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An assessment of the utility of green gravel as a kelp restoration tool in wave-exposed intertidal habitats

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Abstract

Kelp forests are being degraded and/or lost in many regions, and as such, interest in active kelp restoration approaches to reinstate forests is growing. 'Green gravel' is a promising new kelp restoration technique that involves seeding small rocks with kelp zoospores, rearing the gametophyte and juvenile sporophyte stages in aquaria before outplanting them at restoration sites. However, to be considered a viable approach to kelp forest restoration, the efficacy of this technique needs to be assessed across a range of environmental contexts and kelp species. Here, we aimed to understand the utility of green gravel as a kelp restoration technique for wave-exposed intertidal shores. Two substrate types – gravel and cobbles – were seeded with *Saccharina latissima*, reared in the aquarium and outplanted at two sites along the north-east coast of England. Outplanted rocks were monitored for retention, and the density and length of *S. latissima*. Juvenile sporophytes persisted on both rock types, although declines in density and variations in length were observed over time. Substrate retention was low, with gravel more likely to be removed from restoration sites compared to cobbles, and all outplanted rocks were lost after eight months. While our initial testing of the green gravel restoration technique on wave-exposed shores was not successful, our results provide important insights for developing/refining the technique and a baseline for comparison for future efforts. However, prior to commencing large-scale kelp restoration in wave-exposed areas using green gravel, further testing of the technique and comparisons with other restoration approaches are needed.

Introduction

Habitat degradation and destruction resulting from human activities have become increasingly common, and can have severe consequences for the structure and functioning of ecosystems (Crain *et al.*, 2009; Wernberg *et al.*, 2024). Coastal marine ecosystems, which are among the most productive and valuable ecosystems in the world (Costanza *et al.*, 1997, 2014), are particularly vulnerable to degradation and/or destruction due to the combined effects of multiple concurrent stressors operating across varying spatiotemporal scales (Harley *et al.*, 2006; Airoldi *et al.*, 2021; Wernberg *et al.*, 2024). Degradation and/or loss of complex, productive and biodiverse habitats can often lead to shifts to less complex, less desirable habitats (Hughes, 1994; Filbee-Dexter and Wernberg, 2018). Positive feedbacks often favour the persistence of the less-complex systems and inhibit natural recovery (Nystrom *et al.*, 2012; Filbee-Dexter and Scheibling, 2014; Filbee-Dexter and Wernberg, 2018). Consequently, interest in restoration as a tool to initiate or accelerate the recovery of habitats that have been degraded or lost is growing. In the marine realm, restoration has somewhat lagged behind terrestrial systems, although advances have been made in multiple habitat types including mangrove forests (Kamali and Hashim, 2011), seagrass meadows (Bull *et al.*, 2004; Marion and Orth, 2010; van Katwijk *et al.*, 2016; Unsworth *et al.*, 2019), coral reefs (Rinkevich, 2005; Young *et al.*, 2012; Boström-Einarsson *et al.*, 2020), oyster reefs (Brumbaugh and Coen, 2009; Richardson *et al.*, 2022), and more recently kelp forests (Westermeier *et al.*, 2016; Fredriksen *et al.*, 2020; Graham *et al.*, 2021; Earp *et al.*, 2022; Miller and Shears, 2022; Eger *et al.*, 2022a).

Kelp (large brown macroalgae of the order Laminariales) dominate temperate and subpolar rocky reefs and are found along up to a third of the world's coastlines (Wernberg *et al.*, 2019; Jayathilakea and Costello, 2021). These forests are highly diverse and productive ecosystems that support a range of ecological functions and ecosystem services (Steneck *et al.*, 2002; Smale *et al.*, 2013; Bennett *et al.*, 2016; Eger *et al.*, 2023), yet despite this, significant declines in kelp abundance have been observed in 40–60% of ecoregions for which long-term data are available (Krumhansl *et al.*, 2016; Wernberg *et al.*, 2019), and future predictions show significant range contractions and local extinctions in many regions (Martínez *et al.*, 2018). Traditionally, passive techniques including mitigating the driver of decline, limiting kelp harvesting and establishing protected areas were employed to conserve kelp forests (Eger *et al.*, 2022b). However, such techniques do not always facilitate kelp forest reestablishment, for example despite improvements in water quality in Sydney, the canopy-forming fucoid *Phyllospora comosa* did not reestablish following declines in the 1970's (Coleman *et al.*, 2008; Campbell *et al.*, 2014). As such, active restoration techniques including transplanting and seeding are necessary to restore kelp forests in some regions (Vásquez and Tala, 1995; Hernandez-Carmona *et al.*, 2000; Campbell *et al.*, 2014; Westermeier *et al.*, 2014; Layton *et al.*, 2021). Active restoration can however be costly, challenging to implement at relevant spatial scales, and has often only been trialled on one species and/or in one environmental context (Earp *et al.*, 2022). In particular, the majority of Laminarian kelp restoration efforts have been undertaken in subtidal areas (Earp *et al.*, 2022), with limited information regarding the suitability of trialled restoration techniques in intertidal environments.

'Green gravel' is a novel kelp restoration technique that aims to overcome some of the challenges facing kelp restoration, particularly cost and scalability (Fredriksen *et al.*, 2020). Simply, the technique involves seeding gravel with kelp, rearing them in aquaria and then outplanting them at restoration sites. The technique was first trialled using the sugar kelp *Saccharina latissima*, with gravel outplanted in a semi-protected area on the Norwegian coast (Fredriksen *et al.*, 2020). The gravel was well retained at the sites and *S. latissima* increased in length and in some cases overgrew the gravel and attached directly to the underlying natural substrate (Fredriksen *et al.*, 2020). However, to be considered a viable approach to kelp forest restoration across a range of environmental contexts, the efficacy of this technique needs to be assessed under different conditions (e.g., wave exposure, tidal heights) and across different kelp species, and the cost of such activities documented.

Along the coastline of the United Kingdom (UK), kelp and other species of canopy-forming macroalgae are estimated to occupy between 8000 to 20,000 km^2 where there is natural or artificial hard substrate, alongside suitable water quality and light (Smale *et al.*, 2013; Yesson *et al.*, 2015). There is emerging evidence that the distribution and abundance of some UK kelp species has changed, for example, the cold-water kelp *Alaria esculenta* is believed to have declined in abundance along the coasts of both the UK and Ireland (Simkanin *et al.*, 2005; Mieszkowska *et al.*, 2006), while the warm-water kelp *Laminaria ochroleuca* is known to have proliferated along its northern range edge in southwest England (Teagle and Smale, 2018; Pessarrodona *et al.*, 2019). In general however, UK kelp forests are believed to be relatively stable (Wilding *et al.*, 2023), with little evidence of widespread losses or local extinctions, except for west Sussex and to a lesser extent the industrialised coast of County Durham (Hardy *et al.*, 1993; Sussex IFCA, 2020). That being said, ecosystems along the UK coastline are not exempt from the impacts of climate change and anthropogenic activities, meaning declines and/or losses of kelp in the near future are possible. As such, it is important to prioritise the conservation of UK kelp forests, alongside testing and refining restoration techniques so that they can be implemented in a swift manner if and when required. As such, we aimed to test the efficacy of 'green gravel' as a kelp restoration technique for wave-exposed intertidal shores in the UK.

Materials and methods

Aquarium culture

Fertile blades from the sugar kelp, *S. latissima* (Linnaeus) C.E. Lane, C. Mayes, Druehl and G.W. Saunders, was collected from intertidal sites along the northeast coast of England (Beadnell: 55.558440 N, −1.626257 W, and Seaton Sluice: 55.075683 N, −1.45875 W) in mid-November 2020. The blades were transported in cool box of seawater to the laboratory where zoospore release was induced following standard protocols for Laminarian kelps (Alsuwaiyan *et al.*, 2019; Fredriksen *et al.*, 2020). Specifically, blades were rinsed with sterilised seawater to remove epiphytes and sori were exorcised, wrapped in dry paper towel, and stored for 12 to 24 h at 4°C. Sori were then cut into ~8 cm^2 segments and placed in a beaker of sterilised seawater in at 4°C for ~3 h to stimulate zoospore release. Zoospore density was assessed by counting spores under a microscope using a haemocytometer and was estimated to be approximately 500,000 spores per ml.

In the aquarium, the spore solution (1 L), was sprayed onto two categories of rock, gravel (3–5 cm length, 50–100 g), and cobbles (6.4–11 cm length, 200–300 g). Inoculated rocks were gently submerged in seawater and left in the dark for 12 h to facilitate spore settlement. After 12 h, aquarium lights and flowing seawater were added. Constant lighting (TMC UK, Aquabar Ultra Daylight LED) was provided for the first 4.5 months, with a 12:12 h light: dark cycle used for the final two weeks to minimise stress at outplanting. Seawater was pumped from ~1.5 m depth in Cullercoats Bay, UK (55.034237 N, −1.429911 W). Initial seawater flow was 20 l per hour, and this was increased to 60 l per hour after three months and two impellor pumps were added to increase water movement and strengthen holdfast attachment.

Prior to outplanting, a small quantity of non-toxic coloured putty (Lyox Silicone Coral Putty) was added to each rock to ease identification of experimental rocks in the field. This material was selected over other marking techniques as it involved minimal disturbance to the established *S. latissima* and because the pilot experiments revealed that the putty was more resilient/robust than other adhesives (i.e., Reefix thermal polymer glue, Nyos reef cement, DD Aquascape Aquarium Epoxy) (Earp *et al.*, unpublished).

Field deployment

After five months of aquarium incubation, in March 2021, rocks were outplanted at two wave-exposed sites along the northeast coast of England (Figure 1). At each site, 21 individual gravel pieces and 10 cobbles were placed in three replicate patches on the low intertidal during spring low tides. Each patch was characterised by pools/gullies that remained covered by at least 5 cm of seawater at all states of the tide, and where *S. latissima* occurred naturally. Patches were monitored monthly to assess the retention of gravel and cobbles, as well as the density and maximum length of *S. latissima* sporophytes per rock. Monitoring surveys were undertaken by a snorkeller who spent ~10–15 min searching each patch plus a radius of ~3 m around each patch for deployed gravel and cobbles. Gravel and cobbles within the ~3 m radius were considered 'retained' and the density and length of *S. latissima* on these rocks was monitored. Due to time constraints, areas beyond the ~3 m radius were not monitored. A subset of gravel and cobbles were retained in the aquarium as controls and were monitored monthly for density and maximum length.

To determine whether the sites were suitable for the growth and persistence of *S. latissima*, within each patch, 10 naturally occurring *S. latissima* sporophytes were tagged and hole-punched 5 and 10 cm above the meristem (Parke, 1948). The total length and growth (i.e., distance between the meristem and the punched holes) of tagged individuals was monitored monthly. Due to inconsistencies in the monitoring protocol, hole-punch data from April to June was excluded. Dislodgement of tagged individuals was not monitored because it was not always clear if an

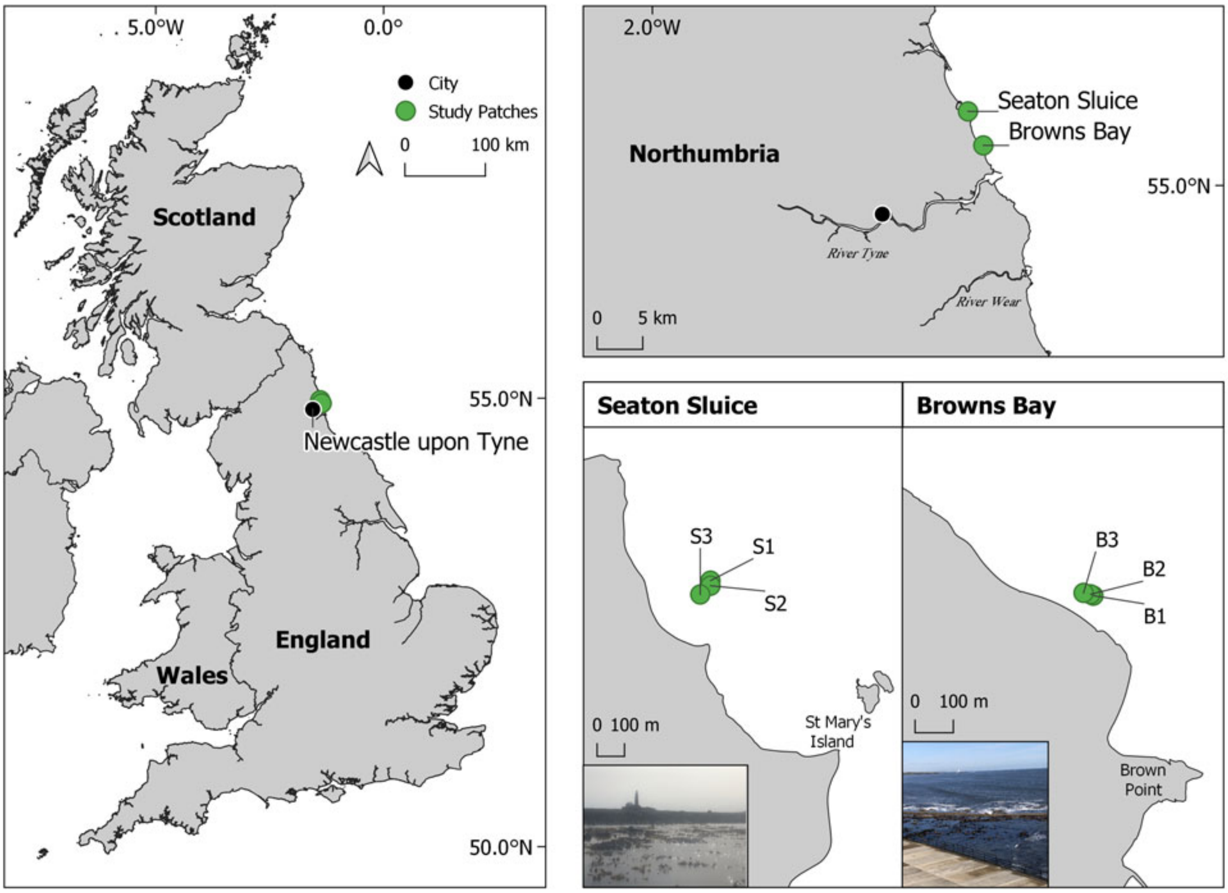

Figure 1. Location of the two wave-exposed restoration sites, Seaton Sluice and Browns Bay, on the northeast coast of England, and within each site, the three replicate patches (S1-3 and B1-3).

individual or just the tag had been dislodged, and new individuals were tagged at some time periods to ensure a robust sample size for length and growth measurements.

Environmental conditions

Site exposure was calculated using a wave fetch model (Burrows, 2020) and shores were considered sheltered if <2, moderately exposed if between 2–3.5, and exposed if >3.5 (Burrows, 2012). Wave and sea surface temperature (SST) data were obtained from the Newbiggin Ness Waverider Buoy (55.185167 N, −1.478167 W) and were made available by the Northeast Regional Coastal Monitoring Programme. Between July and October 2021, temperature, and light conditions were monitored intermittently at one patch per site using HOBO pendant loggers (Onset, USA). Between July and November 2021, two sediment traps with 0.5 cm baffles were deployed at each site (Appendix 1). Sediment samples were collected monthly, dried at 60°C, and sieved for particle size analysis.

Cost of restoration

To estimate the cost of the restoration experiment, we quantified the number of person hours and aquarium days dedicated to kelp cultivation, field installation and monitoring, and the mileage travelled to the restoration sites over an eight-month period. Labour costs were estimated using the UK Government national minimum wage for an individual >23 years of age (as of April 2023), aquarium costs were based on the aquarium hire fee charged by the Marine Biological Association of the UK (Harvey, personal communication), and travel costs were calculated using the UK Government mileage rate from 2011 onwards. The cost of aquarium hire to monitor aquarium controls over the course of the experiment was not included. The cost of consumables was based on approximate purchase costs of the items in 2020.

Data analysis

Prior to analysis, data from the patches at each site were pooled to enhance statistical power. To investigate differences in rock retention and the density of *S. latissima* on the rocks, generalised linear models (GLMs) with a Poisson distribution were used, and to test for differences in the maximum length of *S. latissima* on the rocks, a GLM with a Gaussian distribution was used. Each GLM included site (fixed; two/three levels), rock type (fixed; two levels), month (fixed; nine levels) and their interactions as factors. Variation in the number of levels for site in the models (i.e., two/three) is due to the inclusion of data from the two field restoration sites and the aquarium controls in the density and length analyses, whereas only data from the two restoration sites was included in the rock retention analysis.

The daily growth rate of naturally occurring *S. latissima* adults was calculated as follows:

$$\text{Growth} = [(H1e - H1s) + (H2e - H2s)]/t$$

Where $H1s$ and $H2s$ refer to the starting distance of holes 1 and 2 from the meristem (i.e., 5 and 10 cm respectively), $H1e$ and $H2e$ represent the distance of holes 1 and 2 from the meristem on subsequent monitoring periods, and t represents the number of days between the holes being punched and the subsequent monitoring

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period. The daily growth rate and length of naturally occurring *S. latissima* sporophytes was analysed using Gaussian GLMs with site (fixed; two levels), month (fixed; six/nine levels) and their interactions as factors. Variation in the number of levels for month in the models (i.e., six/nine) is because data from April, May and June was excluded from the growth rate analysis due to inconsistencies in the monitoring protocol, whereas data from the entire nine-month monitoring period was included in the length analysis.

Daily sediment deposition per site was calculated by dividing the weight of sediment collected in each trap by the number of days the trap was deployed in the field. Variation in daily sediment deposition was investigated using a Gaussian GLM with site (fixed; two levels) and month (fixed; four levels) and their interactions as factors.

Analyses were undertaken in the statistical software R [v.4.1.2] (R Core Team, 2021). GLMs were generated using the 'glm' function of the 'lme4' package (Bates *et al.*, 2015), and model fits were determined through visual examination of the residuals. Site/Treatment was not included as a random effect in the models because variance estimates may be imprecise when there are fewer than five levels of a random variable (Harrison *et al.*, 2018; Gomes, 2022). Type II sum of squares were calculated using the 'Anova' function of the 'car' package (Fox and Weisberg, 2019). Graphs were produced using the 'ggplot2' package (Wickham, 2016).

Results

Rock retention, and kelp abundance and length

After one month of deployment (i.e., in April), only 8% of gravel and 28% of cobbles remained at the sites and declines in retention continued until November 2021, eight months post deployment when 98% of rocks had been lost (Figure 2A). We recorded a significant Site by Rock Size interaction, with cobbles better retained compared to gravel, and retention, irrespective of rock size generally greater at Seaton Sluice where rocks often became wedged in small cervices, while at Browns Bay, many were buried/lost in the sediment (Table 1A; Figure 2A).

Seeded *S. latissima* showed evidence of self-thinning over the course of the experiment, transitioning from a dense cover of individuals at the time of outplanting to a smaller number of individuals towards the end of the experiment (Figure 2B). We found a significant Treatment by Monitoring period interaction, with declines in density most apparent during the first month of deployment and more severe on rocks in the field compared to those held in the aquarium (Table 1B; Figure 2B). A second noticeable decline in density occurred in the aquarium controls between June-August (Figure 2B). Across all monitoring periods and treatments, *S. latissima* density was greatest on cobbles (Figure 2B; Table 1B).

The maximum length of *S. latissima* sporophytes on the rocks was highly variable and we recorded a significant Treatment by Monitoring period interaction (Table 1C). At both field sites, the maximum length of individuals on each rock size sharply declined within one month of deployment (April), before gradually increasing towards June-July and declining again towards autumn (Figure 2C). Patterns in the maximum length of outplanted *S. latissima* were generally mirrored by the aquarium controls and naturally occurring *S. latissima* sporophytes, with the exception of the decline following deployment and the additional increase in maximum length at Browns Bay between August and October (Figures 2C & 3A). There was also a significant difference in maximum length across rock types, with individuals on cobbles consistently longer than those on gravel (Table 1C; Figure 2C).

Both Browns Bay and Seaton Sluice were suitable sites for the growth and persistence of mature, naturally occurring *S. latissima* sporophytes (Figure 3). Sporophyte length followed a similar pattern to that of individuals seeded on the rocks, with sporophytes increasing in length between March and July before declining from August onwards (Figures 2C & 3A). Maximum lengths were, however, significantly greater at Seaton Sluice compared to Browns Bay (Table 2A). Increases in length corresponded with seasonal growth rates which were generally greater in April-July (Figure 3B). Per month, there was site-level variability in growth (i.e., Site by Month interaction; Table 2B), which occurred over a more prolonged period at Browns Bay, while at Season Sluice there was a more noticeable peak and decline, although this may have been influenced by smaller sample sizes in July (Figure 3B).

Environmental conditions

Both Seaton Sluice and Browns Bay were considered exposed with wave fetch values of 4.27 and 4.33, respectively (Burrows, 2020). The field deployment in March coincided with an unseasonably warm spell where air temperatures were in the region of 18°C. This was followed by a cold snap in early April whereby a brisk northerly airstream brought a cold Arctic Maritime airmass across the UK (Kendon, 2021). During this period, air temperatures dropped to ~6°C, and on the 7th April 2021, significant wave heights peaked at 4.06 m and this coincided with high tide, resulting in significant wave action and seawall overtopping along the northeast coast of England (Kendon, 2021; National Network of Regional Coastal Monitoring Programmes, 2021). In the following months, environmental conditions were generally calmer and more stable, with monthly maximum wave heights generally ≤ 4 m, average significant wave heights often below 1 m, and seawater temperatures between 13–15°C (Figure 4A–C; Appendix 3A). Light intensity was broadly comparable across the sites, although it was often greater at Browns Bay (Appendix 3B). Sediment deposition varied significantly across the sites but not over time and was greatest at Browns Bay (Figure 4D; Appendix 3C). Generally, sediment deposited at Browns Bay was coarser, with over half of particles > 500 µm (i.e., coarse sand or larger), whereas at Seaton Sluice, sediment was finer and a greater proportion of particles were < 125 µm (i.e., fine sand or smaller) (Figure 4E).

Cost of restoration

Approximate costings of the restoration experiment outlined above (i.e., kelp cultivation, rearing in the aquarium on ~150 gravel and 70 cobbles, outplanting and field monitoring over eight months at two sites) are detailed in Table 3. In total we estimate that rearing, outplanting and monitoring of green gravel on a ~2 m^2 area on two wave-exposed intertidal sites (i.e., total of 4 m^2) would cost £4884.99, which is equates to approximately £1221.25 per m^2 (or £859.45 per m^2 excluding the eight-month monitoring).

Discussion

Despite the recognised importance of kelp forests and the reports of degradation and/or loss of these habitats from many regions, efforts to restore these ecosystems have generally lagged behind those of other marine systems (Krumhansl *et al.*, 2016; Filbee-Dexter *et al.*, 2022). With initiatives such as the UN Decade on Ecosystem Restoration and the Kelp Forest Challenge (https://kelpforestalliance.com/kelp-forest-challenge) now underway, there is a growing interest in kelp forest restoration. However, restoration

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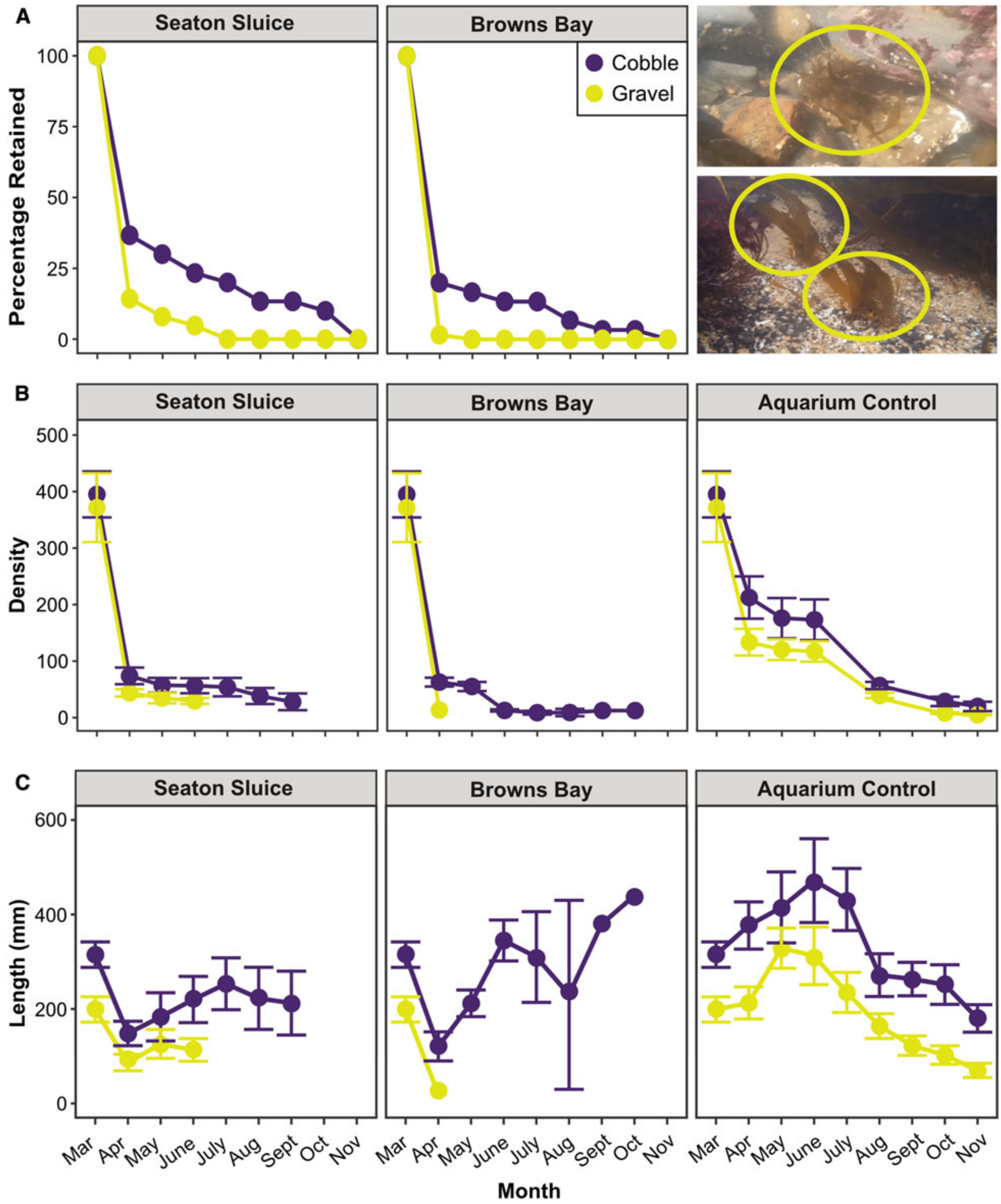

Figure 2. Mean rock retention (A), and S. latissima density (B) and maximum length (C) (± 1 standard error) on gravel and cobbles across the two wave exposed restoration sites and aquarium controls from deployment in March to the final monitoring point in November 2021. n values can be found in Appendix 4b. Inset images show green gravel (circled) wedged in a rocky crevice at Seaton Sluice (top) and buried in the sediment at Browns Bay (bottom) after three months of field deployment.

of kelp forests is challenged by the dynamic nature of temperate reefs, alongside the need to scale-up restoration interventions to match the scale of loss (Filbee-Dexter *et al*., 2022), as well as propagule limitation and a lack of hard substrate in some areas (Burek *et al*., 2018; O'Brien and Scheibling, 2018; Eger *et al*., 2022b). Green gravel has been advocated as a simple, cost-effective and scalable approach to kelp restoration, however its effectiveness has only been tested in limited environmental contexts (Fredriksen *et al*., 2020; Alsuwaiyan *et al*., 2022), although additional investigations are underway as part of the Green Gravel Action Group (see greengravel.org). Here we build on this growing body of literature by investigating the utility of green gravel as a kelp restoration technique along wave-exposed intertidal rocky shores.

Our results showed that *S. latissima* spores were able to adhere and successfully develop on rocks in the aquarium, and despite

Table 1. Generalised linear models to test for differences in retention (A), the density (B), and the maximum length of *S. latissima* (C) on gravel and cobbles across the sites/treatments during the monitoring period

	Retention			Density			Maximum length		
	df	*F*-value	*P*	*df*	*F*-value	*P*	*df*	*F*-value	*P*
Site/Treatment (S)	1	16.671	**<0.001**	2	13.179	**<0.001**	2	24.265	**<0.001**
Rock size (R)	1	26.409	**<0.001**	1	8.063	**0.005**	1	61.519	**<0.001**
Month (M)	8	6.033	**<0.001**	8	96.909	**<0.001**	8	8.643	**<0.001**
S × R	1	6.608	**0.016**	2	0.031	0.969	2	0.554	0.575
S × M	8	0.368	0.933	11	6.732	**<0.001**	13	3.227	**<0.001**
R × M	8	1.659	0.126	6	1.689	0.125	8	0.331	0.953
S × R × M	8	0.214	0.998	4	0.197	0.939	4	0.246	0.912
Residuals	64			189			213		

Significance was accepted at $P < 0.05$ and significant values are indicated in bold.

initial declines in density and length following the field deployment, they were able to persist and followed similar patterns of growth to naturally occurring *S. latissima* sporophytes. Initial declines in density and length were likely stress-related, associated with transportation and changes in environmental conditions (i.e., light levels), coupled with the more dynamic nature of the restoration sites (i.e., wave and storm-induced erosion, and herbivory). The sustained decline in density over time mirrors the findings of Fredriksen *et al.* (2020) and was likely a consequence of self-thinning, with larger, more robust individuals able to outcompete smaller individuals. It is important to note, however, that *S. latissima* is better adapted to wave-sheltered conditions, and although mature sporophytes were found at the wave-exposed field sites, intertidal populations are comparatively small and other species (e.g., *Laminaria digitata*) are more abundant. Natural populations of *S. latissima* declined substantially over

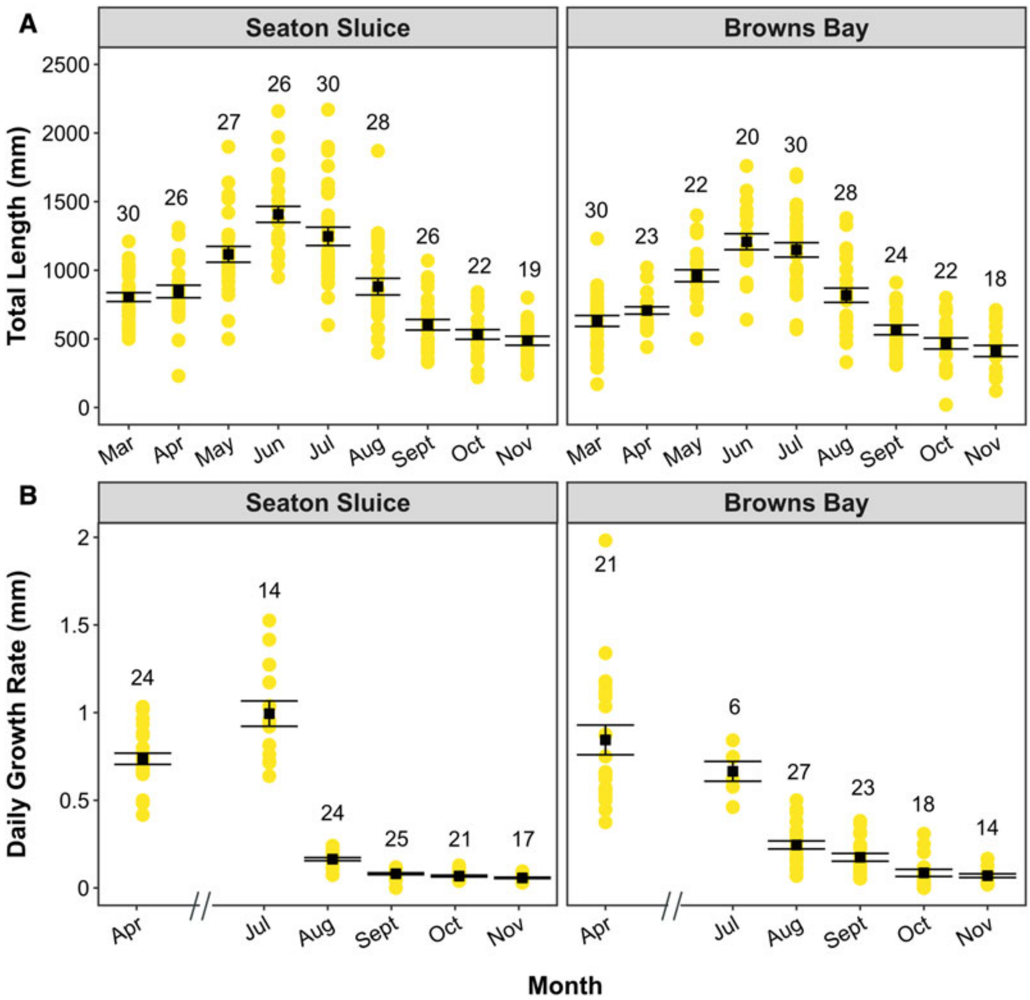

Figure 3. Total length (A) and daily growth rate (B) of naturally occurring S. latissima sporophytes per site between March and November 2021. Black squares and error bars represent the mean (±1 standard error). Yellow circles represent values per individual S. latissima and values represent the n per site per growth period.

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Table 2. Generalised linear models to test for differences in *S. latissima* sporophyte length (A.) and daily growth rates (B.) over time at the sites

	Length			Growth rate		
	df	*F*-value	*P*	*df*	*F*-value	*P*
Site (S)	1	24.1492	**< 0.001**	1	3.5556	0.060
Month (M)	8	74.0294	**< 0.001**	5	189.5395	**< 0.001**
S × M	8	0.6703	0.717	5	5.5948	**< 0.001**
Residuals	433			222		

Significance was accepted at $P < 0.05$ and significant values are indicated in bold.

winter months (Earp, personal observation), likely as a result of wave action and storm disturbance, before increasing in the spring/summer, with propagules potentially supplied from subtidal populations. As such, *S. latissima* may not be the most appropriate target restoration species for this area or wave-exposed shores more broadly, and trials involving other more exposure-tolerant species (e.g., *L. digitata*) are needed.

We found that seeded cobbles, which were greater in size and mass, were better retained within the patches than gravel, suggesting that at wave-exposed sites and/or on intertidal shores larger rock sizes could enhance restoration success. While using larger rocks may limit the reattachment of kelp directly to the substrate, it could improve substrate retention and ultimately increase the likelihood of kelp sporophytes reaching maturity so they may release propagules that seed the underlying substrate. In the case of *S. latissima*, rock retention would need to be for approximately 15–20 months for individuals to reach maturity (White and Marshall, 2007), which is almost double the retention period observed here. It is, however, important to note that the loss of gravel/cobbles from the restoration sites may not necessarily represent failed restoration as it is plausible that the rocks had simply moved beyond the searched area and remained viable elsewhere.

Given that this study is the first of its kind to deploy green gravel on wave-exposed intertidal shores, several lessons were learned that may inform future efforts. Firstly, it would be beneficial to characterise environmental conditions and/or processes (e.g., sedimentation) at restoration sites/patches prior to the deployment of green gravel to ensure they are suitable. It is also important to consider the timing of field deployments (e.g., after winter storms) to reduce the risk of initial losses. Furthermore, improving methods to mark/identify seeded rocks in the field would be valuable given that several of our markers were lost over time, meaning it was challenging to identify whether the rocks had been displaced, or whether just the *S. latissima* had been dislodged. Applying such marking/identification techniques is advised prior to seeding the rocks to minimise disturbance to juvenile sporophytes.

Future efforts should incorporate additional components that were beyond the scope of this research. For example, assessing the impact of seeding at different spore densities, as well as onto different rock types, textures, and shapes. Seeding density, which although has been found not to influence *S. latissima* and *Ecklonia radiata* growth on green gravel (Fredriksen *et al.*, 2020; Alsuwaiyan *et al.*, 2022), can result in reduced survival and growth in high density cultures due to competition for nutrients and space (Steen, 2003). While the influence of seeding rocks of different shapes (i.e., round vs thin and flat) has yet to be investigated, seeding different rock types and textures has been found to influence success, with greater detachment occurring on rocks with rougher textures (due to greater initial settlement), and severe tissue bleaching observed in individuals on limestone rocks (Alsuwaiyan *et al.*, 2022). It would also be interesting to understand the influence of genetic diversity on success as elevated genetic diversity was found to increase survival and density in seagrass restoration (Reynolds *et al.*, 2012).

In our study we found that rocks deployed at Seaton Sluice were better retained as they often became wedged in small crevices which is one of the key mechanisms by which this restoration technique is supposed to work. The viability of this restoration technique, however, is not dependent upon rocks remaining within a specific patch, but rather that they remain within the site. As such, future efforts would also benefit from expanding the search time and monitoring area beyond the 15 min and 3 m radius used here, alongside quantifying the distance moved by rocks within a site. Deploying tagged, non-seeded rocks as controls for rock movement and/or loss may also be beneficial in this case. In addition, work is underway on wave-exposed shores in Chile to determine whether it is feasible to attach green gravel to the underlying bedrock (Pérez-Matus, personal communication), and it could be beneficial to explore similar techniques on exposed shores around the UK, both as a methodological development, but also to monitor for kelp growth and holdfast overgrowth/attachment on to the underlying substrate.

Estimating the cost of restoration is not simple and only a limited number of studies have reported kelp restoration costs (but see Carney *et al.*, 2005, Campbell *et al.*, 2014, Fredriksen *et al.*, 2020, Eger *et al.*, 2022b). However, costings are inconsistently reported, making it challenging for practitioners to determine whether, what, where, how, and how much to restore (Bayraktarov *et al.*, 2015). Furthermore, costings often exclude the cost of pilot research, robust long-term monitoring, and non-consumable laboratory materials, and are often variable depending on the nature and distance of transport required. For example, initial research involving green gravel estimated that the technique costed ~£6 per m^2 (approximate conversion of US$ 6.75 to GBP, June 2023), however this value was exclusive of bench fees, vessel hire and long-term monitoring (Fredriksen *et al.*, 2020). Here we estimated the cost of restoring and monitoring wave-exposed intertidal shores (nearby to the aquarium facilities) using the green gravel technique over an eight-month period to be in the region of £1221 per m^2, with the cost inclusive of transport, salaries, and aquarium hire. This value, however, represents a minimum cost and is likely an underestimate, with the cost of salaries highly variable depending upon the organisation and qualifications of the individuals involved, and the cost of travel dependent on the location of the aquarium facilities and restoration sites. By comparison, similar research along the southwest coast of the UK estimated that the cost of restoring *S. latissima* across four subtidal sites (total area of 8 m^2) using the green gravel technique was approximately £1437.50 per m^2 including the cost of vessel hire and a commercial dive team, but not the cost of long term-monitoring (Wilding *et al.*, 2023). As such, restoration of intertidal zones may appear a more cost-effective option, but it is important to highlight that our work involved moving heavy

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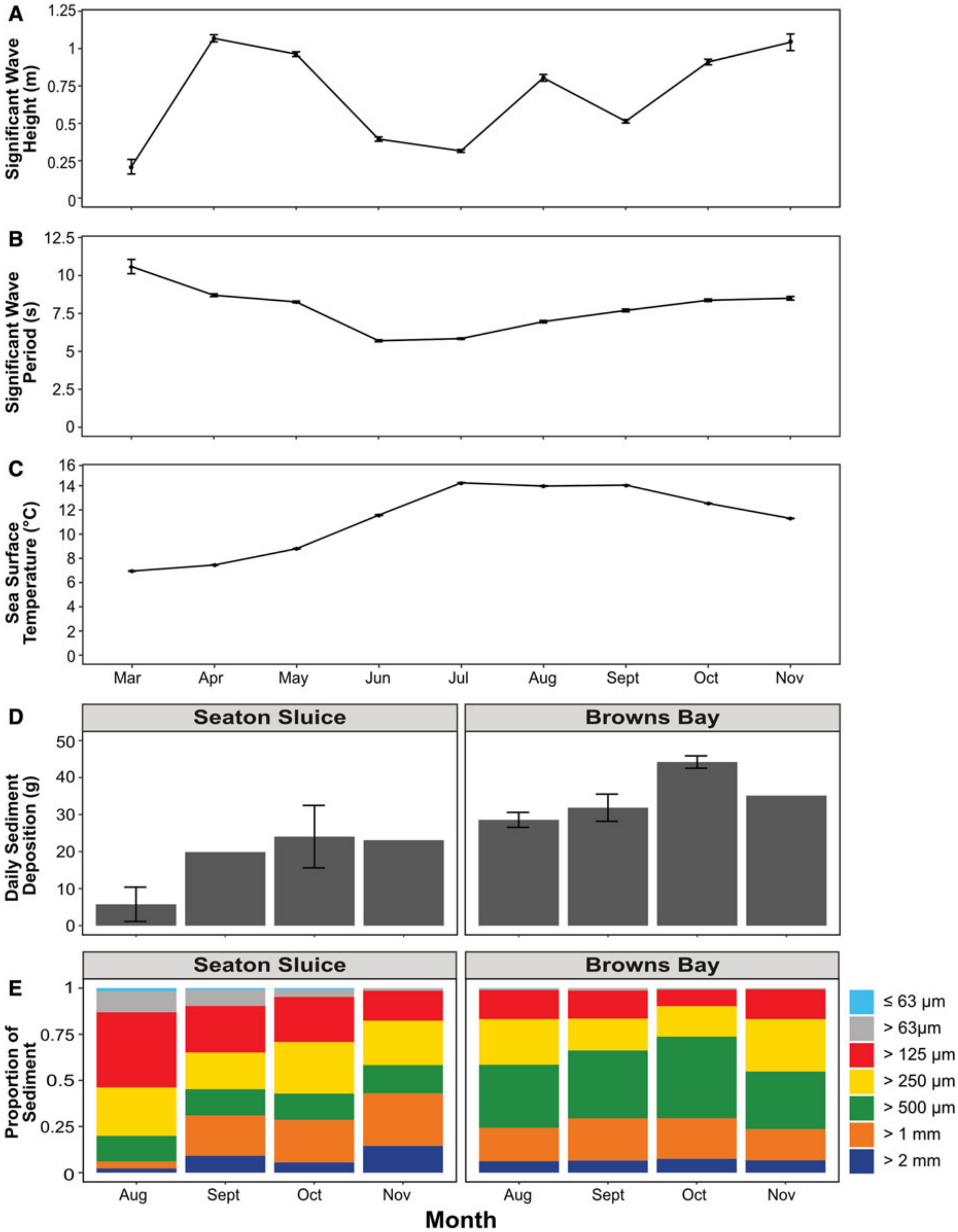

Figure 4. Average monthly environmental conditions (± 1 standard error) at the restoration sites including significant wave height and period (A, B) sea surface temperature (C) daily sediment deposition (D), and proportion of sediment size classes (E). Wave and sea surface temperature data was obtained from the Newbiggin Ness Waverider Buoy (Northeast Regional Coastal Monitoring Programme). Note: for graphs A-C, data for March and November are for the study period only (i.e., 30–31st March and 1st–5th November 2021). Sediment size classes in D-E are based on the Wentworth Scale (Wentworth, 1922) whereby particles > 2 mm are considered granules, > 1 mm are very coarse sand, > 500 μm are coarse sand, > 250 μm are medium sand, > 125 μm are fine sand, > 63 μm are very fine sand, and < 63 μm are silt/mud. Note: for graph D, the absence of error bars indicates where only one sediment trap was collected. Additional environmental information can be found in Appendix 3.

material across hazardous terrain and this may not be possible in all circumstances.

When compared to the cost of other kelp restoration techniques, Fredriksen et al. (2020) found that the costs associated with green gravel may be comparable or even lower, but that its potential for upscaling to match the scale of kelp degradation/loss may be greater. Scaling-up our green gravel costings suggest that kelp forests may be among one of the most expensive coastal marine systems to restore (£8594,500 per ha excluding monitoring), with costs exceeding averages for coral reefs and seagrass

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Table 3. Approximate costings of the restoration experiment

Activity	Details	Basis for cost	Cost
Preparation	Collection of spore material, spore extraction, collection and cleaning of substrates, aquarium set-up – three days for one person	• UK Government National Minimum wage of an individual >23 years as of April 2023 = £10.42. • Working week of 37.5 h.	£234.45
Facilities	Bench fees for aquarium and laboratory use for the rearing and cultivation – 138 days	– Marine Biological Association aquarium hire fee (incl. VAT) = £19.07 per tank per day	£2631.66
Field deployment	One days work for a four-person team	– See preparation basis	£312.68
Monitoring	Nine days work for a two-person team	– See preparation basis	£1406.70
Travel	Eleven return trips from the laboratory to both sites for spore material, deployment, and monitoring	– Round trip to both sites = 10 miles. – UK Government mileage rates from 2011 onwards = 45p per mile for first 10,000 miles in a tax year.	£49.50
Consumables	Aquarium pumps, aquarium lights, aerators, plasticware, microscope tiles	– Approximation based on 2020 cost of items outlined in methods section and plasticware for rearing and transporting	£250
		TOTAL	£4884.99

meadows (~£8159,215 and £602,346, costs scaled to 2023 GBP; Bayraktarov *et al.*, 2015) that are considered the most expensive systems to restore. Although, developments in the green gravel protocol may reduce costs, for examples, SeaForester (https://www.seaforester.org/) have developed 'mobile restoration containers' to rear kelp, which may eliminate expenses associated with hiring aquarium facilities and/or bench fees, as well as improving the applicability/feasibility of this technique, particularly in regards to remote areas (Vanbeek, personal communication). In addition, we suggest that research is undertaken to investigate the possibility of simplifying technical aspects of the protocol while maintaining the viability of the kelp spores/recruits, for example by using non-sterilised seawater in the cultivation process, rearing using seawater changes as opposed to running seawater, and rearing under natural light conditions, so that the technique may be employed by groups who may not have access to laboratory facilities such as artisanal fishermen.

In summary, while there is little evidence of kelp forest declines and/or losses around the UK, there is a need to test and refine restoration techniques in a variety of contexts and including a range of species so that in the future swift action can be taken to mitigate declines and conserve kelp forests. Green gravel is one technique within a suite of restoration tools (Earp *et al.*, 2022) that could be used to combat future declines and/or losses of both kelp and other forest-forming macroalgae, and it could be used to propagate resilient genotypes and 'future-proof' vulnerable kelp forests to future stressors (Wood *et al.*, 2019; Coleman *et al.*, 2020). While our initial testing of this technique on wave-exposed intertidal shores was unsuccessful, it provides important insights for developing/refining the technique further for a wider range of environmental conditions, as well as a baseline for comparison for future efforts.

Supplementary material. The supplementary material for this article can be found at https://doi.org/10.1017/S0025315424000225

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Authors' contributions. HSE, DS and PJM conceived the idea. HSE, HC and PJM conducted the fieldwork. HSE conducted the analyses and drafted the manuscript. All authors took part in discussions and contributed equally to subsequent drafts.

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