules. Alternatively, microscopic juveniles generated from laboratory spore cultures are released (Yu et al., 2012, North, 1976, Vásquez and Tala, 1995). For example, the Seacare community group (Sanderson et al., 2003, Layton et al., 2020b) used a suite of methods including transplanted sporophylls and ropes seeded with lab-cultivated juveniles to attempt to restore giant kelp (*M. pyrifera*) in Tasmania, although success was limited (Layton et al., 2020b).

Seeding techniques remain relatively undeveloped by comparison to transplants, probably due to the naturally high mortality rates of early kelp life stages (Schiel and Foster, 2006). Although, Eger et al. (2021) recently proposed adapting a method trialled for coral restoration whereby ships distribute propagules (Doropoulos 2019) to seed areas with kelp spores. Seeding techniques can also be relatively labour intensive if divers deploy the spore bags, and empty spore bags would need to be subsequently collected unless they are biodegradable. The method is however, relatively scalable (Morris et al., 2020b), and in many kelp species, reproductive material can be collected without substantial damage to the donor plant.

Earp et al. (2022) identified seeding as a successful restoration method, particularly for Laminarian kelp species which represent key habitat forming species in the UK, although they noted that the finding was based on a small number of studies, with success more likely at sites where adult conspecifics are also present (Layton et al., 2019). Successful recruitment of *Macrocystis pyrifera* from spore bags appears to be dependent on clearance of competing understory algae (Hernandez-Carmona et al 2000). However, in some cases, seeding success has been limited (Westermeier et al., 2014). For example, (North, 1976), reported that only one in 100,000 released microscopic juvenile kelp attached to the seabed. Nonetheless, due to the potential for lower cost and high scalability (Saunders et al., 2020), seeding techniques remain promising, and are currently under development.

Artificial habitat creation

Creation of artificial reefs requires installing structures on the seabed which offer suitable substrate for kelp recruitment and growth. This method is not considered to be restoration in the true sense, but more afforestation, as rather than repairing an existing natural habitat it is replaced by a new artificial one (Eger et al., 2021). Artificial habitats may be comprised of natural or manmade materials (e.g. concrete), and may be either 'recycled' (ships or other vehicles), or new structures, the latter of which can be specifically engineered to promote kelp settlement (i.e. with increased rugosity, or infusion of the material with nutrients to improve growth) (Eger et al., 2021). Artificial habitats are also often used in conjunction with seeding or transplanting techniques (e.g. Falace et al., 2006).

Artificial reefs are often located where they are easy to maintain, and methods for attaching transplants incorporated into their design, make it simpler and cheaper to transplant onto them compared to natural reefs (Eger et al., 2021). They may also be engineered to suit local abiotic / biotic conditions (Morris et al., 2020b). For example, (Yu et al., 2012) reported high success by creating artificial cement pools, seeded with *Sargassum thunbergia* germlings, on natural intertidal reefs. With a history or use, and the potential to be applied at large scales, artificial habitat creation is thought to be one of the most common techniques used globally (Eger et al., 2021). The approach was used in 33% of projects reviewed by (Morris et al., 2020b), and can yield high success rates as a sole technique (Earp et al., 2022). However, creation of artificial reefs is costly and requires substantial investment (Eger et al., 2021). For kelp forests on artificial structures to become established, a supply of propagules is also necessary, either from nearby kelp populations, or from seeding/transplants. By comparison with other passive methods, creation of artificial reefs alone is generally less successful in achieving restoration outcomes (Layton et al., 2020b).

Notable kelp restoration projects utilising artificial habitats have been conducted in Japan, Korea, the USA and (unsuccessfully) Norway (Eger et al., 2021). In a well-known example, the Californian government mandated the creation of "Wheeler North Reef" to offset the loss of giant kelp forests caused by a warm water outflow from a nuclear power station (Reed et al., 2019). The project was well funded and included robust monitoring and evaluation by an independent team of scientists (see Best Practice – monitoring section). In this case, a failure of the restored area to meet several performance standards relative to a natural reef reference area was evident during the first 10 years (2009-2018) following installation, suggesting that artificial reefs are not always an adequate replacement for their natural counterparts (but see Reed 2019). Despite this, a combination of transplanting and artificial reefs in Tasmania allowed for the development of diverse assemblages, including economically and ecologically valuable lobsters and oysters (Layton et al., 2020b, Shelamoff et al., 2019).

Artificial reefs can also include nutrient enrichment, involving the release of nutrients to stimulate algal growth (e.g. Yamamoto et al., 2010). Structures can be impregnated with slow release nutrients to overcome nutrient limitation, or even to counteract grazing pressure (Eger et al., 2021). For example, Hayashi et al. (2011) created artificial habitat from a mixture of steelmaking slag and dredged soil. Dissolved iron within the slag was linked to increases in the biomass of two large brown seaweed species compared to those from adjacent areas.

Ecological engineering

"Greening" or "ecological engineering" of artificial marine infrastructure can involve modifying the structural complexity, heterogeneity (i.e. by integration of artificial rockpools) or rugosity (Chapman and Underwood, 2011) in order to improve ecological value. These actions can mitigate the environmental impact of infrastructure installation (Perkol-Finkel and Airoldi, 2010, Morris et al., 2020b), and benefit ecosystem service delivery (e.g. coastal erosion etc. Morris, 2020).

When used in parallel with transplants, this has been found to improve seaweed recruitment (Perkol-Finkel and Airoldi, 2010). Incorporating green engineering into restoration strategy could allow the costs associated with restoration to be shared with coastal developers, if biodiversity improvements through eco-engineering or creation of new artificial structures were necessitated by the permissions process as a condition of licence for development. Green engineering is currently being explored for bivalve restoration through The World Harbour Project (www.worldharbourproject.org/bivalve-restoration), which aims to develop eco-engineering techniques for enhancing bivalve populations.

Herbivore removal

Exclusion or removal of grazers from restoration sites involves practices that remove the target species, or the installation of exclusion devices. Removal of urchins can be achieved by collecting and relocating them, harvesting them for market, or culling, either manually by crushing or with quicklime (Eger et al., 2021). Exclusion devices can include cages, fences, electric fences, chemical barriers such as copper paint or antifoulants, bubble curtains, or the use of plastic kelp mimics to exclude urchins by the abrasive sweeping motion of the fronds (Sharma et al., 2021, Earp et al., 2022).

These techniques have generally been focussed on areas with intense urchin grazing, where there is a risk that marine forests can shift to an alternative stable urchin barren state (Filbee-Dexter and Scheibling, 2014), although in parts of Japan, California and Australia herbivorous fish have also been targeted (Vergés et al., 2014).

Urchin management is more scalable than transplanting, has a history of use (Morris et al., 2020b), and coupled with competitor removal has been identified as the most successful restoration technique reviewed by Earp et al. (2022). Research in Australia found that urchin removal in small areas allowed for recovery of *E. radiata* kelp forests which supported similar communities to natural kelp habitats (Ling, 2008). However, the technique is labour intensive, requiring ~13 dives per hectare (Tracey et al., 2015b). Urchin removal rate is dependent on local conditions such as topography and urchin density, although can be made more efficient by baiting urchins into dense congregations (Eger et al., 2021).

A promising solution in regions affected by urchin grazing is a market-based approach, whereby a fishery is established for urchins, which are sold for human food (e.g. "umi" in Japan). Although urchins from barrens are generally of poor nutritional quality, they can be improved by feeding an aquaculture diet prior to market (www.urchinomics.com, Pert et al., 2018). These approaches are being trialled in Japan, California and Norway, where they generate employment in coastal communities, and may encourage further restoration/conservation efforts by increasing the perceived value of kelp forests (Eger et al., 2021). Despite being proven as a successful restoration technique, in order to be viable over meaningful time scales, grazers must continually be removed, presenting an ongoing cost. Research has shown that urchin abundance must be maintained at densities of less than 1.5 individuals m^2 (Ling et al., 2009). Also, long term success is reliant on a supply of propagules from mature kelp populations.

Excluding sea urchins from barren areas has been found to successfully restore kelp forests and can reverse phase shifts (Ling et al., 2019), although is context specific, as small (30cm high) fences were less effective within kelp forests than in barren areas (Sharma et al., 2021).

The use of quicklime (CaO) over urchin barrens is less resource intensive so can improve scalability, but would not be suitable in the UK and so is not discussed here in further detail – it should only be considered for barrens or similarly impoverished systems, due to the risk of wider ecological damage (see Strand et al., 2020).

While the UK is not currently known to have widespread urchin barrens (Burrows et al., 2014) there are some reports of elevated urchin densities in the kelp forests of northeast England / southeast Scotland with small localised areas denuded of kelp (Moore pers. comm., 2022. Figure 2). Given that urchin removal has been a successful restoration in many other regions, it could also be beneficial tool for protecting kelp forests in the UK should urchin numbers increase in future.

Competitor removal

The removal of species such as turf algae, which would complete for resources with forest forming species, is a restoration technique often used in conjunction with techniques such as seeding or transplanting. Competitor removal methods have been primarily developed in Japan (Eger et al., 2021), and include labour intensive manual removal of competing algae, or mechanical techniques such as chains anchored to the seabed which are spun by wave motion to dislodge competing algae (Eger et al., 2021). While Morris et al. (2020b) consider these methods to be applicable at medium (1000-100,000m^2) scales, Eger et al. (2021) consider it only applicable to small scales.

Along with herbivore removal, this method has been identified as the most successful restoration technique reviewed by Earp et al. (2022). However, as with other passive methods, long term success is reliant on a supply of propagules from mature populations in close proximity, resulting in many restoration efforts using a combination of this and active techniques (Earp et al., 2022).

Furthermore, there can be benefits (e.g. facilitation cascades) to co-restoration of multiple marine forest species (discussed in Eger, 2020), such as enhanced recruitment or reduced environmental stress. These are context specific, therefore while removal of turf algae is likely to benefit canopy formers, suppressing a range of assumed competitor species may be detrimental.

While small scale competitor removal may be used in conjunction with other techniques in the UK, widespread clearance of the seabed for kelp restoration is not likely to be feasible or ethical in the UK.

Emerging and additional techniques

Facilitative ecological interactions

As the rate of environmental change continues to accelerate globally, the corresponding increased focus on restoration is likely to facilitate the development of emerging techniques (Wood et al., 2019). This creates an opportunity to "play to the positives" by incorporating facilitative ecological interactions into restoration strategies (Eger et al., 2020a), or to include resilient genotypes, either by selective breeding or direct genetic manipulation, to restore populations that are tolerant to predicted future environmental conditions (Coleman et al., 2020, Layton and Johnson, 2021).

Positive intraspecific facilitation interactions are likely to become increasingly important under future conditions of increased environmental stress (Eger et al., 2020a). For example, modification of the physical environment by conspecific adult kelp can facilitate recruitment, growth, and survivorship of juveniles in conditions that would be otherwise unsuitable (Layton et al., 2019). The "whiplash" motion of adult kelp fronds may also deter urchins, thus reducing grazing and facilitating the growth of juveniles (Vasquez et al., 1998). This has led to the suggestion that restoration success could be enhanced by transplanting adults or selecting restoration sites near intact forests to reduce the impact of grazing species (Eger et al., 2020a).

While passive restoration approaches can potentially benefit a diverse array of marine forest species, active techniques have so far been monospecific or species-centric (Morris et al., 2020a, Earp et al., 2022). Methods which aim to restore multiple marine forest species could facilitate positive interspecific interactions, potentially enhancing success (Earp et al., 2022, Eger et al., 2020a). These interactions can be complex and must be fully understood – for example, the density of the *Ecklonia* canopy affects whether it has a positive or negative impact on *Sargassum* recruitment (Bennett and Wernberg, 2014) – but none-the-less have great potential. Increased diversity of canopy forming species may also offer greater ecological function and resilience (Graham et al., 2013), particularly if the reference community or goal of restoration is a multi-species seaweed assemblage (Graham et al., 2013).

Although not tested for marine forests, mixed species planting has been successful in both coastal wetland (Silliman et al., 2015) and seagrass habitats (Williams et al., 2017). These positive interactions can mitigate stressors, so are likely to become increasingly important under increasingly stressful environmental conditions (Eger et al., 2020a).

Co-restoration and ecosystem-scale approaches

While marine management rarely specifically targets forest-forming seaweeds (Woodcock et al., 2017), restoration goals may potentially be achieved by holistic management and conservation strategies, and co-restoration of organisms which facilitate marine forests.

For example, traditional management of kelp harvesting appears to maintain sustainable kelp fisheries in Japan, Chile and France (Buschmann et al., 2014, Frangoudes and Garineaud, 2015, Fujita, 2011, Wilding et al., 2021) (although see Gouraguine et al. (2021) on juvenilisation in Chile and Werner and Kraan (2004) on shifting species composition in France). Broader conservation management measures such as Marine Protected Areas (MPA's) can also be effective restoration methods (Eger and Baum, 2020). Eger et al. (2021) highlight that in New Zealand, healthy kelp (*Ecklonia radiata*) and wrack (Fucales) populations are now sustained inside the Cape Rodney to Okakari Point Marine Reserve, while adjacent areas outside the reserve are dominated by urchin barrens (Shears and Babcock, 2003).

In the UK, the Sussex "Help our Kelp" campaign (see Kelp restoration in UK waters section) aims to restore kelp habitats by managing fishing activities. Declines are thought to have been driven in part by damage from bottom trawling fishing gear and resultant increases in sedimentation, so a ban on trawling along a section of coastline was introduced in March 2021, and recovery is being monitored.

Co-restoration of marine forest species alongside organisms that facilitate their survival (Caselle et al., 2015, Lester et al., 2009, Ferrari et al., 2018), for example of lobsters and sea otters that predate on kelp grazers (Estes and Duggins, 1995) is a promising approach in some regions. Although requiring further investigation into feasibility at scale (Eger et al., 2020a), such co-restoration could allow positive facilitation cascades, promoting resilience in kelp ecosystems (Eger et al., 2020a). Measures to conserve predators can include protective policies (Estes and Duggins, 1995), reintroduction programs (Marzloff et al., 2011, Eger et al., 2020a), or incorporation into marine reserves and no-take MPAs (Eger and Baum, 2020, Shears and Babcock, 2002). While not always successful for kelp restoration specifically, these methods are likely to have wider benefits and be more affordable than active restoration, so should not be overlooked, particularly in ecologically simple systems (Eger et al., 2020a).

Marine forest restoration can also be achieved through improvements to water quality (Hawkins et al., 1999, Foster and Schiel, 2010). Treatment of wastewater, by removing suspended solids or denitrification of sewage outflow, allowed for recovery of intertidal fucoid *Hormosira banksii* in southern Australia (Bellgrove et al., 2010). This method was found to be an effective yet somewhat overlooked technique by Earp et al. (2022), employed in only two experiments but with a success rate of 100%. Despite this, passive methods such as improving water quality are not always sufficient to result in natural forest re-establishment, likely due to limited propagule supply or early post-settlement mortality (Wernberg et al., 2019b, Campbell et al., 2014), as was the case for *Phyllospora comosa* in Sydney (Verges et al., 2020) (See case-study in Best Practice section).

"Future-proofing" – genetics and microbiome

Microbiome manipulation has been identified as an emerging approach which could potentially enhance kelp resistance or resilience to future environmental conditions (Wood et al., 2019, Eger et al., 2020a, Li et al., 2023). The microbial community on the surface of seaweeds can influence seaweed development, growth, photosynthesis, and reproduction (reviewed by Egan et al., 2013) and may even effect interactions between seaweeds and their grazers or epiphytes (Campbell et al., 2014, Marzinelli et al., 2018). This has led to the suggestion of, for example, inoculating early kelp life stages with beneficial microbial taxa prior to out planting as a restoration technique (Eger et al., 2020a).

There is growing consideration of developing restoration beyond simply recovering what has been lost, to creating "future-proofed" populations that will persist in the face of changing environmental conditions (Coleman and Goold, 2019, Coleman et al., 2020, Wood et al., 2019, Wood et al., 2020). Experiments are necessary to identify suitably-adapted genotypes (Coleman et al., 2020) or highly tolerant species that perform similar ecological functions to the species that has been lost (Wood et al., 2019, Coleman et al., 2020). These are particularly pertinent in cases where it may not be possible to ameliorate the driver of decline within the necessary time frame (i.e. ocean warming), which can otherwise result in unsuccessful, wasted restoration efforts (Layton et al., 2020b).

In Tasmania, where kelp populations have declined by over 95% due to ocean warming, a pioneering project has trailed a technique to develop "super" strains of Giant Kelp *Macrocystis pyrifera* (Layton et al., 2020b). Surviving *M. pyrifera* individuals were selected as donors from which reproductive material was non-destructively collected and used to initiate self-replenishing gametophyte cultures in the lab. Following experimentation to identify warm-water tolerant lineages, field trials out-planted lab grown juveniles at restoration sites. While longer term monitoring is vital in evaluating success, early results from this work are encouraging.

The risk of introducing novel or exotic genotypes necessitates further considerations (Coleman and Goold, 2019, Wood et al., 2019), as for example selection for thermal tolerance can increase vulnerability to grazing pressure (Coleman and Goold, 2019), these emerging technologies have the potential to boost resilience to predicted environmental conditions (Coleman et al., 2020). As widespread kelp declines have yet to be recorded in the UK, development of tools which provide for anticipatory actions to confer improved adaptive capacity represent a promising opportunity for UK marine forest habitats.

Synergies with anthropogenic activities

Finally, potential synergies between human activities could be incorporated into future restoration planning. The licensing process for coastal development could legislate the creation of artificial reefs to offset biodiversity loss due to development, similar to the "Wheeler North Reef" case in California (see 'Artificial Habitat Creation' above).

Kelp aquaculture could provide a source of propagules, restoration technology and selected strains, while simultaneously providing biomass which can reduce harvesting pressure on wild populations. Optimisation of cultivation processes to provide transplants for restoration has already begun for certain species, such as *E. radiata* (Suebsanguan et al., 2021).

Other forms of aquaculture could benefit from nutrient cycling service provision by healthy local kelp forests, motivating them to part-finance restoration activities (Eger et al., 2020a). Lastly, market based solutions to urchin grazing (see 'Herbivore removal' above) can incentivise restoration while contributing to coastal economies (Eger et al., 2020a).

Factors influencing restoration success

No one restoration technique is applicable to all situations, with environmental conditions (abiotic and abiotic), the impact of global stressors which are difficult to mitigate for, and socio ecological context all potential influencers of restoration success. While some of these factors are well understood for terrestrial systems, they have often yet to be investigated in detail for marine forests (Morris et al., 2020b).

Abiotic conditions

Clearly, the nature of the environment at the restoration site will impact the most appropriate restoration technique. For example, areas subject to strong currents or wave energy are unlikely to be suitable for green gravel, as light weight gravel will be easily displaced. If green gravel is displaced from the monitoring area, assessment of success will be impossible, as their ultimate fate (i.e. whether the kelp form an attachment to substrate in a nearby habitat or reach maturity and contribute to the supply of propagules) will remain unknown. Work is currently underway to assess the effectiveness of this technique across a range of global environmental conditions through the Green Gravel Action Group (GGAG). By contrast, transplants on bedrock or heavy artificial substrates have been found to withstand strong swell, wind and currents (Layton, 2021), although some biodegradable transplant mats are not sufficiently robust for shallow, high-energy environments (Vergés et al., 2020).

Depth and water turbidity or quality are also important factors, as they influence light availability and in turn kelp growth. Along a gradient from high to low light availability, there is likely to be a "facilitation gradient", in which a shift from increased dependency of juveniles on the presence of adult conspecifics to reduce high irradiance, towards inhibition of growth due to competition for light resources occurs (Layton et al., 2019 and references therein). This has implications for interaction between depth and the target density of restoration transplants (see 'Patch dynamics', below).

Temperature and other physical factors are also likely to impact restoration success.

Ocean warming

In addition to local environmental conditions, global stressors will also influence restoration outcomes. The impact of ocean warming can have mixed effects on marine forest formers, either driving declines and impeding restoration efforts (Wernberg et al., 2016, Vergés et al., 2016, Qiu et al., 2019) or facilitating them (Filbee-Dexter et al., 2019). One positive example is the partial natural recovery of *L. hyperborea* and *S. latissima* forests in mid-Norway over the past ~20 years. Here, warming has reduced the larval survival of the cold adapted urchin *Strongylocentrotus droebachiensis*, and also is likely to be the driver of a northward range expansion of an urchin predator – the edible crab *Cancer pagarus* (Christie et al., 2019), with both factors facilitating kelp recovery. However, the relationship of ocean warming with multitrophic interactions is complex, having various impacts on urchin recruitment and kelp recovery along a regional latitudinal gradient, interacting with substrates which function as refuge for sea urchins, and the abundance of a possible predator of crabs, the coastal cod *Gadus morhua*. This demonstrates the need for a detailed understanding of interactions between local environmental conditions and global stressors prior to undertaking restoration work.

Grazing

Herbivory is spatiotemporally variable, and can impact heavily on restoration success (Carney et al., 2005). Although urchin exclusion is not always necessary for transplant success (Graham et al 2021), in cases where small or sparse patches of kelp are transplanted, they can quickly become overgrazed. This can be mitigated by transplanting large patches at high density, saturating the herbivore population and diffusing grazing pressure (Grant J, 1982, Morris et al., 2020b, Eger et al., 2021), by protecting juvenile transplants with exclusion devises like cloches or meshes (Whitaker et al., 2010, Carney et al., 2005), by transplanting larger individuals, which are less susceptible to herbivory (Lubchenco, 1983), or simply by focussing restoration efforts at times and locations where grazers are naturally less active or abundant (Eger et al., 2021). Increasing transplant density can also enhance chemical communication between individual plants, allowing chemical cues released in response to grazing to reach nearby conspecifics, stimulating the production of defensive compounds, thus reducing the impact of grazing (Toth and Pavia, 2000, Rohde et al., 2004).

Patch dynamics (size and density)

There is clear evidence of the importance of patch dynamics to restoration success. Minimum patch sizes of adult transplants are necessary to maintain the marine forest canopy, which reduces water flow, sedimentation, and irradiance in the sub-canopy, affecting recruitment, growth and survival (Layton et al., 2020b, Layton et al., 2019). Large, dense patches of kelp may also be more resistant to grazing pressure (see 'Grazing', above), and competition from turf algae (Reeves, 2018).

The density of adult canopy formers also influences population structure and recruitment patterns, albeit to a lesser extent. Overall recruitment may benefit from variability in adult canopy density, or even be negatively affected by high density through light limitation (Layton et al., 2019). This has led to recommendations that a "mosaics of canopy densities" may improve overall habitat resilience, by facilitating ontogenetic shifts in habitat requirements (Layton et al., 2019), which should be a consideration for restoration.

Layton et al. (2019) found that for transplanted *Ecklonia radiata*, a minimum patch size ($2m^2$) and kelp density (> 15 kelp per m^2) was necessary, and that recruitment increased with increasing patch size. Furthermore, when the density of adults was lower, larger minimum patch sizes were necessary to achieve restoration success. High densities of adults resulted in the greatest recruitment, suggesting that this would be an optimal approach for initiating restoration.

Species

Not all restoration techniques have been tested on all species, for example transplants have mostly been employed using *Macrocystis* spp. (Hernandez-Carmona et al. 2000; Westermeier et al. 2016), *Lessonia* spp. (Westermeier et al. 2016), and *Ecklonia* spp. (Reeves, 2018, Layton et al., 2021). In terms of UK species, limited work has been conducted using *S. latissima* (Fredriksen et al., 2020) and *Laminaria* spp. in Norway (Eger et al., 2021).

Technical specifications of methods will also vary by species. Layton et al. (2021) found that when transplanting *E. radiata*, materials including cable ties, wire and rope were not sufficient to allow holdfasts to re-attach to the substratum, and rope often resulted in further damage to transplanted kelp stipes/holdfasts. However, the authors also noted that rope methods have been used successfully with other species (Fejtek et al., 2011). Further, while holdfast re-attachment is important in transplanting some species, it may not be for others. Bouyant, floating kelps such as *Macrocystis pyrifera* appear to be more resilient to detachment than non-bouyant species, with *M. pyrifera* successfully transplanted without re-attachment (North, 1976, Layton and Johnson, 2021).

Age class

Generally, it appears that restoration of a mixed-age class population is more likely to be successful compared to single age class restoration, presumably because adults supply spores, ameliorate abiotic stressors (e.g. irradiance, water flow, sedimentation) and can either dilute herbivory or deter grazers with the sweeping action of the blades (Layton et al., 2020b, Layton et al., 2019, Reed and Foster, 1984, Toohey et al., 2004, Wernberg et al., 2005). Indeed, transplanting juveniles into areas where adult conspecifics were absent has resulted in significant mortality (Layton et al., 2019).

Very high mortality of microscopic life stages is natural given the 'low cost-high volume' reproductive strategy of kelp (Schiel and Foster, 2006) but is thought to limit the success of seeding methods. Survival of out-planted microscopic sporophytes is also very low, with rates of 353 in 2,500,000 (Layton et al., 2019) and one in 100,000 (North, 1976) reported.

When transplanting, the age of selected individuals appears to have a large impact on success. There may be an age class "sweet spot" for transplant survival, somewhere between small juveniles which acclimate more quickly, and larger individuals that may survive better post-transplant disturbance (Fowler-Walker et al., 2006). (Layton et al., 2021) found greater survivorship of juvenile kelp, along with greater ease of transplanting, compared to adults. Even within adults, smaller individuals showed greater post-transplant re-attachment rates and survival than larger ones. This was attributed to greater developmental plasticity and vigour in younger plants to adapt to new environmental situations, in addition to the practical difficulty of effectively re-attaching large adult specimens without damage (Layton et al., 2021). These findings have been replicated elsewhere (Graham et al 2021), although the magnitude appears to vary across transplanting techniques (Reeves, 2018).

Season

Spatial factors may also vary seasonally. In some areas seasonality has not been found to affect kelp restoration success (for example *E. radiata* transplanted by Layton, 2021), while others have reported high seasonal variation in restoration success (Correa et al., 2006), particularly in areas subject to seasonal variation in herbivore abundance or activity (Carney et al., 2005). For giant kelp (*M. pyrifera)* transplanted onto holdfast stumps of *Eisenia arborea* in two different seasons, survival was greater in winter (41%) compared to spring (7%) (Hernandez-Carmona et al 2000). There is also an interaction between season and patch dynamics, (Layton et al., 2019) with canopy shading improving *E. radiata* transplant success during summer when irradiance is high and at times stressful (Layton et al., 2019). However, in winter when irradiance is lower, the canopy lowers survival and reduces maturation rates, presumably due to light limitation (Graham et al., 2021).

Extent of decline and connectivity

In restoration work globally, the extent of decline has varied from ~95% for *Macrocystis* in Tasmania to 10-30% in Korea (Eger et al., 2021), which clearly affects the magnitude of restoration effort needed. Increasing proximity to existing natural populations is an important predictor of success (Eger et al., 2020a, Layton et al., 2020a). Eger et al. (2020b) reported this pattern in areas with remnant kelp populations from Japan, Norway, and California. Conversely, failure to restore at ecologically meaningful scales was reported from several regions which lack healthy remnant populations nearby (Eger et al., 2021). This may be particularly important for passive restoration techniques, which are unlikely to be successful without sufficient propagule supply to allow for natural recovery.

Given that phase shifts from marine forests to alternative stable states are often reinforced by positive feedback loops, meaning they are difficult to reverse (Filbee-Dexter and Wernberg, 2018) (although see Christie et al., 2019), there is clear evidence that restoration at sites where marine canopy formers have declined but not disappeared completely is likely to be both more cost-effective and more successful (Eger et al., 2021), highlighting the need to be prepared to take early interventions in the UK.

Socioeconomic context

Engagement with coastal communities is likely to enhance restoration outcomes, generating increased marine stewardship that can potentially result in increased funding or investment for restoration projects (Morris et al., 2020b). Good examples of engagement in marine forest restoration include the "Help our Kelp" initiative in the UK, "Get Inspired" in the USA, and " Operation Crayweed" in Australia (Morris et al., 2020b).

Restoration methods evaluation

Direct comparison between methods is difficult, due to a lack of standardisation in monitoring and reporting, and few investigations which directly compare across environmental contexts, species or life stages (but see Westermeier et al., 2016, Graham et al., 2021). Furthermore, the use of multiple techniques is common, which makes it challenging to isolate and evaluate the success of individual techniques (Earp et al., 2022, Eger et al., 2021). The most appropriate technique, and the likelihood of success are dependent on local conditions, species, and the scale of degradation and/or loss, so a case-by-case assessment is required to identify the most promising approach(es).

There is a bias in the literature towards author inference of success (generally found to be ≥50% by Earp et al. 2022), reflecting limited reporting of restoration 'failures' despite these being equally valuable in terms of drawing comparative assessments and progressing the field (Earp et al., 2022). Further, what is perceived as success in the literature does not necessarily correspond to ecologically meaningful, long-term recovery. Survival as a measure of success can be misleading for small-scale projects, particularly when the scale of degradation is much larger. Defining success in ecosystem restoration is therefore challenging (Eger et al., 2021), and a true "before and after" design is rarely possible because restoration is usually enacted in response to declines. To address this, Earp et al. (2022) performed a literature review and meta-analysis, finding variation in restoration success across countries, taxa and restoration technique. This section draws on this research and other key reviews by Morris et al. (2020b), Eger et al. (2021).

From the meta-analysis, active techniques were found to be more widely used than passive techniques (75% and 25% respectively), despite results showing that the latter has a higher success rate (active=65%; passive=80%) (Earp et al., 2022). Transplanting was the most commonly used restoration technique (Earp et al., 2022), and is also considered one of the most successful (Morris et al., 2020b) despite low survival rates (Earp et al., 2022). Seeding represented a small proportion of studies but was found to be highly successful for Laminarian species, although less so for Fucoids (Earp et al., 2022). Success rates were improved by the presence of adult conspecifics (Layton et al., 2019). Herbivore exclusion was found to have success rates exceeding 80% for both Laminarian and Fucalean species (Earp et al., 2022).

The two most successful methods reviewed by Earp et al. (2022) were competitor exclusion and grazer control, although sample sizes were small. There is conflict in the literature, with some authors (e.g. Layton et al., 2020b) considering active measures to be more successful than passive, in contrast to the findings of Earp et al. (2022).

All passive techniques are reliant on a supply of propagules for populations to become self-sustaining. Therefore, combining passive with active restoration techniques is common (Earp et al., 2022), with active techniques such as transplanting bringing the benefit of the facilitative presence of the canopy, which immediately begins to ameliorate the environmental to improve recruitment. While passive restoration techniques have been found to benefit multiple marine canopy forming species, active restoration of multiple forest species simultaneously has not yet been attempted (Morris et al., 2020b, Earp et al., 2022).

In addition to restoring marine forests, restoration also aims to restore the associated function and ecosystem services which they provide. Recovery of biodiversity has been found to lag behind re-establishment of the marine forest canopy (Marzinelli et al., 2016, Galobart et al., 2023), particularly with regard to higher trophic levels which can take decades (e.g. Babcock et al., 2010, Reed, 2017). Despite this, restoration efforts with kelps (*E. radiata*), and fucoids (*P. comosa*, *Gongolaria barbata*), have reported of that recovered marine forests can support similar communities and functional complexity to their natural counterparts, given time (Marzinelli et al., 2016, Galobart et al., 2023).

Scale

Small-scale projects have value in identifying appropriate techniques, demonstrating proof of concept, and can be effective in the short-term to conserve genetic diversity and local adaptations (Boström-Einarsson et al., 2020). However, large-scale restoration efforts have the potential to be more successful, as larger, denser patches may be more resilient to competition from turf algae, dilute grazing pressure (Hambäck and Englund, 2005, Morris et al., 2020b, Earp et al., 2022) and confer facilitative interactions (i.e. transplanted adults ameliorate the environment, enhance recruitment and growth) to a greater extent (Layton et al., 2019, Eger et al., 2020b).

Passive methods (e.g., artificial reefs and urchin removal) were found by Morris et al. (2020b) to be more scalable than active methods, involving projects covering more than 10,000 m^2. However this is not necessarily always the case. For example, it has been suggested that due to it's labour intensive nature, urchin removal may be more suited to localised, tactical interventions, to temporarily boost the resilience of existing forests (Layton et al., 2019, Ling et al., 2009), mitigate developing barrens (Ling, 2008, Tracey et al., 2015b), or in support active restoration efforts (Sanderson et al., 2003).

In terms of time scale, examples of restoration from *Cystoseria* spp. the Mediterranean have demonstrated successful outcomes over a ten year period, including expansion beyond the original restoration area, with comparable marine forest size structure, functional and species diversity to reference areas (Gran et al., 2022, Galobart et al., 2023).

In practice, scale is more likely to be informed by availability of funding, although ideally the scale of the restoration activity should be consummate with the scale at which the driver of loss is occurring (Morris et al., 2020b).

Cost

In a recent review, Eger et al. (2021) reported high variability between methodologies and project costings globally (summarised in Table 3). The lowest cost restoration technique was found to be urchin removal (quickliming followed by manual), followed by seeding and then transplanting (Eger et al., 2021). Fredriksen et al. (2020) also found herbivore removal to be the cheapest method as US$ 2 (~£1.60) m^{-2}, followed by artificial reefs US$ 8 (~£6) m^{-2}, seeding US$ 48-118 (~£38-93) m^{-2} and transplanting US$ 6-160 (~£5-125) m^{-2}.

In Norway, the Marine Ecosystem Restoration in Changing European Seas (MERCES) project focussed on two species that occur in the UK, *Laminaria hyperborea* and *Saccharina latissima* (Groeneveld et al., 2019). They found that removing sea urchins using lime was the cheapest method, costing € 129,000 (~£111,000) for an area of 0.9 km^2. Transplanting was about € 10,473 (~£9,000) for an area of $100m^2$, while creation of artificial reefs was the most expensive method, costing € 209,466 (~£180,200) for 500 m^2. The authors note, however, that a fair comparison of these interventions would require success rates, which are yet unknown. While similar for most methods, the costing reported here for artificial reef creation are in contrast with the estimations of Fredriksen et al. (2020), who estimated this method as one of the cheapest. Clearly costing is highly context dependant, illustrating the challenge of evaluating the cost of each method.

Seeding and transplanting are expected to incur similar costs in terms of materials, transport and fuel, equipment, and hourly rates for manual labour (e.g., divers and drivers) (Carney *et al.,* 2005; Campbell *et al.,* 2014). The cost of transplanting can also be reduced by incorporation into artificial reef structures, as attachment methods can be included in the design, avoiding the need to attach to bedrock (Eger et al., 2021).

Widely considered a prime example of restoration success (Bellgrove et al., 2004, Layton et al., 2020b, Vergés et al., 2020), Operation Crayweed reported costs for installing 6 x 2 m^2 patches of *P. comosa* transplants on mesh mats at densities of 15 m^{-2} at 11 restoration sites. Using a team of four working for approximately five days (including preparation and collections from donor populations), costs were estimated at ∼US$ 6,850 (~£5,400) per site (i.e., ∼$570 or ~£450 m^{-2}). Costs included basic consumables, boat and tow-vehicles, SCUBA equipment and air fills, but did not cover project management and monitoring, which were estimated at a further US$18,500 (~£14,600) per annum for all sites. Research to underpin decision making and restoration strategy was also not costed (Layton et al., 2020b, Campbell et al., 2014).

The costs of transplanting adult fucoids for Operation Crayweed are broadly comparable with those reported for transplanting juvenile kelp. When Graham et al. (2021) explored transplanting juvenile kelp on tiles, they estimated that to cover an area of 140 m^2 at a density of 8.3 sporophytes/m^2 would incur similar costs to those reported for *P. comosa* on mesh mats and of *S. latissima* on green gravel (Layton et al., 2020b, Fredriksen et al., 2020). By contrast, Fredriksen et al. (2020), estimated the costs of green gravel to be less than seeding and transplanting techniques at US$7 (£~£5.50) m^{-2}, based on the staff time, materials and fuel needed to produce and deploy 116 kg green gravel. However, in keeping with other restoration projects, subsequent monitoring, bench fees, vessel, vehicles and other fixed infrastructure costs were not included in estimated costings.

As an illustrative UK example, approximate costings are provided for pilot trials using the green gravel technique to seed *Saccharina latissima* in Plymouth, conducted in 2022 (Wilding pers. comm., 2022). Grow-out in the laboratory is estimated to take three months, prior to deployment at sea and monitoring over the course of the following year. Planning and preparation, including collection of fertile material, spore extraction, cleaning and preparation of gravel substrates, inoculation, laboratory cultivation (cleaning, water changes, preparation of nutrients, and modification of lighting intensity) was estimated at five days for a team of two staff (costed at £300 per day), totalling £3,000. Deployment and monitoring of success would require 5 days of a commercially qualified dive team and vessel, costed at £1,000 per day. Consumables (nutrient media, lab equipment, filtered sea water) totalled £1,500 and the use of the aquarium facilities (tanks, chillers, air supply, lighting, nutrient media, technician time) £2,000. Therefore, an estimated £11,500 is required to seed a small area ($2m^2$) at four sites.

Applicability to the UK

As there have yet to be widespread losses of marine forests reported in the UK it is unlikely that active restoration will be a conservation priority in the near future. However, as detrimental impacts on marine forest habitats are likely to continue to increase with increasing rate and intensity of environmental change, the identification, testing and development of restoration techniques which will be feasible and successful in the UK would be advantageous and timely. This is particularly important given that restoration success is more likely in areas where declines are less extensive, and where healthy donor populations persist nearby. Consequently, there is a need to establish interventions which can be implemented swiftly, and for monitoring to identify declines as they occur.

Table 2 evaluates restoration techniques with selected examples, and suggests potential candidate species in the UK that would be appropriate for each technique. Few species that occur in the UK have been tested with active restoration techniques, with the exception of *S. latissima* (and to a lesser extent *L. hyperborea*) (see Eger et al., 2021). In Norway, using these species, transplanting, seeding and competitor removal have been attempted (Moy and Christie, 2012), although the majority of interventions have focussed on urchin removal, so are of limited applicability to the UK at present. UK urchin abundance may increase following storm disturbance (Earp et al., in prep.), and areas of localised urchin barrens in southeast Scotland and northeast England have been reported (Moore pers. comm., 2022), so this region could be a suitable candidate for development of urchin management techniques if restoration interventions are considered necessary.

Green gravel is a promising technique, due to its scalability. Extensive testing of the applicability of this technique across different environmental contexts and species is currently underway (Green Gravel Action Group), with early results indicating it is suitable for several UK species, including *S. latissima* and *L. ochroleuca* (Franco pers. comm., 2022). Initial trials in the UK using this method suggest it may be more applicable to subtidal environments or wave sheltered conditions, with poor retention of gravel in the intertidal attributed to storms and wave action in dynamic UK systems (Earp et al. unpublished), but further research is required to support this.

A further approach which has great potential is the selection of "future-proof", warm-adapted genetic lineages, which can be reared in the aquarium and transplanted into the field to boost resilience of restored populations to predicted environmental conditions (Layton et al., 2020b). This could also be pre-emptively applied to natural forests in anticipation of losses. Although the ethics of these "assisted evolution" approaches should be given full consideration and regulation (Coleman and Goold, 2019, Coleman et al., 2020, Filbee-Dexter and Smajdor, 2019), and the cost of genetic methods may be inaccessible to some practitioners, these pioneering techniques represent one of very few promising solutions to global stressors such as ocean warming. Subject to substantial further research, these methods could be applied to *A. esculenta* and *L. hyperborea*, as both are cool adapted species close to their southern distribution limit in the UK. Due to its disproportionally high habitat value and large spatial extent, *L. hyperborea* could arguably be the priority candidate.

The use of artificial reefs for marine forest restoration has yet to be tested in the UK, however "green engineering" has been the focus of some research (Evans et al., 2021), and has the potential to mitigate the impact of marine infrastructure and development (Dafforn et al., 2015, Morris et al., 2018). One example is the Ecostructure Project, which ran from 2017 to 2022 to explore eco-engineering and biosecurity solutions for coastal adaptation to climate change (see www.ecostructureproject.aber.ac.uk). Eco-engineering has been trialled with *Laminaria digitata*, in Newcastle (Moore pers. comm., 2023) and with intertidal fucoids which colonised artificial rockpools retrofitted to seawalls (e.g. Drakard et al., 2023).

A discussion of other UK species which, to our knowledge, have yet to be tested with any active restoration techniques is as follows (based on authors opinion). *L. digitata* could be suitable, using a range of techniques, due to early maturity and fast growth. It is close to its southern range edge in the UK and declining in Brittany. However, it's preference for wave exposed intertidal habitats is likely to present significant logistical challenges. Opportunistic *Sacchoriza polyschides* is a "climate change winner", as is *L. ochroleuca*, hence both are unlikely to require restoration focus on the UK. As intertidal restoration efforts have so far been less successful than those in the subtidal (Earp et al., 2022), the wracks *Fucus* spp. and Sea Spaghetti *H. elongata* would need further research to establish restoration feasibility. *Ascophyllum nodosum* is unlikely to be well suited to restoration, due to extremely high first year mortality, slow growth rates, and the primary method of growth being vegetative, from a small basal plate which is likely to be easily damaged during transplanting.

Best practice recommendations

Due to the high cost of restoration, priority should be given first to monitoring and conserving marine forest systems before significant losses occur (Morris et al., 2020b, Layton et al., 2020b, Eger et al., 2021). When restoration is necessary, there is no "one size fits all" solution, and consideration should be given to local conditions at the site, species, scale of degradation, the nature and cost of the action required, the likelihood of success and the socioeconomic context (Morris et al., 2020a, Wilson et al., 2011). A restoration toolkit comprised of a suite of methods may be necessary to adapt or modify for applicability to any given situation. General principles of restoration from various habitats have been used to guide development of best practices (Morris et al., 2020), and flow charts for selection of methods can be utilised (Eger et al., 2022b, Smith et al., in press, Cebrian et al., 2021).

The combined use of multiple methods in synergy has been demonstrated to be effective (Earp et al., 2022, Cebrian et al., 2021), for example transplanting juveniles and seeding with spore bags (Hernandez-Carmona et al 2000), transplanting adults onto artificial structures (Layton et al., 2020b) and transplanting with urchin removal (Arroyo-Esquivel et al 2021).

Parallels can be drawn from restoration efforts in other coastal habitats, such as shellfish reefs (Fitzsimons et al., 2020), seagrasses (Fonseca, 1998). For example, research from seagrass meadows has demonstrated the need to ameliorate the drivers of loss prior to commencing active restoration, the importance of proximity to healthy donor populations, and the benefit of large-scale out-planting on survival and growth (van Katwijk et al., 2016).

To be successful at ecologically meaningful scales, restoration requires the application of key principles and current best practice, summarised from Morris et al. (2020b) and Earp et al. (2022) as follows:

(1) research to identify the driver(s) and scale of loss (Layton et al., 2020b, Morris et al., 2020b)

(2) amelioration of these drivers, without which restoration will be unfeasible (Morris et al., 2020b)

(3) setting appropriate goals and objectives, i.e. to restore to a natural reference habitat, or to the most resilient state possible, which could involve novel, more-suitable, species or genotypes (Wood et al., 2019, Layton et al., 2020b). Aims should be specific, time-bound, measurable, and clearly defined in magnitude and in terms of ecosystem attributes (e.g., species composition, density, diversity, ecosystem function)

4) understanding the factors that influence success, including local abiotic and biotic interactions, the impact of global stressors, and the socio-ecological context

5) development of techniques which are commensurate with the scale of loss or if constrained by funding, of a suitable scale to meet objectives and deliver the intended benefits

6) development of robust, long-term monitoring to evaluate progress, and allow for adaptive management actions

7) social, financial and institutional support (e.g., academic, governmental, industrial, as well as non-governmental organisations and community groups) and strong stakeholder and community engagement.

Ecological considerations

As restoration is more effective at sites close to healthy populations, quick action to restore before losses become widespread is likely to improve success (Eger et al., 2021). Ecological considerations for restoration strategy should include appropriate spatial scales for population connectivity, either by restoring multiple connected sites or a single larger area, allowing populations to become self-sustaining. This could, for example, be achieved through connected MPAs which include marine forest habitats, or by mimicking the connectivity of MPA network design in restoration planning (Coleman et al., 2017, Palumbi, 2003, Almany et al., 2009). Planning would also benefit from taking into account positive facilitation factors (Eger et al. 2021), patch dynamics and the landscape context (Layton et al., 2019), seasonal impacts on growth rate (i.e. out-plant when growth rates are high, in winter to spring for most UK kelp species), grazing pressure, and biofouling which can cause erosion of the blade. Modelling can also be used to inform restoration strategy, prior to spending on costly interventions, such as indicating the outcomes of transplanting over different spatial, temporal, and ecological scales (Arroyo-Esquivel et al., 2021).

Avoiding negative impacts to donor populations in marine forests which are already declining is imperative. The use of laboratory based cultivation to “bio-bank” strains and set up gametophyte cultures (Layton and Johnson, 2021) allows for non-destructive collection of a relatively small amount of fertile material, which can then be used to generate large numbers of cultured specimens for out-planting. Another suitable approach for collection of juvenile sporophytes is to source them from “sink populations”, which have good recruitment but experience mortality prior to reaching maturity (for example shallow habitats subject to high drag forces from wave action). By removing juveniles, rather than established mature plants, the ecological impact on the donor population can be minimised (Carney et al., 2005, Graham et al., 2021). Non-destructive recruitment enhancement techniques have recently been found to result in self-sustaining populations of *Cystoseira barbata*, with wild-collected zygotes found to be cost effective by comparison with lab-culturing (Verdura et al., 2018). Further recommendations on sourcing material for seeding or transplants are provided in Eger et al. (2022b).

Frameworks, synergies with human activities and social support

Frameworks for decision making have recently been developed for marine forest restoration (Eger et al., 2022b, Cebrian et al., 2021, Smith et al., in press), including the development of a monitoring and reporting focussed framework (Eger et al., 2022a). Roadmaps can be used to assist with decision-making and assessment of cost effectiveness (Layton et al., 2020b, Cebrian et al., 2021), as well as clarifying whether restoration is feasible. Although originally designed for terrestrial systems, the Society for Ecological Restoration (SER) “5-star recovery system” has recently been adopted by Operation Crayweed (Layton et al., 2020b). This provides a conceptual framework and a set of International Standards for the practice of ecological restoration, with a set of consistent criteria (e.g., survival, growth, health indicators, genetic diversity, and recruitment) against which key ecosystem attributes can be assessed (Layton et al., 2020b). Together, these techniques should guide appropriate actions and allow for tracking and evaluation of success, standardised, comparable reporting, and adaptive management. Ultimately, this could allow for improved understanding and predictability of restoration success, improved project planning, and more successful restoration projects (Christie et al., 2021).

Restoration goals may be facilitated where there are synergies with human activities, for example through creation of market-based solutions to urchin pressure (Pert et al., 2018), provision of propagules, selected strains, technology, techniques, and infrastructure from seaweed aquaculture (e.g. Suebsanguan et al., 2021) or integrated multi-trophic aquaculture applications, whereby the excess nutrients created by shellfish or finfish cultivation are absorbed by kelp, reducing negative environmental impacts (Hadley et al., 2018, Buschmann et al., 2017) which could create motivation to fund restoration through the “restoration economy” (BenDor et al., 2015). Further, MPA’s which lack marine forests may represent good candidate sites for restoration activities, or when healthy marine forests are already present could provide a source of propagules (Gianni et al., 2013). By co-location on restoration within MPA’s, there is the potential for MPA enforcement or amelioration of additional local stressors benefiting restoration efforts.

One of the most successful restorations attempts, Operation Crayweed, has exemplified good stakeholder engagement throughout. The project has connected with the public, community groups, schools, and artists using both traditional science communication, crowdfunding, art installations and storytelling (Vergés et al., 2020) to raise awareness and promote marine stewardship.

Monitoring

Marine forest restoration is a rapidly evolving field, where there has been very limited standardisation of monitoring and reporting of outcomes, with monitoring often limited in scope and duration (Earp et al., 2022, Eger et al., 2022a). This limits our understanding of the drivers of success, impairing growth of the field and preventing knowledge sharing to improve practices (Eger et al., 2022a). Generally, projects have been characterised by a lack of monitoring or reporting of environmental variables, and a bias towards the reporting of abundance (i.e. density or biomass) and morphological (i.e. holdfast diameter, thallus length etc.) response variables (Earp et al., 2022). For example, Earp et al. (2022) found that ~30% of studies failed to specify the depth at which the experiment was conducted, while many provided only limited detail of other potentially important environmental variables (e.g. wave exposure, light availability). A lack of standardisation in surveying techniques and recording protocols has also been identified as limiting to upscaling of marine restoration in other systems (Boström-Einarsson et al., 2020).

The field will progress through comparison of restoration techniques across both a range of species and environmental contexts, which requires the detailed reporting of environmental variables. The global Green Gravel Action Group (GGAG) has recently been initiated, aiming to understand how variation in environmental conditions influences success. Similar work is ongoing to investigate eco-engineering methods aiming to enhance bivalve populations, as part of the World Harbour Project (www.worldharbourproject.org/bivalve-restoration).

As a case study which exemplifies effective, rigorous monitoring allowing for adaptive management, Morris et al. (2020b) highlight "Wheeler North Reef" in California. The creation of an artificial reef was mandated to offset kelp losses resulting from warm wastewater emitted by a nuclear power station. Monitoring, conducted by an independent scientific team, assessed the reef against performance standards to evaluate whether it was achieving the goal of replacing the habitat which had been lost. When, ten years after its installation, it was found to be failing because it was not large enough, creation of additional reef area was mandated (Reed et al., 2019). The combination of measurable objectives and adaptive management will ensure that restoration aims are ultimately achieved.

Crayweed case-study

One of the most successful active restoration attempts to date is "Operation Crayweed" in Sydney (Australia) (Layton et al., 2020b). Declines in crayweed *Phyllospora comosa* were driven by poor water quality in the 1970s, however forests did not recover in response to water quality improvements (Verges 2021). Restoration involved transplanting individuals onto plastic mesh mats at several sites. Success was variable; however survival of transplants was found to be comparable to that of natural populations. Crucially, the stress of transplant disturbance appears to have stimulated reproduction, with some transplanted populations reproducing at greater rates than their natural counterparts (Campbell et al 2014). As a result, populations at restoration sites have become self-sustaining, with juvenile "craybies" from multiple generations now identifiable. Individuals have now spread beyond the original restoration area (Verges 2021) and are beginning to deliver ecosystem functions to the same extent as natural forests (Marzinelli 2016). This project highlights the importance mitigating the driver of loss, ensuring good propagule supply, and of the presence of adult canopy in facilitating recruitment.

3. Future directions

3.1. Challenges

As no literature was found on marine forest restoration from the UK, research is required to identify suitable sites, mitigate drivers of loss and understand the applicability of restoration techniques for local species and environmental contexts.

Defining success in marine forest restoration remains challenging, as author inference is often biased ("failures" go unreported), and survival (particularly when inferred from short-term monitoring), can be misrepresentative, by failing to reflect season variation, stochastic events, or the lagged return of ecosystem processes (Earp et al., 2022, Eger et al., 2021, Galobart et al., 2023).

Political, institutional and financial support for restoration, is critical to longevity and success at ecologically meaningful scales (Eger et al., 2020b, Smith et al., in press). However, securing this support is dependent on socioeconomic motivation. Currently, despite recent advances in understanding, the benefits of marine forest habitats remain poorly understood, which limits motivation for investment in their restoration (De Groot et al., 2013). This challenge can be addressed by quantifying and promoting the value of ecosystem services, both at the community level and by incorporation into legislative decision making (Eger et al., 2020b).

Scale

The aim of restoration should be to undertake interventions on a scale commensurate with loss, however the size of interventions are usually limited by funding or institutional support (Morris et al., 2020b, Earp et al., 2022, Eger et al., 2021). Given that, to date, most restoration work has been small-scale (Eger et al., 2020b, Earp et al., 2022, Morris et al., 2020b, Layton et al., 2020b), and studies at larger spatial scales represent a key knowledge gap, up-scaling of restoration projects represents a key challenge. Where larger scale projects do exist (i.e. more than 10,000 m^2), they have so far mostly utilised passive restoration techniques (e.g., artificial reefs and urchin removal) (Morris et al., 2020b). Operation Crayweed is an example of a relatively small-scale intervention which has expanded beyond the restored site and delivers comparable ecosystem functioning to natural forests (Layton et al., 2020b, Marzinelli et al., 2016). This project involved passive restoration first (improvements to water quality), followed by active transplanting.

In most situations, pilot scale interventions are likely to be necessary initially in order to demonstrate proof of concept, with larger restorations efforts following small scale success (Eger et al., 2022b). In addition to spatial scale, longer temporal scale projects are also likely to be limited due to funding constraints. With increased scale there will be an inevitable increase in cost (discussed further below), although economies of scale are likely as cost does not necessarily scale linearly (Eger et al., 2022b).

To achieve successful large scale restoration, research and monitoring will be essential in order to identify losses, mitigate drivers, inform decision making, and evaluate success (Lake and Restoration, 2001, Earp et al., 2022, Boström-Einarsson et al., 2020, Wood et al., 2019, Lake, 2001). Institutional support, government funding, and strong engagement with communities and stakeholders are also commonalities shared by the four largest restoration projects globally, which are thought to have enabled their large scale (Eger et al., 2020b).

In the case of the most widely used active technique, transplanting, implementation at scale for kelp has so far been rare and received mixed success (Sanderson et al., 2003). Despite this, the fact that the method has been applied extensively, and often successfully, for both kelp (Bennett et al., 2017, Fowler-Walker et al., 2006, Layton and Johnson, 2021) and fucoids (Campbell et al. 2014) at research scales is encouraging. In one example, using tiles to transplant juvenile kelp *E. radiata,* deployments could be up scaled to cover over 140 m^2 at a density of 8.3 sporophytes/m^2, with costs comparable to those of green gravel (Graham et al., 2021). Green gravel is a promising method for application at scale, and mobile "restoration containers", suitable for use in remote coastal communities that may be isolated from institutional support, are currently under development by SeaForester (www.seaforester.org) in Portugal (Verbeek pers comm., 2022) to facilitate and expand the use of this method.

Licencing / adaptive legislation

Marine forest restoration efforts are often limited by substantial financial (see costs section), legal and permitting barriers. To allow for pilot-scale testing followed by scaling-up of successful approaches, adaptive legislative frameworks which allow for adjustments informed by robust monitoring will be required (Eger et al., 2021).

There are several marine protection designations in the UK, each with its own set of regulations. The most common are outlined below:

- Marine Conservation Zones (MCZs): MCZs are areas designated to protect nationally important marine habitats, species, and features. There are currently 91 MCZs in English and Welsh waters, and more are expected to be designated in the future.
- Special Areas of Conservation (SACs): SACs are designated under the European Union's Habitats Directive to protect important habitats and species. There are currently 112 SACs in UK waters.
- Special Protection Areas (SPAs): SPAs are designated under the European Union's Birds Directive to protect important populations of bird species. There are currently 143 SPAs in UK waters.
- Sites of Special Scientific Interest (SSSIs): SSSIs are designated under UK legislation to protect important geological or biological features. There are currently 103 SSSIs in UK waters. SSSIs principally cover the intertidal zone, with only a few extending into areas of significant kelp forest.
- Ramsar sites: Ramsar sites are wetlands of international importance designated under the Ramsar Convention. There are currently 173 Ramsar sites in the UK, many of which include marine areas.

- Nature Conservation Marine Protected Areas (NCMPAs): NCMPAs are designated in Scottish waters to protect important habitats, species, and features. There are currently 30 NCMPAs in Scotland.

The licencing required and legislation protecting these sites vary. Any large-scale restoration project that falls within or may affect these protected sites will need the consent and agreement by the Statutory Natural Conservation Bodies (such as Natural England, NatureScot, Natural Resources Wales or DAERA). In many respects, restoration within protected sites may face more challenges that outside due to the heightened compliance required. There is however, potential for restoration projects to be permitted if classed as management activities for the designated features of the protected site - this is worth exploring further.

Further to this, the deployment of substrate (such as green gravel) into the marine environment will require consultation with the Marine Management Organisation (MMO), and the sea-bed owner, which is usually the Crown Estate. In addition to land-owner permission, licenses must be obtained from the MMO if vessels are used as part of operation (expected to be necessary for scale-up).

As there have been very few large-scale marine restoration projects, the litigation and process associated with achieving permission is not well defined. It is worth noting that UK litigation was written with a focus on reducing declines without consideration of restoration or afforestation activities. As a result, restoration activities in the UK face substantial legislative barriers. The UK is set to introduce a new designation type called Highly Protected Marine Areas (HPMAs), which are hoped to have the flexibility to allow for positive restoration action.

Lessons can be learnt from success restoration projects such as seagrass and oyster beds. Policies with specific, focussed targets (as opposed to generic targets i.e. restoration of 30% of degraded habitats by 2030) would also be beneficial in achieving restoration outcomes, going forward (see Smith et al., in press).

Cost

Marine restoration is expensive and labour intensive, with costs averaging hundreds of thousands of US dollars per intervention (Bayraktarov et al., 2016). Due to these high costs, it is likely that in most cases conservation of remaining habitats will be prioritised over restoration (e.g. Johnson et al., 2017). This could be achieved through holistic management, or use of passive techniques which mitigate the drivers of loss and can be more cost effective than active methods. Where active methods are deemed necessary, the most suitable and cost-effective approach can be identified by cost–benefit analyses (Morris et al., 2020b, Birch et al., 2010). Costs can potentially be reduced by utilising synergies with anthropogenic activities such as aquaculture or exploring market driven models (e.g., a profitable urchin fishery), and by minimizing labour or increasing mechanisation. For marine forest forming species such as fucoids, which have rapidly sinking zygotes, non-destructive wild collection can be cheaper than lab-culturing (Verdura et al., 2018). By contrast, lab culture of green gravel is considered relatively cost effective (Fredriksen et al., 2020).

Estimating costs is difficult, given that only a few authors have reported cost (but see Carney et al., 2005, Campbell et al., 2014, Tracey et al., 2015a, Fredriksen et al., 2020, Layton et al., 2020b), with detailed cost-benefit analyses at appropriate spatial scales so far completely absent (Morris et al., 2020b, Eger et al., 2021). Of the costing exercises which have been conducted, cost breakdown has been inconsistently reported (Bayraktarov et al., 2016), limiting the scope for decision making on whether, what, how, where, and how much to restore, and impeding the creation of accurate budgets or cost-benefit trade-offs (Iacona et al., 2018).

It is also important to note that costings rarely include those associated with the research necessary to underpin decisions, such as the identification of appropriate techniques, selection of donor and restoration sites, and restoration landscape strategies (e.g. patch size) (Campbell et al., 2014, Groeneveld et al., Cebrian et al., 2021), nor the costs to finance robust monitoring which is essential for successful implementation (Layton et al., 2020b, Campbell et al., 2014). Furthermore, the costs of non-consumable laboratory equipment (e.g. stereomicroscopes or dive gear) needed for these restoration projects are not included in estimates as pre-existing facilities are usually utilised. Also, the distance between sites and the required facilities strongly affects the cost-effectiveness and feasibility of restoration efforts because transport duration and conditions are critical to the survival of recruits and germlings (Cebrian *et al.,* 2021).

Bayraktarov et al. (2016) categorised groups of costs for various marine habitat restoration projects (marine forests not considered) including planning, land acquisition, construction, financing, maintenance, monitoring and equipment repair or replacement as factors which affect the cost, feasibility, and likelihood of success of marine restoration. Marine forest habitats can be expected to incur a similar breakdown of costings.

Ecosystem restoration is expensive, far more so in marine than terrestrial systems (Eger et al., 2020b). Marine forest restoration seems to be more costly than restoration of other marine habitats, although this is based on a small number of marine forest restoration projects which report costs (Eger et al., 2021). This may be due to the emerging nature of the field, in which case costs might be expected to come down as techniques are refined and efficiency improved (Eger et al., 2021). Economies of scale should also bring down the cost per unit of area with increasing project size (Turner and Boyer, 1997). Work is underway by SeaForester (www.seaforester.org) in Portugal, to trial scalable mobile container units for the green gravel rearing, which could also improve accessibility for coastal communities who would not otherwise have access to laboratory facilities. Selecting sites connected to intact kelp populations and short distances from the required restoration facilities will also reduce project costs (Morris et al., 2020b, Layton et al., 2020b). In addition, more research into the ecological processes that underpin survival and proliferation of target marine forest species will help to guide project planning and improve techniques, further reducing costs (Campbell et al., 2014).

3.2. Gaps and solutions

Recent reviews have highlighted several key knowledge gaps which are currently limiting capacity building and up-scaling of restoration efforts globally. These include a lack of understanding of what drives success (Eger et al., 2022a) which arises from inconsistent application of methodologies, monitoring and reporting of environmental variables (Earp et al., 2022, Eger et al., 2021). Also identified is a lack of ecological understanding of larger scales and duration (although see Eger et al., 2020b), and of the interaction between multiple methodologies, limited comparative research across environmental conditions, and limited evidence to inform cost-benefit analysis, budget planning and facilitate best practice (Morris et al., 2020b, Eger et al., 2021, Earp et al., 2022, Smith et al., in press). Projects have also been geographically and taxonomically clustered, with some kelp dominated regions, including the UK, not represented in the literature (Eger et al., 2021, Earp et al., 2022).

As the majority of restoration projects to date have been of short duration, up-scaling represents a key knowledge gap. Although experiments of limited duration can be useful to demonstrate methodological efficacy, monitoring over limited time frames can result in misinterpretation of results, for example by failing to capture seasonal variation in growth rates, biomass loss, or dislodgement (Earp et al., 2022, Brown et al., 1997, Hernandez-Carmona et al., 2000, Graham et al., 2021, de Bettignies et al., 2013). Short-term monitoring also fails to represent the impact of stochastic events (e.g. storm surge - see De La Fuente et al., 2019), or the return of associated communities and ecosystem-level functioning, which can lag considerably behind canopy reestablishment (Christie et al., 1998, Galobart et al., 2023).

Following development of a suitable approach through pilot research, additional ecological data is required to inform the appropriate spatial (i.e., size, density) and temporal (i.e., trajectory of restoration) scales needed to achieve self-sustaining marine forest populations (Morris et al., 2020b). There is a lack of knowledge of factors which influence success, and the ecological responses of different species (for example patch dynamics and size dependency, see Morris et al., 2020b), which limit restoration strategy and prioritisation. This may be particularly important, given that increasing patch size and density of mature transplants can improve survival of recruits (Layton et al., 2019, Eger et al., 2020b), and dilute grazing pressure (Morris et al., 2020b, Earp et al., 2022) to improve overall success. Also, there is a paucity of knowledge on the impacts of genetic diversity on restoration success (but see Wood et al., 2020) despite it’s potential to improve restoration outcomes (Reynolds et al., 2012).

While the UK has yet to trial marine forest restoration beyond a small pilot study, globally there is a need for larger scale interventions and longer-term monitoring. These are often limited by restricted funding (Eger et al., 2020b) and a lack of core restoration knowledge (Morris et al., 2020b) but are needed to better understand the influence of seasonal variation and stochastic events on the long-term sustainability of restored marine forest habitats. Further, as the re-establishment of the canopy is a prerequisite, restoration of associated communities and ecosystem structure, functioning and services will lag considerably behind, highlighting the need for long-term monitoring

(Earp et al., 2022). Formation of partnerships (i.e between universities, industry, government agencies and community groups), along with strong institutional and financial support may enable restoration to take place at larger scales (Eger et al., 2020b, Smith et al., in press).

One solution to address the issue of high costs (see also Partnerships and Funding section below) could be through community initiatives and the use of citizen scientists. Several examples of seagrass restoration in the UK have used this approach, including the Community Seagrass Initiative (www.csi-seagrass.co.uk) and Project Seagrass (www.projectseagrass.org/programmes). These approaches has also been successful abroad (Ferretto et al., 2021). Similarly, data collected by recreational divers through the Seasearch programme (www.seasearch.org.uk) could be used to monitor for declines or the impact of restoration actions, as in Lucrezi (2021) and Hart (2016). The use of online tools has also created opportunities for volunteers to help process and analyse data (Jones et al., 2018).

3.3. Partnerships and funding

As marine forest restoration is expensive, securing sufficient funding is vital for project success, as is institutional support through provision of logistical, legal, and social frameworks (see Eger et al., 2020b). Furthermore, neglecting to engage with stakeholders and local communities can impair success, as they are the primary interface interacting with target ecosystems.

Multidisciplinary collaborations can be beneficial in addressing socioeconomic barriers, and are common to the four largest kelp restoration projects to date (Eger et al., 2020b). NGO's, private industry and community groups can increase support and ensure that the needs of the community are represented, and government institutions can mandate or incentivise restoration (Clewell and Aronson, 2006), adding legitimacy and legal backing (Eger et al., 2020b). Academic partners provide expert advice, aiding with development of methodologies and ongoing monitoring, while industrial partners can lend technical and logistical support.

Eger et al. (2020b) found that financial support often arises as a result of institutional support, as motivated institutions can leverage funding. Government backing appears to be key to enabling restoration at large scales (Eger et al., 2020b), due to the considerable funding resources available (Meyers et al., 2020). While substantial funding does not guarantee success (Bayraktarov et al., 2016), and does not negate the need for sound planning and ecological knowledge, it does enable application of best practices for restoration (Eger et al., 2020b). Long-term funding allows time for managers to develop and trial approaches, evaluate early results and refine approaches based on evidence (Eger et al., 2020b).

Funding resources may also be mobilised through international agreements. As part of the UN Decade for Ecosystem Restoration, and Ocean Science for Sustainable Development, support and binding targets have been set for kelp restoration, which should incentivise national projects and green businesses (Eger et al., 2020b). The Kelp Forest Alliance have initiated a global 'Kelp Forest Challenge' target, encouraging pledges to increase awareness, protect or restore kelp forests: www.kelpforestalliance.com/kelp-forest-challenge.

Feasible funding streams include payment for ecosystem services (Meyers et al., 2020), which could attract for-profit (industry) organisations, while non-profit (universities, NGO's, philanthropies and governments) are likely to be motivated by the provision of publicly accessible services, such as fisheries or cultural services. Carbon and nutrient credits could be awarded to restoration projects, potentially creating the opportunity to profit from restoration work (Rutherford et al., 2009, Herr et al., 2017). Investment could be motivated by national government pledges to reduce greenhouse gas emissions through nature based solutions (Eger et al., 2022a, Eger et al., 2021) and from carbon offsetting by industry (Vanderklift et al., 2018). Industrial gains, for example from fisheries dependent on restored kelp habitats, or urchin fisheries, could justify both government and business investment. Investment may also be beneficial to re-invigorate the economy following the Covid-19 pandemic by increasing spending, such as the 2009 USA oyster reef restoration which formed part of an economic stimulus package (Smaal et al., 2019).

Through development of a "restoration economy", restoration efforts can have significant positive economic and employment outputs (BenDor et al., 2015). Furthermore, "green businesses" and private enterprises could potentially fund restoration, allowing them to build social capital by "giving back" while still generating a sustainable revenue (Eger et al., 2021). Existing companies include Urchinomics (www.urchinomics.com) and Greenwave (www.greenwave.org) (Eger et al., 2021). Clearly there is considerable scope and potential for funding restoration projects, and capacity to promote development of the field though coordinated information sharing and partnerships.

3.4. Future priorities and next steps for restoration in the UK

To our knowledge, there are currently no clear localised or widespread losses of marine forests in the UK which can be identified as urgent priorities for active restoration. This is not to say that our coastal ecosystems are free from anthropogenic impacts, far from it. Species substitutions and range shifts, community reconfigurations, and increases in invasive species have been documented, however these are yet to result in complete loss of marine forest habitats. Prioritization of conservation and sustainable management of existing habitats is therefore imperative.

With the increasing severity of climate change and anthropogenic impacts on marine forests, losses in the near future are a plausible reality. There is compelling evidence to suggest that restoration efforts are likely to be more successful in areas subject to less dramatic declines (i.e., 10-30% compared to 95%) and in areas in close to remaining healthy marine forests. Therefore, restoration efforts are likely to be more successful if responses to losses are swift and targeted. As such, testing and refining restoration approaches to establish proof-of-concept and develop a UK restoration toolkit is timely.

Action is required to monitor the status of UK marine forests, to enhance ecological knowledge, establish baselines and allow for reliable detection of significant changes. A kelp habitat suitability model is under development by the Joint Nature Conservation Committee, as part of the Healthy Biologically Diverse Seas Group (HBDSEG) project. Potentially, if this identifies areas where kelp habitats should exist but do not, ground truthing could contribute to identification of losses, and potential drivers modelled. For example, benthic trawling impacts and sedimentation are more likely on low relief ground than on high relief reef, which could be relatively easy to manage following a similar approach to the IFCA bylaw in Sussex.

Once losses are identified, the drivers would require amelioration, and work flows applied. Cases where the drivers of loss have been mitigated include the Durham coastline (following a clean-up of coal mining waste) and Sussex (where a by-law banning bottom trawling aims to improve water quality). Local losses of *Laminaria hyperborea* forest have been reported from shallow waters around the Farne Islands and St Abbs, following the storms of winter 2022 (Moore 2022 prs comm.). As intact healthy marine forests are present nearby these areas, it is anticipated that recovery will occur naturally, now that the drivers of loss have ceased. Robust scientific monitoring of these sites will provide evidence of re-establishment, or lack thereof, following which further actions to actively restore should be considered.

In cases where drivers of loss are largescale (i.e. ocean warming) and therefore difficult to ameliorate, conservation priorities should be to boost resilience of remaining populations, for example through MPA designation, as there is some evidence that resilience to global stressors may be greater when local stressors are ameliorated, particularly if multiple stressor interactions are synergistic (Bates et al., 2019). Further work to identify thermally tolerant genotypes could be conducted, cultures initiated and transplants out-planted. However, genetic techniques are costly, even by comparison with active restoration, and require ethical consideration (see “Future-proofing”, above). Due to declines at its southern trailing range edge, *Alaria esculenta* could be a suitable candidate for restoration using such techniques. However, this species is found in high to very high wave energy environments, which would pose substantial logistical challenges that may make restoration unfeasible.

Results from a survey to identify possible losses and declines of marine forests in the UK (Appendix 1) have identified species substitutions in southwest England as a concern, as *Laminaria ochroleuca* extends it range northwards with ocean warming, potentially displacing *Laminaria hyperborea.* Given that, despite evidence of ecological impact resulting from this substitution (Smale and Vance, 2015, Teagle and Smale, 2018), dense kelp forests are still present in these areas, there is unlikely to be sufficient motivation to justify costly restoration interventions, particularly those that would likely require genetic selection to boost thermal tolerance and resilience of *Laminaria hyperborea*. In addition to difficulties in securing funding or action from conservation managers for such work, social support is likely to be low, as indicated by the survey response that "there are still kelp forests [there]".

In summary, few restoration techniques have been tested with UK marine forest forming species or at UK sites, and it may be that a suite of methods is required to account for variation in environmental and social contexts. The field of restoration of UK marine forests lags significantly behind other regions, and insufficient technique development and knowledge will hinder restoration efforts if/when needed in response to local and global stressors. Should active techniques be necessary, green gravel is promising due to its relatively low cost and high scalability, and if transplanting were considered, out-planting of mixed age classes or combining with seeding techniques to allow populations to become self-sustaining is likely to be beneficial.

4. Glossary

Active restoration - actions taken to restore the ecosystem by direct manipulation of the target species, for example by transplanting or seeding, with the aim of afforestation.

Blade - the flattened part of a seaweed that resembles the leaf of land plants.

Epibiont - an organism living attached to the surface of another organism, without any detriment or benefit to the host. Epibionts found on kelp include other seaweeds, sponges, sea squirts, bryzoans, hydroids, crustaceans, gastropods and bivalves.

Frond - the visible plant body, also called the thallus or sporophyte, comprised of the holdfast, stipe and blade.

Fucales - an order of large brown seaweeds including Fucus (wracks), Sargassum (wireweed), and Cystoseria spp.

Haptera - the root like structures which form macroalgae holdfasts.

Harmful microalgal bloom - the rapid and excessive growth of microalgae, with negative impacts on other organisms. Also known as a "red tide" or harmful algal blooms (HABs), often occur in response to excess nutrients and eutrophication. Impacts can affect people, the environment, and the economy through the production of algal toxins or oxygen depletion as the bloom is broken down by other microbes.

Holdfast - the attachment structure at the base of a seaweed, comprised of haptera. It resembles the roots of land plants, but its function is solely for attachment, not for nutrient uptake.

Kelp - a large brown seaweed comprised of a holdfast, stipe and blade, which form forest-like marine habitats. Generally refers to the order Laminariales, but can include Fucales and Tilopteridales.

Laminariales - an order of large brown seaweeds referred to as "true kelps".

Macroalgae - a group of multicellular, macroscopic marine algae known as seaweed, divided into red, brown and green seaweeds.

Passive restoration - the process of ecological restoration whereby natural recovery as allowed to take place. In marine forest restoration, actions are taken on the surrounding environment, rather than afforestation by direct manipulation of individual seaweeds themselves. Examples include removing the original driver of decline, improving environmental conditions, creation of artificial reef habitat, and arguably removal of herbivores or competitors.

Sporophyte - the multicellular, diploid life stage of algae, which produces spores. In kelp, the sporophyte is the macroscopic, plant-like, thallus life stage which alternates with the microscopic, haploid, gametophyte stage.

Stipe - the stalk of a seaweed connecting the holdfast and blade. This resembles a stem in land plants.

Stipitate - having or supported by a stipe.

Thallus - the plant body of algae, fungi and lichens which lacks true roots or a vascular system. In marine forest species the thallus is the sporophyte, comprised of the holdfast, stipe and blade.

Tilopteridales - an order of brown algae which includes the "pseudo-kelp" *S. polyschides.*

5. References

ALMANY, G., CONNOLLY, S., HEATH, D., HOGAN, J., JONES, G., MCCOOK, L., MILLS, M., PRESSEY, R. & WILLIAMSON, D. J. C. R. 2009. Connectivity, biodiversity conservation and the design of marine reserve networks for coral reefs. 28, 339-351.

ANDERSON, M. J., DIEBEL, C. E., BLOM, W. M. & LANDERS, T. J. 2005. Consistency and variation in kelp holdfast assemblages: Spatial patterns of biodiversity for the major phyla at different taxonomic resolutions. *Journal of Experimental Marine Biology and Ecology,* 320, 35-56.

ANDERSON, R., RAND, A., ROTHMAN, M., SHARE, A. & BOLTON, J. J. A. J. O. M. S. 2007. Mapping and quantifying the South African kelp resource. 29, 369-378.

ARROYO-ESQUIVEL, J., BASKETT, M. L., MCPHERSON, M. & HASTINGS, A. 2021.

BABCOCK, R. C., SHEARS, N. T., ALCALA, A. C., BARRETT, N. S., EDGAR, G. J., LAFFERTY, K., MCCLANAHAN, T. R. & RUSS, G. R. J. P. O. T. N. A. O. S. 2010. Decadal trends in marine reserves reveal differential rates of change in direct and indirect effects. 107, 18256-18261.

BATES, A. E., COOKE, R. S., DUNCAN, M. I., EDGAR, G. J., BRUNO, J. F., BENEDETTI-CECCHI, L., CÔTÉ, I. M., LEFCHECK, J. S., COSTELLO, M. J. & BARRETT, N. 2019. Climate resilience in marine protected areas and the 'Protection Paradox'. *Biological Conservation,* 236, 305-314.

BAYRAKTAROV, E., SAUNDERS, M. I., ABDULLAH, S., MILLS, M., BEHER, J., POSSINGHAM, H. P., MUMBY, P. J. & LOVELOCK, C. E. J. E. A. 2016. The cost and feasibility of marine coastal restoration. 26, 1055-1074.

BELLGROVE, A., CLAYTON, M. N. & QUINN, G. P. 2004. An integrated study of the temporal and spatial variation in the supply of propagules, recruitment and assemblages of intertidal macroalgae on a wave-exposed rocky coast, Victoria, Australia. *Journal of Experimental Marine Biology and Ecology,* 310, 207-225.

BELLGROVE, A., MCKENZIE, P. F., MCKENZIE, J. L. & SFILIGOJ, B. J. J. M. E. P. S. 2010. Restoration of the habitat-forming fucoid alga Hormosira banksii at effluent-affected sites: competitive exclusion by coralline turfs. 419, 47-56.

BENDOR, T. K., LIVENGOOD, A., LESTER, T. W., DAVIS, A. & YONAVJAK, L. J. R. E. 2015. Defining and evaluating the ecological restoration economy. 23, 209-219.

BENNETT, S. & WERNBERG, T. 2014. Canopy facilitates seaweed recruitment on subtidal temperate reefs. *J. Ecol.,* 102, 1462–1470.

BENNETT, S., WERNBERG, T., CONNELL, S. D., HOBDAY, A. J., JOHNSON, C. R. & POLOCZANSKA, E. S. 2016. The 'Great Southern Reef': social, ecological and economic value of Australia's neglected kelp forests. *Marine and Freshwater Research,* 67, 47-56.

BENNETT, S., WERNBERG, T. & DE BETTIGNIES, T. 2017. Bubble Curtains: Herbivore Exclusion Devices for Ecology and Restoration of Marine Ecosystems? *Frontiers in Marine Science,* 4.

BERTOCCI, I., ARAÚJO, R., OLIVEIRA, P. & SOUSA-PINTO, I. 2015. Potential effects of kelp species on local fisheries. *Journal Applied Ecology,* 52, 1216–1226.

BIRCH, J. C., NEWTON, A. C., AQUINO, C. A., CANTARELLO, E., ECHEVERRÍA, C., KITZBERGER, T., SCHIAPPACASSE, I. & GARAVITO, N. T. J. P. O. T. N. A. O. S. 2010. Cost-effectiveness of dryland forest restoration evaluated by spatial analysis of ecosystem services. 107, 21925-21930.

BLAMEY, L. K. & BOLTON, J. J. 2018. The economic value of South African kelp forests and temperate reefs: Past, present and future. *Journal of Marine Systems,* 188, 172-181.

BOLTON, J. J. 2010. The biogeography of kelps (Laminariales, Phaeophyceae): a global analysis with new insights from recent advances in molecular phylogenetics. *Helgoland Marine Research,* 64, 263-279.

BOSTRÖM-EINARSSON, L., BABCOCK, R. C., BAYRAKTAROV, E., CECCARELLI, D., COOK, N., FERSE, S. C., HANCOCK, B., HARRISON, P., HEIN, M. & SHAVER, E. J. P. O. 2020. Coral restoration–A systematic review of current methods, successes, failures and future directions. 15, e0226631.

BROWN, M., NYMAN, M., KEOGH, J. & CHIN, N. J. M. B. 1997. Seasonal growth of the giant kelp Macrocystis pyrifera in New Zealand. 129, 417-424.

BUÉ, M., SMALE, D. A., NATANNI, G., MARSHALL, H. & MOORE, P. J. 2020. Multiple-scale interactions structure macroinvertebrate assemblages associated with kelp understory algae. *Diversity and Distributions,* online early.

BURROWS, M. T., FOX, C. J., MOORE, P., SMALE, D. A., SOTHERAN, I., BENSON, A., GREENHILL, L., MARTINO, S., PARKER, A., THOMPSON, E. & ALLEN, C. J. 2018. Wild Seaweed Harvesting as a Diversification Opportunity for Fishermen. *A report by SRSL for HIE.*

BURROWS, M. T., SMALE, D., O'CONNOR, N., VAN REIN, H. & MOORE, P. 2014. Marine strategy framework directive indicators for UK kelp habitats part 1: developing proposals for potential indicators.

BUSCHMANN, A. H., CAMUS, C., INFANTE, J., NEORI, A., ISRAEL, Á., HERNÁNDEZ-GONZÁLEZ, M. C., PEREDA, S. V., GOMEZ-PINCHETTI, J. L., GOLBERG, A. & TADMOR-SHALEV, N. 2017. Seaweed production: overview of the global state of exploitation, farming and emerging research activity. *European Journal of Phycology,* 52**,** 391-406.

BUSCHMANN, A. H., PRESCOTT, S., POTIN, P., FAUGERON, S., VASQUEZ, J. A., CAMUS, C., INFANTE, J., HERNÁNDEZ-GONZÁLEZ, M. C., GUTIERREZ, A. & VARELA, D. A. 2014. The status of kelp exploitation and marine agronomy, with emphasis on Macrocystis pyrifera, in Chile. *Advances in botanical research.* Elsevier.

CAMPBELL, A. H., MARZINELLI, E. M., VERGES, A., COLEMAN, M. A. & STEINBERG, P. D. 2014. Towards restoration of missing underwater forests. *PLoS One,* 9**,** e84106.

CARNEY, L. T., WAALAND, J. R., KLINGER, T. & EWING, K. J. M. E. P. S. 2005. Restoration of the bull kelp Nereocystis luetkeana in nearshore rocky habitats. 302**,** 49-61.

CASELLE, J. E., RASSWEILER, A., HAMILTON, S. L. & WARNER, R. R. 2015. Recovery trajectories of kelp forest animals are rapid yet spatially variable across a network of temperate marine protected areas. *Scientific Reports,* 5**,** 14102.

CEBRIAN, E., TAMBURELLO, L., VERDURA, J., GUARNIERI, G., MEDRANO, A., LINARES, C., HEREU, B., GARRABOU, J., CERRANO, C., GALOBART, C. & FRASCHETTI, S. 2021. A Roadmap for the Restoration of Mediterranean Macroalgal Forests. *Frontiers in Marine Science,* 8.

CHAPMAN, M. & UNDERWOOD, A. 2011. Evaluation of ecological engineering of “armoured” shorelines to improve their value as habitat. *Journal of experimental marine biology ecology,* 400**,** 302-313.

CHOI, C. G., SERISAWA, Y., OHNO, M. & SOHN, C. H. 2000. Using the spore bag method. *Algae,* 15**,** 179-182.

CHRISTIE, A. P., AMANO, T., MARTIN, P. A., PETROVAN, S. O., SHACKELFORD, G. E., SIMMONS, B. I., SMITH, R. K., WILLIAMS, D. R., WORDLEY, C. F. & SUTHERLAND, W. J. J. C. B. 2021. The challenge of biased evidence in conservation. 35**,** 249-262.

CHRISTIE, H., FREDRIKSEN, S. & RINDE, E. 1998. Regrowth of kelp and colonization of epiphyte and fauna community after kelp trawling at the coast of Norway. *Hydrobiologia,* 375-376**,** 49-58.

CHRISTIE, H., GUNDERSEN, H., RINDE, E., FILBEE-DEXTER, K., NORDERHAUG, K. M., PEDERSEN, T., BEKKBY, T., GITMARK, J. K., FAGERLI, C. W. J. E. & EVOLUTION 2019. Can multitrophic interactions and ocean warming influence large-scale kelp recovery? 9**,** 2847-2862.

CHRISTIE, H., JORGENSEN, N. M., NORDERHAUG, K. M. & WAAGE-NIELSEN, E. 2003. Species distribution and habitat exploitation of fauna associated with kelp (*Laminaria hyperborea)* along the Norwegian coast. *Journal of the Marine Biological Association of the UK,* 83**,** 687-699.

CLEWELL, A. F. & ARONSON, J. J. C. B. 2006. Motivations for the restoration of ecosystems. 20**,** 420-428.

COLEMAN, M. A., CETINA-HEREDIA, P., ROUGHAN, M., FENG, M., VAN SEBILLE, E. & KELAHER, B. P. J. G. C. B. 2017. Anticipating changes to future connectivity within a network of marine protected areas. 23**,** 3533-3542.

COLEMAN, M. A. & GOOLD, H. D. J. J. O. P. 2019. Harnessing synthetic biology for kelp forest conservation. 55**,** 745-751.

COLEMAN, M. A., WOOD, G., FILBEE-DEXTER, K., MINNE, A. J. P., GOOLD, H. D., VERGÉS, A., MARZINELLI, E. M., STEINBERG, P. D. & WERNBERG, T. 2020. Restore or Redefine: Future Trajectories for Restoration. *Frontiers in Marine Science,* 7.

CORREA, J. A., LAGOS, N. A., MEDINA, M. H., CASTILLA, J. C., CERDA, M., RAMÍREZ, M., MARTÍNEZ, E., FAUGERON, S., ANDRADE, S., PINTO, R. & CONTRERAS, L. 2006. Experimental transplants of the large kelp Lessonia nigrescens (Phaeophyceae) in high-energy wave exposed rocky intertidal habitats of northern Chile: Experimental, restoration and management applications. *Journal of Experimental Marine Biology and Ecology,* 335**,** 13-18.

DAFFORN, K. A., MAYER-PINTO, M., MORRIS, R. L. & WALTHAM, N. J. 2015. Application of management tools to integrate ecological principles with the design of marine infrastructure. *J Environ Manage,* 158**,** 61-73.

DE BETTIGNIES, T., WERNBERG, T., LAVERY, P. S., VANDERKLIFT, M. A. & MOHRING, M. B. 2013. Contrasting mechanisms of dislodgement and erosion contribute to production of kelp detritus. *Limnology and Oceanography,* 58**,** 1680-1688.

DE GROOT, R. S., BLIGNAUT, J., VAN DER PLOEG, S., ARONSON, J., ELMQVIST, T. & FARLEY, J. J. C. B. 2013. Benefits of investing in ecosystem restoration. 27**,** 1286-1293.

DE LA FUENTE, G., CHIANTORE, M., ASNAGHI, V., KALEB, S. & FALACE, A. 2019. First ex situ outplanting of the habitat-forming seaweed Cystoseira amentacea var. stricta from a restoration perspective. *PeerJ,* 7**,** e7290.

DRAKARD, V. F., EVANS, A. J., CROWE, T. P., MOORE, P. J., COUGHLAN, J. & BROOKS, P. R. 2023. Artificial rockpools: Seaweed colonisation and productivity vary between sites but are consistent across environmental contexts. *Marine Environmental Research,* 188**,** 106022.

EARP, H. S., SMALE, D. A., ALMOND, P., CATHERALL, H. J. N., GOURAGUINE, A., WILDING, C. M. & MOORE, P. J. in prep. Storm disturbance alters the structure, abundance, and composition of Laminaria hyperborea forests and associated understorey assemblages

EARP, H. S., SMALE, D. A., REZER-MATUS, A., GOURAGUINE, A., SHAW, P. W. & MOORE, P. 2022. A quantitative synthesis of approaches, biases, successes and failures in marine forest restoration with considerations for future work. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 32**,** 1717-1731.

ECKMAN, J. E., DUGGINS, D. O. & SEWELL, A. T. 1989. Ecology of under story kelp environments. I. Effects of kelps on flow and particle transport near the bottom. *Journal of Experimental Marine Biology and Ecology,* 129**,** 173-187.

EGAN, S., HARDER, T., BURKE, C., STEINBERG, P., KJELLEBERG, S. & THOMAS, T. 2013. The seaweed holobiont: understanding seaweed–bacteria interactions. *FEMS Microbiology Reviews,* 37**,** 462-476.

EGER, A., MARZINELLI, E., CHRISTIE, H., FUJITA, D., HONG, S., KIM, J. H., LIEE, L. C., MCHUGH, T. A., NISHIHARA, G. N. & VASQUEZ, A. P. G. 2021. Global Kelp Forest Restoration: Past lessons, status, and future goals.

EGER, A. M. & BAUM, J. K. 2020. Trophic cascades and connectivity in coastal benthic marine ecosystems: a meta-analysis of experimental and observational research. *Marine Ecology Progress Series,* 656**,** 139-152.

EGER, A. M., EARP, H. S., FRIEDMAN, K., GATT, Y., HAGGER, V., HANCOCK, B., KAEWSRIKHAW, R., MCLEOD, E., MOORE, A. M., NINER, H. J., RAZAFINAIVO, F., SOUSA, A. I., STANKOVIC, M., WORTHINGTON, T. A., BAYRAKTAROV, E., SAUNDERS, M., VERGÉS, A. & REEVES, S. 2022a. The need, opportunities, and challenges for creating a standardized framework for marine restoration monitoring and reporting. *Biological Conservation,* 266.

EGER, A. M., LAYTON, C., MCHUGH, T. A., GLEASON, M. & EDDY, N. 2022b. Kelp Restoration Guidebook: Lessons Learned from Kelp Projects Around the World. Arlington, VA, USA: The Nature Conservancy.

EGER, A. M., MARZINELLI, E., GRIBBEN, P., JOHNSON, C. R., LAYTON, C., STEINBERG, P. D., WOOD, G., SILLIMAN, B. R. & VERGÉS, A. 2020a. Playing to the Positives: Using Synergies to Enhance Kelp Forest Restoration. *Frontiers in Marine Science,* 7.

EGER, A. M., MARZINELLI, E. M., BEAS-LUNA, R., BLAIN, C. O., BLAMEY, L. K., BYRNES, J. E., CARNELL, P. E., CHOI, C. G., HESSING-LEWIS, M. & KIM, K. Y. 2023. The value of ecosystem services in global marine kelp forests. *nature communications,* 14**,** 1894.

EGER, A. M., VERGÉS, A., CHOI, C. G., CHRISTIE, H., COLEMAN, M. A., FAGERLI, C. W., FUJITA, D., HASEGAWA, M., KIM, J. H., MAYER-PINTO, M., REED, D. C., STEINBERG, P. D. & MARZINELLI, E. M. 2020b. Financial and Institutional Support Are Important for Large-Scale Kelp Forest Restoration. *Frontiers in Marine Science,* 7.

ESTES, J. A., DANNER, E. M., DOAK, D. F., KONAR, B., SPRINGER, A. M., STEINBERG, P. D., TINKER, M. T. & WILLIAMS, T. M. 2004. Complex Trophic Interactions in Kelp Forest Ecosystems. *Bulletin of Marine Science,* 74**,** 621-638.

ESTES, J. A. & DUGGINS, D. O. 1995. Sea otters and kelp forests in Alaska: generality and variation in a community ecological paradigm. *Ecological Monographs,* 65**,** 75-100.

EVANS, A. J., LAWRENCE, P. J., NATANZI, A. S., MOORE, P. J., DAVIES, A. J., CROWE, T. P., MCNALLY, C., THOMPSON, B., DOZIER, A. E. & BROOKS, P. R. 2021. Replicating natural topography on marine artificial structures–A novel approach to eco-engineering. *Ecological Engineering,* 160**,** 106144.

FALACE, A., ZANELLI, E. & BRESSAN, G. 2006. Algal transplantation as a potential tool for artificial reef management and environmental mitigation. *Bulletin of Marine Science,* 78**,** 161-166.

FEJTEK, S. M., EDWARDS, M. S., KIM, K. Y. J. J. O. E. M. B. & ECOLOGY 2011. Elk Kelp, Pelagophycus porra, distribution limited due to susceptibility of microscopic stages to high light. 396**,** 194-201.

FERRARI, R., MARZINELLI, E. M., AYROZA, C. R., JORDAN, A., FIGUEIRA, W. F., BYRNE, M., MALCOLM, H. A., WILLIAMS, S. B. & STEINBERG, P. D. J. P. O. 2018. Large-scale assessment of benthic communities across multiple marine protected areas using an autonomous underwater vehicle. 13**,** e0193711.

FERRETTO, G., GLASBY, T. M., POORE, A. G., CALLAGHAN, C. T., HOUSEFIELD, G. P., LANGLEY, M., SINCLAIR, E. A., STATTON, J., KENDRICK, G. A. & VERGÉS, A. 2021. Naturally-detached fragments of the endangered seagrass Posidonia australis collected by citizen scientists can be used to successfully restore fragmented meadows. *Biological Conservation,* 262**,** 109308.

FILBEE-DEXTER, K. & SCHEIBLING, R. E. 2014. Sea urchin barrens as alternative stable states of collapsed kelp ecosystems. *Marine Ecology Progress Series,* 495**,** 1-25.

FILBEE-DEXTER, K. & SMAJDOR, A. 2019. Ethics of Assisted Evolution in Marine Conservation. *Frontiers in Marine Science,* 6.

FILBEE-DEXTER, K. & WERNBERG, T. 2018. Rise of turfs: A new battlefront for globally declining kelp forests. *BioScience,* 68**,** 64-76.

FILBEE-DEXTER, K., WERNBERG, T., FREDRIKSEN, S., NORDERHAUG, K. M. & PEDERSEN, M. F. 2019. Arctic kelp forests: Diversity, resilience and future. *Global and Planetary Change,* 172**,** 1-14.

FILBEE-DEXTER, K., WERNBERG, T., NORDERHAUG, K. M., RAMIREZ-LLODRA, E. & PEDERSEN, M. F. 2018. Movement of pulsed resource subsidies from kelp forests to deep fjords. *Oecologia,* 187**,** 291-304.

FITZSIMONS, J. A., BRANIGAN, S., GILLIES, C. L., BRUMBAUGH, R. D., CHENG, J., DEANGELIS, B. M., GESELBRACHT, L., HANCOCK, B., JEFFS, A., MCDONALD, T. J. C. S. & PRACTICE 2020. Restoring shellfish reefs: Global guidelines for practitioners and scientists. 2**,** e198.

FONSECA, M. S. 1998. *Guidelines for the conservation and restoration of seagrasses in the United States and adjacent waters*, US Department of Commerce, National Oceanic and Atmospheric Administration ….

FOSTER, M. S. & SCHIEL, D. R. 2010. Loss of predators and the collapse of southern California kelp forests (?): Alternatives, explanations and generalizations. *Journal of Experimental Marine Biology and Ecology,* 393**,** 59-70.

FOWLER-WALKER, M. J., WERNBERG, T. & CONNELL, S. D. 2006. Differences in kelp morphology between wave sheltered and exposed localities: morphologically plastic or fixed traits? *Marine Biology,* 148**,** 755-767.

FRANGOUDES, K. & GARINEAUD, C. 2015. Governability of kelp forest small-scale harvesting in Iroise Sea, France. *Interactive governance for small-scale fisheries.* Springer.

FREDRIKSEN, S., FILBEE-DEXTER, K., NORDERHAUG, K. M., STEEN, H., BODVIN, T., COLEMAN, M. A., MOY, F. & WERNBERG, T. 2020. Green gravel: a novel restoration tool to combat kelp forest decline. *Scientific Reports,* 10**,** 3983.

FUJITA, D. 2011. Management of kelp ecosystem in Japan. *Cahiers de Biologie Marine,* 52**,** 499-505.

GALOBART, C., BALLESTEROS, E., GOLO, R. & CEBRIAN, E. 2023. Addressing marine restoration success: evidence of species and functional diversity recovery in a ten-year restored macroalgal forest. *Frontiers in Marine Science,* 10**,** 1176655.

GIANNI, F., BARTOLINI, F., AIROLDI, L., BALLESTEROS, E., FRANCOUR, P., GUIDETTI, P., MEINESZ, A., THIBAUT, T. & MANGIALAJO, L. 2013. Conservation and restoration of marine forests in the Mediterranean Sea and the potential role of Marine Protected Areas. *Advances in oceanography and limnology,* 4**,** 83-101.

GOURAGUINE, A., MOORE, P., BURROWS, M. T., VELASCO, E., ARIZ, L., FIGUEROA-FÁBREGA, L., MUÑOZ-CORDOVEZ, R., FERNANDEZ-CISTERNAS, I., SMALE, D. & PÉREZ-MATUS, A. 2021. The intensity of kelp harvesting shapes the population structure of the foundation species Lessonia trabeculata along the Chilean coastline. *Marine Biology,* 168**,** 66.

GRAHAM, N. A., BELLWOOD, D. R., CINNER, J. E., HUGHES, T. P., NORSTRÖM, A. V., NYSTRÖM, M. J. F. I. E. & ENVIRONMENT, T. 2013. Managing resilience to reverse phase shifts in coral reefs. 11**,** 541-548.

GRAHAM, T. D. J., MORRIS, R. L., STRAIN, E. M. A. & SWEARER, S. E. 2021. Identifying key factors for transplantation success in the restoration of kelp (Ecklonia radiata) beds. *Restoration Ecology,* 30**,** 13536.

GRAN, A., MOVILLA, J. I., BALLESTEROS, E., SALES, M., BOLADO, I., GALOBART, C. & CEFALÌ, M. E. 2022. Assessing the expansion and success of a restored population of Gongolaria barbata (Stackhouse) Kuntze (Fucales, Phaeophyceae) using high-precision positioning tools and size distribution frequencies.

GRANT J, W. K., GROVER A, TOGSTAD H. 1982. Early development of Pendleton artificial reef. *Marine Fisheries Review* 44**,** 53–60.

GROENEVELD, R., CHEN, W., HYNES, S., TINCH, R. & PAPADOPOULOU, N. 2019. Restoring marine ecosystems cost-effectively: lessons learned from the MERCES project *Marine Ecosystem Restoration in Changing European Seas.*

HADLEY, S., WILD-ALLEN, K., JOHNSON, C. & MACLEOD, C. 2018. Investigation of broad scale implementation of integrated multitrophic aquaculture using a 3D model of an estuary. *Marine pollution bulletin,* 133**,** 448-459.

HAGGER, V., DWYER, J. & WILSON, K. J. R. E. 2017. What motivates ecological restoration? 25**,** 832-843.

HAMBÄCK, P. A. & ENGLUND, G. J. E. L. 2005. Patch area, population density and the scaling of migration rates: the resource concentration hypothesis revisited. 8**,** 1057-1065.

HARLEY, C. D. G., ANDERSON, K. M., DEMES, K. W., JORVE, J. P., KORDAS, R. L., COYLE, T. A. & GRAHAM, M. H. 2012. Effects of climate change on global seaweed communities. *Journal of Phycology,* 48**,** 1064-1078.

HART, L. 2016. Citizen contributions to the science and monitoring of bull kelp (Nereocystis luetkeana) in Puget Sound, WA.

HAWKINS, S., ALLEN, J., BRAY, S. J. A. C. M. & ECOSYSTEMS, F. 1999. Restoration of temperate marine and coastal ecosystems: nudging nature. 9**,** 23-46.

HAYASHI, A., TOZAWA, H., SHIMADA, K., TAKAHASHI, K., KANEKO, R., TSUKIHASHI, F., INOUE, R. & ARIYAMA, T. 2011. Effects of the seaweed bed construction using the mixture of steelmaking slag and dredged soil on the growth of seaweeds. . *ISIJ International,* 51**,** 1919-1928.

HERNANDEZ-CARMONA, G., GARCÍA, O., ROBLEDO, D. & FOSTER, M. 2000. Restoration techniques for Macrocystis pyrifera (Phaeophyceae) populations at the southern limit of their distribution in Mexico.

HERR, D., VON UNGER, M., LAFFOLEY, D., MCGIVERN, A. J. A. C. M. & ECOSYSTEMS, F. 2017. Pathways for implementation of blue carbon initiatives. 27**,** 116-129.

HIRSH, H. K., NICKOLS, K. J., TAKESHITA, Y., TRAIGER, S. B., MUCCIARONE, D. A., MONISMITH, S. & DUNBAR, R. B. 2020. Drivers of biogeochemical variability in a central California kelp forest: implications for local amelioration of ocean acidification. *Journal of Geophysical Research: Oceans,* 125**,** e2020JC016320.

IACONA, G. D., SUTHERLAND, W. J., MAPPIN, B., ADAMS, V. M., ARMSWORTH, P. R., COLESHAW, T., COOK, C., CRAIGIE, I., DICKS, L. V. & FITZSIMONS, J. A. J. C. B. 2018. Standardized reporting of the costs of management interventions for biodiversity conservation. 32**,** 979-988.

IFCA, S. 2022. *Inshore Fisheries Conservation Authority* [Online]. Available: https://www.sussex-ifca.gov.uk/nearshore-trawling-byelaw [Accessed February 2022].

IRVING, R. A. & NORTHERN, K. O. 2012. Isles of Scilly SAC Diving Monitoring Studies, 2011. Natural England Commissioned Reports, Number 104.

JAYATHILAKE, D. R. M. & COSTELLO, M. J. 2020. A modelled global distribution of the kelp biome. *Biological Conservation,* 252**,** 108815.

JIANG, Z., LIU, J., LI, S., CHEN, Y., DU, P., ZHU, Y., LIAO, Y., CHEN, Q., SHOU, L. & YAN, X. 2020. Kelp cultivation effectively improves water quality and regulates phytoplankton community in a turbid, highly eutrophic bay. *Science of the Total Environment,* 707**,** 135561.

JOHNSON, C. R., CHABOT, R. H., MARZLOFF, M. P. & WOTHERSPOON, S. J. R. E. 2017. Knowing when (not) to attempt ecological restoration. 25**,** 140-147.

JOHNSON, L. E. & BRAWLEY, S. H. 1998. Dispersal and recruitment of a canopy-forming intertidal alga: the relative roles of propagule availability and post-settlement processes. *Oecologia,* 117**,** 517-526.

JONES, B. L., UNSWORTH, R. K. F., MCKENZIE, L. J., YOSHIDA, R. L. & CULLEN-UNSWORTH, L. C. 2018. Crowdsourcing conservation: The role of citizen science in securing a future for seagrass. *Marine Pollution Bulletin,* 134**,** 210-215.

KAIN, J. M. 1979. A view of the genus *Laminaria*. *Oceanography and Marine Biology Annual Review,* 17**,** 101-161.

KENNELLY, S. J. 1989. Effects of kelp canopies on understorey species due to shade and scour. *Marine Ecology Progress Series,* 50**,** 215-224.

KING, N., MOORE, P. J., WILDING, C., JENKINS, H. & SMALE, D. A. 2021. Multiscale spatial variability in the structure of epibiont assemblages associated with stipes of the kelp *Laminaria hyperborea* in the northeast Atlantic. *Mar Ecol Prog Ser,* 672**,** 33-44.

KING, N. G., MOORE, P. J., THORPE, J. M. & SMALE, D. A. 2023. Consistency and Variation in the Kelp Microbiota: Patterns of Bacterial Community Structure Across Spatial Scales. *Microbial Ecology,* 85**,** 1265-1275.

KRAUSE-JENSEN, D. & DUARTE, C. M. 2016. Substantial role of macroalgae in marine carbon sequestration. *Nature Geosciences,* 9**,** 737-742.

KRAUSE-JENSEN, D., MARBÀ, N., SANZ-MARTIN, M., HENDRIKS, I. E., THYRRING, J., CARSTENSEN, J., SEJR, M. K. & DUARTE, C. M. 2016. Long photoperiods sustain high pH in Arctic kelp forests. *Science Advances,* 2**,** e1501938.

LAKE, P. S. 2001. On the maturing of restoration: linking ecological research and restoration. *Ecological Management & Restoration,* 2**,** 110-115.

LAKE, P. S. J. E. M. & RESTORATION 2001. On the maturing of restoration: linking ecological research and restoration. 2**,** 110-115.

LAYTON, C., CAMERON, M. J., SHELAMOFF, V., TATSUMI, M., WRIGHT, J. T. & JOHNSON, C. R. J. R. E. 2021. A successful method of transplanting adult Ecklonia radiata kelp, and relevance to other habitat-forming macroalgae. 29**,** e13412.

LAYTON, C., CAMERON, M. J., TATSUMI, M., SHELAMOFF, V., WRIGHT, J. T. & JOHNSON, C. R. 2020a. Habitat fragmentation causes collapse of kelp recruitment. *Marine Ecology Progress Series,* 648**,** 111-123.

LAYTON, C., COLEMAN, M. A., MARZINELLI, E. M., STEINBERG, P. D., SWEARER, S. E., VERGÉS, A., WERNBERG, T. & JOHNSON, C. R. 2020b. Kelp Forest Restoration in Australia. *Frontiers in Marine Science,* 7.

LAYTON, C. & JOHNSON, C. R. 2021. Assessing the feasibility of restoring giant kelp forests in Tasmania.

LAYTON, C., SHELAMOFF, V., CAMERON, M. J., TATSUMI, M., WRIGHT, J. T. & JOHNSON, C. R. J. P. O. 2019. Resilience and stability of kelp forests: The importance of patch dynamics and environment-engineer feedbacks. 14**,** e0210220.

LEINAAS, H. & CHRISTIE, H. 1996. Effects of removing sea urchins (*Strongylocentrotus droebachiensis*): Stability of the barren state and succession of kelp forest recovery in the east Atlantic. *Oecologia,* 105**,** 524-536.

LESTER, S. E., HALPERN, B. S., GRORUD-COLVERT, K., LUBCHENCO, J., RUTTENBERG, B. I., GAINES, S. D., AIRAMÉ, S. & WARNER, R. R. 2009. Biological effects within no-take marine reserves: a global synthesis. *Marine Ecology Progress Series,* 384**,** 33-46.

LI, J., WEINBERGER, F., DE NYS, R., THOMAS, T. & EGAN, S. 2023. A pathway to improve seaweed aquaculture through microbiota manipulation. *Trends in Biotechnology,* 41**,** 545-556.

LING, S. D. 2008. Range expansion of a habitat-modifying species leads to loss of taxonomic diversity: a new and impoverished reef state. *Oecologia,* 156**,** 883-894.

LING, S. D., JOHNSON, C. R., FRUSHER, S. D. & RIDGWAY, K. R. 2009. Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift. *Proc. Nat. Acad. Sci. USA,* 106**,** 22341-22345.

LING, S. D., KRIEGISCH, N., WOOLLEY, B. & REEVES, S. E. J. E. 2019. Density-dependent feedbacks, hysteresis, and demography of overgrazing sea urchins. 100**,** e02577.

LUBCHENCO, J. J. E. 1983. Littornia and Fucus: effects of herbivores, substratum heterogeneity, and plant escapes during succession. 64**,** 1116-1123.

LUCREZI, S. 2021. Characterising potential participants in kelp monitoring in the recreational diving community: A comparative study of South Africa and New Zealand. *Global Ecology and Conservation,* 28**,** e01649.

LUTZ, S. 2023. Into the Blue: Securing a Sustainable Future for Kelp Forests. Nairobi.

MALLINSON, S. 2020. Sussex Video survey 2019-2020.

MANN, K. H. 1973. Seaweeds, their productivity and strategy for growth. *Science,* 182**,** 975-981.

MARZINELLI, E. M., LEONG, M. R., CAMPBELL, A. H., STEINBERG, P. D. & VERGÉS, A. J. R. E. 2016. Does restoration of a habitat-forming seaweed restore associated faunal diversity? 24**,** 81-90.

MARZINELLI, E. M., QIU, Z., DAFFORN, K. A., JOHNSTON, E. L., STEINBERG, P. D., MAYER-PINTO, M. J. N. B. & MICROBIOMES 2018. Coastal urbanisation affects microbial communities on a dominant marine holobiont. 4**,** 1-7.

MARZLOFF, M. P., DAMBACHER, J. M., JOHNSON, C. R., LITTLE, L. R. & FRUSHER, S. D. J. E. M. 2011. Exploring alternative states in ecological systems with a qualitative analysis of community feedback. 222**,** 2651-2662.

MCPHERSON, M. L., FINGER, D. J. I., HOUSKEEPER, H. F., BELL, T. W., CARR, M. H., ROGERS-BENNETT, L. & KUDELA, R. M. 2021. Large-scale shift in the structure of a kelp forest ecosystem co-occurs with an epizootic and marine heatwave. *Communications Biology,* 4**,** 298.

MEYERS, D., ALLIANCE, C. F., BOHORQUEZ, J., CUMMING, B., EMERTON, L., RIVA, M., FUND, U. J. S. & VICTURINE, R. J. C. F. A. 2020. Conservation finance: a framework. 1-45.

MOORE, P. G. 1971. The nematode fauna associated with holdfasts of kelp (*Laminaria hyperborea*) in northeast Britain. *J. Mar. Biol. Assoc. UK,* 51**,** 589-604.

MOORE, P. G. 1973. The kelp fauna of northeast Britain. II. Multivariate classification: Turbidity as an ecological factor. *J. Exp. Mar. Biol. Ecol.,* 13**,** 127-163.

MORK, M. 1996. The effect of kelp in wave damping. *Sarsia,* 80**,** 323-327.

MORRIS, R. L., GRAHAM, T. D. J., KELVIN, J., GHISALBERTI, M. & SWEARER, S. E. 2020a. Kelp beds as coastal protection: wave attenuation of Ecklonia radiata in a shallow coastal bay. *Annals of Botany,* 125**,** 235-246.

MORRIS, R. L., HALE, R., STRAIN, E. M. A., REEVES, S. E., VERGÉS, A., MARZINELLI, E. M., LAYTON, C., SHELAMOFF, V., GRAHAM, T. D. J., CHEVALIER, M. & SWEARER, S. E. 2020b. Key Principles for Managing Recovery of Kelp Forests through Restoration. *BioScience,* 70**,** 688-698.

MORRIS, R. L., KONLECHNER, T. M., GHISALBERTI, M. & SWEARER, S. E. 2018. From grey to green: Efficacy of eco-engineering solutions for nature-based coastal defence. *Glob Chang Biol,* 24**,** 1827-1842.

MOY, F., CHRISTIE, H., STTEN, H., STÅLNACKE, P., AKSNES, D., ALVE, E., AURE, J., BEKKBY, T., FREDRIKSEN, S. & GITMARK, J. 2009. Sluttrapport Fra Sukkertareprosjektet 2005-2008. Final report from the Sugar Kelp Project 2005-2008.

MOY, F. E. & CHRISTIE, H. 2012. Large-scale shift from sugar kelp (*Saccharina latissima*) to ephemeral algae along the south and west coast of Norway. *Mar. Biol. Res.,* 8**,** 309-321.

NORTH, W. J. J. J. O. T. F. B. O. C. 1976. Aquacultural techniques for creating and restoring beds of giant kelp, Macrocystis spp. 33**,** 1015-1023.

O'BRIEN, J. M. & SCHEIBLING, R. E. 2018. Turf wars: competition between foundation and turf-forming species on temperate and tropical reefs and its role in regime shifts. *Marine Ecology Progress Series,* 590**,** 1-17.
PALUMBI, S. R. J. E. A. 2003. Population genetics, demographic connectivity, and the design of marine reserves. 13**,** 146-158.
PARADA, G. M., TELLIER, F. & MARTÍNEZ, E. A. J. B. M. 2016. Spore dispersal in the intertidal kelp Lessonia spicata: Macrochallenges for the harvested Lessonia species complex at microscales of space and time. 59**,** 283-289.
PARKE, M. 1948. Studies on British Laminariaceae. I. Growth in *Laminaria Saccharina* (L.) Lamour. *Journal of the Marine Biological Association of the United Kingdom,* 3**,** 651-709.
PEDERSEN, M. F., NEJRUP, L. B., PEDERSEN, T. M. & FREDRIKSEN, S. 2014. Sub-canopy light conditions only allow low annual net productivity of epiphytic algae on kelp Laminaria hyperborea. *Marine Ecology Progress Series,* 516**,** 163-176.
PERKOL-FINKEL, S. & AIROLDI, L. 2010. Loss and Recovery Potential of Marine Habitats: An Experimental Study of Factors Maintaining Resilience in Subtidal Algal Forests at the Adriatic Sea. *PLoS ONE,* 5**,** e10791.
PERT, C. G., SWEARER, S. E., DWORJANYN, S., KRIEGISCH, N., TURCHINI, G. M., FRANCIS, D. S. & DEMPSTER, T. J. A. E. I. 2018. Barrens of gold: gonad conditioning of an overabundant sea urchin. 10**,** 345-361.
PESSARRODONA, A., FOGGO, A. & SMALE, D. A. 2019. Can ecosystem functioning be maintained despite climate-driven shifts in species composition? Insights from novel marine forests. *Journal of Ecology,* 107**,** 91-104.
PESSARRODONA, A., MOORE, P. J., SAYER, M. D. J. & SMALE, D. A. 2018. Carbon assimilation and transfer through kelp forests in the NE Atlantic is diminished under a warmer ocean climate. *Global Change Biology,* 24**,** 4386-4398.
PIAZZI, L., CECCHERELLI, G. J. E., COASTAL & SCIENCE, S. 2019. Effect of sea urchin human harvest in promoting canopy forming algae restoration. 219**,** 273-277.
QIU, Z., COLEMAN, M. A., PROVOST, E., CAMPBELL, A. H., KELAHER, B. P., DALTON, S. J., THOMAS, T., STEINBERG, P. D. & MARZINELLI, E. M. J. P. O. T. R. S. B. 2019. Future climate change is predicted to affect the microbiome and condition of habitat-forming kelp. 286**,** 20181887.
REED, D. C. & FOSTER, M. S. 1984. The effects of canopy shading on algal recruitment and growth in a giant kelp forest. *Ecology,* 65**,** 937-948.
REED, D. C., SCHROETER, S. C. & HUANG, D. 2006. An experimental investigation of the use of artificial reefs to mitigate the loss of giant kelp forest habitat.
REED, D. C., SCHROETER, S. C. & PAGE, M. 2019. Annual Report of the Status of Condition C: Kelp Reef Mitigation. *Submitted to the California Coastal Commission.* Marine Science Institute, University of California Santa Barbara.
REED, D. C., SCHROETER, S. C., AND PAGE, M. 2017. Annual Report of the Status of Condition C: Kelp Reef Mitigation. San Diego, CA: University of California.
REEVES, S. E. 2018. *Mechanisms of ecosystem stability for kelp beds in urban environments.* University of Tasmania.
REYNOLDS, L. K., MCGLATHERY, K. J. & WAYCOTT, M. J. P. O. 2012. Genetic diversity enhances restoration success by augmenting ecosystem services. 7**,** e38397.
RINDE, E. & SJØTUN, K. 2005. Demographic variation in the kelp *Laminaria hyperborea* along a latitudinal gradient. *Marine Biology,* 146**,** 1051-1062.
ROHDE, S., MOLIS, M. & WAHL, M. J. J. O. E. 2004. Regulation of anti-herbivore defence by Fucus vesiculosus in response to various cues. 1011-1018.
RUIZ-JAEN, M. C. & MITCHELL AIDE, T. 2005. Restoration success: how is it being measured? *Restoration ecology,* 13**,** 569-577.
RUTHERFORD, K., COX, T. J. W. & ATMOSPHERE 2009. Nutrient trading to improve and preserve water quality. 17**,** 12-13.
SALLAND, N. & SMALE, D. 2021. Spatial variation in the structure of overwintering, remnant *Saccorhiza polyschides* sporophytes and their associated assemblages. *Journal of the Marine Biological Association of the United Kingdom,* 101**,** 639-648.
SANDERSON, J. C., IBBOTT, S. & FOSTER, J. 2003. Restoration of String Kelp (*Macrocystis pyrifera*) habitat on Tasmania's east and south coasts. *Final report to Natural Heritage Trust for Seacare.* Seacare.
SAUNDERS, M. I., DOROPOULOS, C., BAYRAKTAROV, E., BABCOCK, R. C., GORMAN, D., EGER, A. M., VOZZO, M. L., GILLIES, C. L., VANDERKLIFT, M. A., STEVEN, A. D. L., BUSTAMANTE, R. H. & SILLIMAN, B. R. 2020. Bright Spots in Coastal Marine Ecosystem Restoration. *Curr Biol,* 30**,** R1500-R1510.

SCHIEL, D. R. & FOSTER, M. S. 2006. The Population Biology of Large Brown Seaweeds: Ecological Consequences of Multiphase Life Histories in Dynamic Coastal Environments. *Annual Review of Ecology, Evolution, and Systematics,* 37**,** 343-372.

SCHIEL, D. R. & FOSTER, M. S. 2015. *The Biology and Ecology of Giant Kelp Forests*, University of California Press.

SHARMA, R., SWEARER, S. E., MORRIS, R. L. & STRAIN, E. M. A. 2021. Testing the efficacy of sea urchin exclusion methods for restoring kelp. *Mar Environ Res,* 170**,** 105439.

SHEARS, N. T. & BABCOCK, R. C. 2002. Marine reserves demonstrate top-down control of community structure on temperate reefs. *Oecologia,* 132**,** 131-142.

SHEARS, N. T. & BABCOCK, R. C. J. M. E. P. S. 2003. Continuing trophic cascade effects after 25 years of no-take marine reserve protection. 246**,** 1-16.

SHELAMOFF, V., LAYTON, C., TATSUMI, M., CAMERON, M. J., WRIGHT, J. T. & JOHNSON, C. R. J. R. E. 2019. Ecosystem engineering by a canopy-forming kelp facilitates the recruitment of native oysters. 27**,** 1442-1451.

SILLIMAN, B. R., SCHRACK, E., HE, Q., COPE, R., SANTONI, A., VAN DER HEIDE, T., JACOBI, R., JACOBI, M. & VAN DE KOPPEL, J. J. P. O. T. N. A. O. S. 2015. Facilitation shifts paradigms and can amplify coastal restoration efforts. 112**,** 14295-14300.

SMAAL, A. C., FERREIRA, J. G., GRANT, J., PETERSEN, J. K. & STRAND, Ø. 2019. *Goods and services of marine bivalves*, Springer Nature.

SMALE, D. A., BURROWS, M. T., EVANS, A. J., KING, N., SAYER, M. D. J., YUNNIE, A. L. E. & MOORE, P. J. 2016. Linking environmental variables with regional-scale variability in ecological structure and standing stock of carbon within kelp forests in the United Kingdom. *Marine Ecology Progress Series,* 542**,** 79-95.

SMALE, D. A., BURROWS, M. T., MOORE, P. J., O' CONNOR, N. & HAWKINS, S. J. 2013. Threats and knowledge gaps for ecosystem services provided by kelp forests: a northeast Atlantic perspective. *Ecology and Evolution,* 3**,** 4016–4038.

SMALE, D. A., EPSTEIN, G., HUGHES, E., MOGG, A. O. M. & MOORE, P. J. 2020. Patterns and drivers of understory macroalgal assemblage structure within subtidal kelp forests. *Biodiversity and Conservation,* 29**,** 4173-4192.

SMALE, D. A., KING, N. G., JACKSON-BUÉ, M. & MOORE, P. J. 2022. Quantifying use of kelp forest habitat by commercially important crustaceans in the United Kingdom. *Journal of the Marine Biological Association of the United Kingdom,* 102**,** 627-634.

SMALE, D. A., LANGLOIS, T. J., KENDRICK, G. A., MEEUWIG, J. J. & HARVEY, E. S. 2011. From fronds to fish: the use of indicators for ecological monitoring in marine benthic ecosystems, with particular reference to the temperate nearshore system of Western Australia. *Reviews in Fish Biology and Fisheries,* 21**,** 311–337.

SMALE, D. A. & MOORE, P. J. 2017. Variability in kelp forest structure along a latitudinal gradient in ocean temperature. *Journal of Experimental Marine Biology and Ecology,* 486**,** 255-264.

SMALE, D. A., PESSARRODONA, A., KING, N. & MOORE, P. J. 2021. Examining the production, export, and immediate fate of kelp detritus on open-coast subtidal reefs in the Northeast Atlantic. *Limnology and Oceanography,* n/a.

SMALE, D. A. & VANCE, T. 2015. Climate-driven shifts in species distributions may exacerbate the impacts of storm disturbances on northeast Atlantic kelp forests. *Marine and Freshwater Research,* 67**,** 65-74.

SMALE, D. A., WERNBERG, T., YUNNIE, A. L. E. & VANCE, T. 2015. The rise of *Laminaria ochroleuca* in the Western English Channel (UK) and preliminary comparisons with its competitor and assemblage dominant *Laminaria hyperborea*. *Marine Ecology,* 36**,** 1033-1044.

SMITH, C. J., VERDURA, J., PAPADOPOULOU, N., FRASCHETTI, S., CEBRIAN, E., FABBRIZZI, E., MONSERRAT, M., DRAKE, M., BIANCHELLI, S. & DANOVARO, R. in press. A decision-support framework for the restoration of Cystoseira sensu lato forests. *Frontiers in Marine Science,* 10**,** 1159262.

STENECK, R. S., GRAHAM, M. H., BOURQUE, B. J., CORBETT, D., ERLANDSON, J. M., ESTES, J. A. & TEGNER, M. J. 2002. Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation,* 29**,** 436-459.

STRAND, H. K., CHRISTIE, H., FAGERLI, C. W., MENGEDE, M. & MOY, F. J. E. E. 2020. Optimizing the use of quicklime (CaO) for sea urchin management—A lab and field study. 143**,** 100018.

SUEBSANGUAN, S., STRAIN, E. M. A., MORRIS, R. L. & SWEARER, S. E. 2021. Optimizing the initial cultivation stages of kelp *Ecklonia radiata* for restoration. *Restoration Ecology,* 29.

TEAGLE, H., HAWKINS, S. J., MOORE, P. J. & SMALE, D. A. 2017. The role of kelp species as biogenic habitat formers in coastal marine ecosystems. *Journal of Experimental Marine Biology and Ecology,* 492**,** 81-98.

TEAGLE, H., MOORE, P. J., JENKINS, H. & SMALE, D. A. 2018. Spatial variability in the diversity and structure of faunal assemblages associated with kelp holdfasts (Laminaria hyperborea) in the northeast Atlantic. *PLOS ONE,* 13**,** e0200411.

TEAGLE, H. & SMALE, D. A. 2018. Climate-driven substitution of habitat-forming species leads to reduced biodiversity within a temperate marine community. *Diversity and Distributions,* 24**,** 1367-1380.

TOOHEY, B., KENDRICK, G. A., WERNBERG, T., PHILLIPS, J. A., MALKIN, S. & PRINCE, J. 2004. The effects of light and thallus scour from *Ecklonia radiata* canopy on an associated foliose algal assemblages: the importance of photoacclimation. *Marine Biology,* 144**,** 1019-1027.

TOTH, G. B. & PAVIA, H. J. P. O. T. N. A. O. S. 2000. Water-borne cues induce chemical defense in a marine alga (Ascophyllum nodosum). 97**,** 14418-14420.

TRACEY, S. R., BAULCH, T., HARTMANN, K., LING, S. D., LUCIEER, V., MARZLOFF, M. P. & MUNDY, C. 2015a. Systematic culling controls a climate driven, habitat modifying invader. *Biological Invasions,* 17**,** 1885-1896.

TRACEY, S. R., BAULCH, T., HARTMANN, K., LING, S. D., LUCIEER, V., MARZLOFF, M. P. & MUNDY, C. J. B. I. 2015b. Systematic culling controls a climate driven, habitat modifying invader. 17**,** 1885-1896.

TURNER, R. & BOYER, M. J. E. E. 1997. Mississippi River diversions, coastal wetland restoration/creation and an economy of scale. 8**,** 117-128.

VAN KATWIJK, M. M., THORHAUG, A., MARBÀ, N., ORTH, R. J., DUARTE, C. M., KENDRICK, G. A., ALTHUIZEN, I. H., BALESTRI, E., BERNARD, G. & CAMBRIDGE, M. L. J. J. O. A. E. 2016. Global analysis of seagrass restoration: the importance of large-scale planting. 53**,** 567-578.

VANDERKLIFT, M., STEVEN, A., MARCOS-MARTINEZ, R., GORMAN, D. J. F. R., DEVELOPMENT CORPORATION, C. O. & ATMOSPHERE, F. P. 2018. Achieving carbon offsets through blue carbon: a review of needs and opportunities relevant to the Australian seafood industry. 60**,** R126.

VASQUEZ, J., MCPEAK, R. J. C. F. & GAME 1998. A new tool for kelp restoration. 84**,** 149-158.

VÁSQUEZ, J. & TALA, F. J. J. A. P. 1995. Experimental repopulation of Lessonia nigrescens (Phaeophyta, Laminariales) in intertidal areas of northern Chile. 7**,** 347-349.

VERDURA, J., SALES, M., BALLESTEROS, E., CEFALÌ, M. E. & CEBRIAN, E. 2018. Restoration of a canopy-forming alga based on recruitment enhancement: methods and long-term success assessment. *Frontiers in Plant Science,* 9**,** 1832.

VERGÉS, A., CAMPBELL, A. H., WOOD, G., KAJLICH, L., EGER, A. M., CRUZ, D., LANGLEY, M., BOLTON, D., COLEMAN, M. A., TURPIN, J., CRAWFORD, M., COOMBES, N., CAMILLERI, A., STEINBERG, P. D. & MARZINELLI, E. M. 2020. Operation Crayweed: Ecological and sociocultural aspects of restoring Sydney's underwater forests. *Ecological Management & Restoration,* 21**,** 74-85.

VERGÉS, A., DOROPOULOS, C., MALCOLM, H. A., SKYE, M., GARCIA-PIZÁ, M., MARZINELLI, E. M., CAMPBELL, A. H., BALLESTEROS, E., HOEY, A. S., VILA-CONCEJO, A., BOZEC, Y.-M. & STEINBERG, P. D. 2016. Long-term empirical evidence of ocean warming leading to tropicalization of fish communities, increased herbivory, and loss of kelp. *Proceedings of the National Academy of Sciences,* 113**,** 13791-13796.

VERGÉS, A., STEINBERG, P. D., HAY, M. E., POORE, A. G., CAMPBELL, A. H., BALLESTEROS, E., HECK JR, K. L., BOOTH, D. J., COLEMAN, M. A. & FEARY, D. A. J. P. O. T. R. S. B. B. S. 2014. The tropicalization of temperate marine ecosystems: climate-mediated changes in herbivory and community phase shifts. 281**,** 20140846.

VOERMAN, S. E., LLERA, E. & RICO, J. M. 2013. Climate driven changes in subtidal kelp forest communities in NW Spain. *Marine Environmental Research,* 90**,** 119-127.

WALKER, F. T. 1953. Distribution of Laminariaceae around Scotland. *Journal du Conseil***,** 160-166.

WALKER, F. T. & RICHARDSON, W. D. 1956. The Laminariaceae off North Shapinsay, Orkney Islands; changes from 1947 to 1955. *Journal of Marine Research,* 15**,** 123-133.

WERNBERG, T., BENNETT, S., BABCOCK, R. C., DE BETTIGNIES, T., CURE, K., DEPCZYNSKI, M., DUFOIS, F., FROMONT, J., FULTON, C. J., HOVEY, R. K., HARVEY, E. S., HOLMES, T. H., KENDRICK, G. A., RADFORD, B., SANTANA-GARCON, J., SAUNDERS, B. J., SMALE, D. A., THOMSEN, M. S., TUCKETT, C. A., TUYA, F., VANDERKLIFT, M. A. & WILSON, S. K. 2016. Climate driven regime shift of a temperate marine ecosystem. *Science,* 353**,** 169-172.

WERNBERG, T., KENDRICK, G. A. & TOOHEY, B. D. 2005. Modification of the physical environment by an *Ecklonia radiata* (Laminariales) canopy and its implications to associated foliose algae. *Aquatic Ecology,* 39**,** 419-430.

WERNBERG, T., KRUMHANSL, K., FILBEE-DEXTER, K. & PEDERSEN, M. F. 2019a. Chapter 3 - status and trends for the world's kelp forests. *In:* SHEPPARD, C. (ed.) *World Seas: an Environmental Evaluation (Second Edition).* Academic Press, Cambridge, Massachusetts, USA.

WERNBERG, T., KRUMHANSL, K., FILBEE-DEXTER, K. & PEDERSEN, M. F. 2019b. Status and trends for the world's kelp forests. *World seas: An environmental evaluation.* Elsevier.

WERNER, A. & KRAAN, S. 2004. Review of the potential mechanisation of kelp harvesting in Ireland. Marine Institute and Taighde Mara Teo report. Marine Environment and Health Series, No. 17. pp. 52.

WESTERMEIER, R., MURÚA, P., PATIÑO, D. J., MUÑOZ, L., ATERO, C. & MÜLLER, D. G. 2014. Repopulation techniques for Macrocystis integrifolia (Phaeophyceae: Laminariales) in Atacama, Chile. *Journal of Applied Phycology,* 26**,** 511-518.

WESTERMEIER, R., MURÚA, P., PATIÑO, D. J., MUÑOZ, L. & MÜLLER, D. G. J. J. O. A. P. 2016. Holdfast fragmentation of Macrocystis pyrifera (integrifolia morph) and Lessonia berteroana in Atacama (Chile): a novel approach for kelp bed restoration. 28**,** 2969-2977.

WHITAKER, S. G., SMITH, J. R. & MURRAY, S. N. J. R. E. 2010. Reestablishment of the southern California rocky intertidal brown alga, Silvetia compressa: an experimental investigation of techniques and abiotic and biotic factors that affect restoration success. 18**,** 18-26.

WILDING, C., TILLIN, H., CORRIGAN, S. E., STUART, E., A., A. I., FELSTEAD, P., LUBELSKI, A., BURROWS, M. & D., S. 2021. Seaweed aquaculture and mechanical harvesting: an evidence review to support sustainable management. *Natural England Commissioned Reports.* Natural England.

WILLIAMS, S. L., AMBO-RAPPE, R., SUR, C., ABBOTT, J. M. & LIMBONG, S. R. J. P. O. T. N. A. O. S. 2017. Species richness accelerates marine ecosystem restoration in the Coral Triangle. 114**,** 11986-11991.

WILSON, K. A., LULOW, M., BURGER, J., FANG, Y. C., ANDERSEN, C., OLSON, D., O'CONNELL, M. & MCBRIDE, M. F. J. J. O. A. E. 2011. Optimal restoration: accounting for space, time and uncertainty. 48**,** 715-725.

WOOD, G., MARZINELLI, E. M., COLEMAN, M. A., CAMPBELL, A. H., SANTINI, N. S., KAJLICH, L., VERDURA, J., WODAK, J., STEINBERG, P. D., VERGÉS, A. J. M. & RESEARCH, F. 2019. Restoring subtidal marine macrophytes in the Anthropocene: trajectories and future-proofing. 70**,** 936-951.

WOOD, G., MARZINELLI, E. M., VERGÉS, A., CAMPBELL, A. H., STEINBERG, P. D., COLEMAN, M. A. & HE, Q. 2020. Using genomics to design and evaluate the performance of underwater forest restoration. *Journal of Applied Ecology,* 57**,** 1988-1998.

WOODCOCK, P., O'LEARY, B. C., KAISER, M. J., PULLIN, A. S. J. F. & FISHERIES 2017. Your evidence or mine? Systematic evaluation of reviews of marine protected area effectiveness. 18**,** 668-681.

WOODWARD, F. N. 1951. The Scottish Seaweed Research Association. *Journal of the Marine Biological Association of the United Kingdom,* 29**,** 719-725.

XIAO, X., AGUSTÍ, S., YU, Y., HUANG, Y., CHEN, W., HU, J., LI, C., LI, K., WEI, F., LU, Y., XU, C., CHEN, Z., LIU, S., ZENG, J., WU, J. & DUARTE, C. M. 2021. Seaweed farms provide refugia from ocean acidification. *Science of The Total Environment,* 776**,** 145192.

YAMAMOTO, M., FUKUSHIMA, M., KISO, E., KATO, T., SHIBUYA, M., HORIYA, S., NISHIDA, A., OTSUKA, K. & KOMAI, T. 2010. Application of iron humates to barren ground in a coastal area for restoring seaweed beds. *Journal of chemical engineering of Japan,* 43**,** 627-634.

YESSON, C., BUSH, L. E., DAVIES, A. J., MAGGS, C. A. & BRODIE, J. 2015. The distribution and environmental requirements of large brown seaweeds in the British Isles. *J Mar Biol Assoc UK,* 95**,** 669-680.

YU, Y. Q., ZHANG, Q. S., TANG, Y. Z., ZHANG, S. B., LU, Z. C., CHU, S. H. & TANG, X. X. 2012. Establishment of intertidal seaweed beds of Sargassum thunbergii through habitat creation and germling seeding. *Ecological Engineering,* 44**,** 10-17.

ZAHAWI, R. A., REID, J. L. & HOLL, K. D. J. R. E. 2014. Hidden costs of passive restoration. 22**,** 284-287.

6. Literature summary tables

Table 1. Key information on UK marine forest species

Life history strategies, habitat, range, and status of key UK marine forest forming species. Adapted from Wilding et al. (2021). Sources: MarLIN (www.marlin.ac.uk); Burrows et al. (2018).

Names	Growth rate / season	Max. typical age (years)	Age at maturity	Reproductive season	Dispersal potential	Habitat	Current range	Predicted changes
A. esculenta Atlantic Wakame, Dabberlocks	Perennial. From base of blade. Peaks in spring (April – May) at 20 cm / month; erodes from June – winter leaving only midrib stalk	4-7 years	8-14 months	Nov – March, spores released from sporophylls at the base of the plant. Recruits appear in spring	Low: 2-10m. Potentially further from mature drift plants	On bedrock and boulders subject to high wave exposure. From the lower intertidal into the shallow subtidal	Greenland / Norway to Brittany. UK populations represent the southerly range limit	Current/future range contraction and loss of trailing (southern) edge populations
A. nodosum Knotted wrack, Egg wrack	Perennial growing from the tip of the blade or regenerating from the base. Growth rate increases with age from 0.2 cm/year in the first year to 8-15cm/year when mature. Growth peaks in spring/summer, slowing in autumn/winter	10-20 years per frond; >60 years per clump	>5 years	Vegetative reproduction from basal shoots is more common than sexual reproduction, which peaks in March-April. Up to 2.5 x109 eggs m^2/year may be produced by mature stands. However early (first year) mortality > 99.9%	Very low. Facilitated beneath adult canopy.	Intertidally on sheltered to moderately exposed shores, attached to bedrock, boulders, or cobbles	Portugal to Norway	Found towards trailing edge in UK. Losses or declines of southern populations predicted in coming decades
F. serratus Toothed wrack, Serrated wrack	Short lived perennial, growth occurs from the tips. Peak growth is spring-early summer, at rates of 4-12 cm/year	2-5 years	1-2 years	Gametes are released from the tips of the frond between late spring-autumn, peaking Aug-Oct	Low. Eggs settle close to parent plants, moving from the water column to attach to the substratum within a few hours of release.	From sheltered to moderately exposed shores on hard substrata (bedrock, cobbles) on the lower shore	Iceland to Portugal	Found towards trailing edge in UK. Losses or declines of southern populations predicted in coming decades
F. vesiculosus Bladder wrack	Short lived perennial, growth occurs from the tips and peaks in summer. Peak growth rates are 0.14 cm/day	2-5 years	1-2 years	Each plant can release over one million eggs from the tips of the frond between winter – late summer, peaking May-June.	Fair, up to 100's of km. Bladders allow floating mature drift plants to release spores over large distances.	Intertidally, from relatively sheltered shores on hard substrata (bedrock, cobbles, and pebbles) from the	Norway to Portugal	Found towards trailing edge in UK. Losses or declines of southern populations predicted in coming decades

						mid shore. Tolerant of reduced salinity.		
H. elongata Sea spaghetti, Thongweed	Bi-annual. The 'button' grows rapidly in diameter during the spring in its first year, producing reproductive 'straps' in autumn which grow rapidly – up to 16mm/day - in Feb-May the following year	2-3 years	9-14 months	Reproductive 'straps' release gametes from June-winter, recruits appear from March	Low. Zygotes are heavy and large, settling close to parent plants	From semi-exposed shores on hard substrata (bedrock, boulders) from the lower shore to shallow subtidal.	Norway to Portugal	Found towards centre of range in UK
L. digitata Oarweed	Perennial. From base of blade. Peaks in winter (Feb – July). Slower in summer (Aug – Jan). Mean growth rate of 1.3 cm / day during max growth season	4-6 years	18-20 months	Sorus material forms on blade year-round, peaking in July-Aug and Nov-Dec. Recruits appear year-round peaking in spring and autumn. Spores must settle in high density (within 1mm of each other) for fertilization to occur	Low – fair. Zoospores may be transported 200-600 m from parent plants, settling after 24 hours	Moderate to high wave exposure, on hard substrata (bedrock, boulders) on the lower intertidal and sublittoral fringe	Greenland to France	Declining in France, exhibiting reduced growth rates in southern England. Predicted to undergo range contraction with loss of trailing (southern) edge populations
L. hyperborea Forest kelp, Cuvie, Tangle	Perennial growing from the base of the blade. Peaks in winter (Nov – June), slower in summer. Growth rate of 0.94 cm / day during max growth season	11-20 years	2-3 years	Sorus material forms on blade year-round, peaking in winter (Sept – April). Recruits appear year-round peaking in spring	Fair, at least 5 km from the parent plant, settling after 24 hours, although spores must settle in high density (within 1mm of each other) for fertilization to occur	Moderate to high wave exposure, on hard substrata (bedrock, boulders) from the sublittoral fringe to depth	Russia and Norway to Portugal	Found towards centre of range in UK
L. ochroleuca Golden kelp	Perennial growing from the base of the blade. Growth peaks in late spring/summer	8-10 years	1-2 years	Sorus material forms on blade from spring to winter, peaking in late summer. Recruits found all year, peaking in summer	Unknown, likely to be low - fair	From sheltered to moderately exposed shores, on hard substrata (bedrock, boulders) from the sublittoral fringe to depth	UK and Ireland to Morocco, Azores and Mediterranean	At its leading (Irving and Northern) range edge in the UK and Ireland, predicted to proliferate and expand polewards
S. latissima Sugar kelp	Annual or short-lived perennial growing from the base of the blade. Peak growth in late	2-4 years	8-15 months	Sorus material forms along the centre of the blade year-round, peaking Oct – April.	Likely to be low (meters to kilometres), with spores short lived spores and fertilization	Sheltered to moderately exposed shores, on hard substrata (bedrock,	Norway to Portugal	Found towards centre of range in UK

	winter – spring (typically 1.1-cm / day). Slower in summer.			Recruits appear in winter - spring peaking in Dec and Jan	dependant on high settlement density	boulders, cobbles, and pebbles) from the sublittoral fringe to depth		
S. polyschides Furbellows	Annual. Peak growth in late spring (6.2cm/week), senescing from mid-summer, absent by late winter	8-18 months	8-14 months	Sorus material forms on the sporophylls and stipe, from which spores are released in summer-autumn. Recruits appear year-round, peaking around June. Recruitment is seasonal and may be blocked by other algae	Large numbers (>1,000,000) of zoospores remain in the water column for 24 hours and may be transported at least 200 m from the parent	Sheltered to moderately exposed shores, on hard substrata (bedrock, boulders, cobbles, and pebbles) from the sublittoral fringe to depth	Norway to Morocco and the Mediterranean	Found towards centre of range in UK. Opportunistic species predicted to increase in abundance due to ocean warming and increased disturbance

Table 2. Restoration methods evaluation

Selected examples of restoration methods, benefits, and limitations. Adapted from Earp et al. (2022), Morris et al. (2020b), and Fredriksen et al. (2020).

Restoration Technique	Benefits	Limitations	Candidate UK species	Example species	References
Transplanting adult and/or juvenile individuals sourced from either a donor population, a laboratory culture, or opportunistic drift/beach cast individuals are installed at restoration sites using an array of techniques	Adult conspecifics can immediately modify environmental conditions to improve recruitment Highly accurate location Established history of use Often successful	High post-transplant mortality / lack or re-attachment High cost, labour intensive Limited scalability Potential impact on donor populations	Morphologically similar species include: *L. hyperborea, L. ochroleuca, L. digitata*	*E. radiata,* a stipate kelp, in Australia	Layton et al. (2019), (2020b)
			F. vesiculosus is morphologically similar	*P. comosa* in Sydney, Australia	Campbell et al. (2014)
			A. esculenta could be suitable for genetic selection although transplanting may not be feasible	*M. pyrifera* (with genetic selection) in Tasmania, Australia	Layton et al. (2021)
Green gravel (technically a type of transplant) rocks are seeded and grown on in the lab to 2-3cm long before out planting	Scalable Relatively low cost (no need for divers)	Yet to be trialled in a range of environmental contexts (although work is underway) e.g. high water flow or wave energy	*S. latissima* and potentially other UK species	*S. latissima* in Norway	Fredriksen et al. (2020); Green Gravel Action Group
Seeding recruitment is enhanced at potential restoration sites through the installation of translocated reproductive tissues/bodies, and the dispersal of early life stage cultures	Relatively low cost Scalable (large volumes of spores released) Potential impact on donor populations	High mortality of early life stages (e.g. one in 100,000 survival) Can be labour intensive if involving divers Empty spore bags require collection	*L. hyperborea, L. ochroleuca, L. digitata, A. esculenta* (and to a partial extent *Saccharina latissima*) could all provide reproductive material with minimal damage to donor populations, provided early survival is not unfeasibly low	*Macrocystis* spp. in California, USA *Macrocystis integrifolia* in Chile	North (1976) Westermeier et al. (2014)
Artificial habitat structures installed on the seabed mimic natural substrate. Often used in conjunction with other interventions such as transplantation and/or seeding	Creates additional habitat Potentially scalable (100,000m^2) Can be engineered to suit conditions (e.g.	Novel conditions create potential for increases settlement of Invasive non-native species Costly	Potentially any UK species, depending on location	*M. pyrifera* in California, USA	Reed et al. (2019)

Restoration Technique	Benefits	Limitations	Candidate UK species	Example species	References
	inserting transplants), potentially reducing costs "Green engineering" can mitigate damage caused by installation of infrastructure Established history of use Does not require costly diving	Requires supply of propagules		*Cystoseira* spp. in the Adriatic Sea (Mediterranean)	Falace et al. (2006)
				Sargassum horneri and *Ecklonia cava* in Korea	Choi et al. (2000)
Competitor exclusion/removal of a species (e.g. turf algae) that would otherwise compete with forest species for resources or inhibit their recruitment. Often used in conjunction with other interventions such as transplantation and/or seeding.	Established history of use (in Japan) Highly successful (Earp)	Scalability (for labour intensive diver methods) Requires supply of propagules Not widely used outside of Japan	Potentially any UK species	*S. latissima* in Norway	Moy et al. (2009), Moy and Christie (2012)
				M. pyrifera in Tasmania	Sanderson et al. (2003)
Herbivore exclusion/removal installation of devices that exclude single or multiple herbivore species, or practices that target herbivore species and remove them.	Established history of use Highly successful (Earp) Market-driven approaches may be scalable / long term	Requires supply of propagules Likely to be temporary or requite ongoing intervention to maintain Scalability (for labour intensive diver methods to maintain cages / cull manually)	Potentially any UK species	*E. radiata* in Australia	Bennett et al. (2017)
				L. hyperborea and *S. latissima* in Norway	Leinaas and Christie (1996)
				Cystoseira spp. in the Mediterranean Sea	Piazzi et al. (2019)
Multiple techniques involve a combination of active techniques to increase the number of individuals and passive techniques to provide a suitable environment for survival.	Potentially more successful	Likelihood of increasing cost with additive methods used will reduce scalability Factors influencing success difficult to isolate by method	Potentially any UK species	*M. pyrifera* in Mexico. Transplanting and competitor removal	Hernandez-Carmona et al. (2000)

Note: Examples listed for individual methods may also have been used in combination with other techniques.

Table 3. Costing of methods

Summary of the average costs of kelp restoration methods, reproduced from Eger et al. (2021) ($/ha). Values converted to British pounds using the 2020 average exchange rate and rounded to the nearest pound.

Restoration Method	Aim	Average cost per hectare
Quickliming	Controlling sea urchins	~£842/ha ($1300 USD)
Manual removal	Controlling sea urchins	~£28,361/ha ($43800 USD)
Seeding	Increase kelp recruitment	~£285,742/ha ($441300 USD)
Transplanting	Increase kelp population	~£367,910/ha ($582100 USD)
Artificial Reef	Increase kelp recruitment	~£384,227/ha ($593400 USD)

Appendix 1 – UK kelp losses survey

To qualitatively explore and potentially identify UK kelp losses, a short written survey was distributed to targeted experts working in this field. This included: The British Phycological Society, Algae-List, Seasearch co-ordinators, The Wildlife Trust marine teams, Project Seagrass, the RanTrans project, a Marine consultancy (MarineSeen), participants at the Coastal Futures 2022 Conference, Statutory Nature Conservation Body staff (Joint Nature Conservation Committee, Environment Agency, Natural England, Natural Resources Wales, Nature Scot), and academic institutions (Scottish Institute for Marine Science, Newcastle University, Swansea University, Marine Biological Association of the UK). The survey was open between 12th January to 7th February 2022. The survey was focussed on identification of losses, which resulted in a respondent bias (i.e. responses from areas where kelp forests were thought to be healthy were not reported). A total of four responses were received through the survey platform with a further two responses submitted by email. Responses are summarised below.

Further data mining was beyond the scope of the survey but would be an essential prerequisite to clearly identify sites prior to initiating restoration pilot studies in the UK.

One respondent was aware of no UK kelp declines. Two responses indicated shifts in species composition in the southwest of England, Lundy and the Isles of Scilly, where *Laminaria ochroleuca* now appears to dominate over *Laminaria hyperborea*, likely driven by ocean warming. In this case, in answer to whether they considered that restoration would be appropriate, one respondent replied, “not now” and the other did not consider restoration action necessary, because “from my observations there are still kelp forests wherever there is suitable physical habitat”. Given that the driver of this shift is a global stressor which is difficult to ameliorate, restoration may only be appropriate if actions were to incorporate genetic selection to boost the resilience of *Laminaria hyperborea*. While this species substitution has been shown to alter associated community composition, habitat facilitation cascades and potentially ecosystem service provision, it is unlikely that conservation managers would deem the benefits of reversing a shift from one kelp species to another, both of which are broadly morphologically similar, worthy of the cost of intervention, or indeed feasible.

Declines of *Alaria esculenta*, also attributed to ocean warming, were also identified in southwest England, along with increases in invasive non-native species *Grateloupia turuturu* and *Caulacanthus okamurae,* with the speculation that this could this be causing losses of other species.

In Scotland, kelp forests were considered to be “quite healthy”, although the presence of the invasive non-native species *Sargassum muticum* was noted in the Firth of Lorne. Also, anecdotal evidence of declines in abundance of *Laminaria hyperborea* from deeper water in the upper Firth of Clyde (Helensburgh) were suggested, potentially driven by poor water quality and turbidity. In answer to whether the respondent considered that restoration would be appropriate, mitigating the water quality issues was identified and an essential measure.

The final respondent highlighted that a biodiversity gap had been identified at the national UK level through Healthy Biologically Diverse Seas Group (HBDSEG), and that UK Marine Strategy indicator development, including acoustic monitoring, was ongoing. This respondent was not aware however of any local declines or losses of kelp.