



Seagrass restoration in the Firth of Forth (2022-25): Final Report

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Executive Summary

This report summarises the progress and key lessons from restoration efforts targeting both *Zostera marina* and *Zostera noltii* in the Firth of Forth between 2022 and 2025. The project contributes to regional ambitions to restore intertidal and subtidal seagrass habitats and recover the biodiversity, coastal protection, and blue carbon benefits they provide. Restoration activities focused on the collection, processing, and planting of reproductive material and seeds across multiple sites, alongside trials to understand the specific requirements of both species. Habitat mapping at sites was also conducted. Reproductive shoots and transplants were harvested without impacting donor populations.

For *Zostera marina*, seed germination was achieved at several sites, demonstrating the feasibility of seed-based restoration in the region. Although long-term survival varied and failed at some sites, populations at one site have been maintained over successive winters and are slowly expanding in coverage. These trials greatly improved understanding of the conditions required for persistence, including sediment stability, hydrodynamic exposure, and water quality. This knowledge provides a basis for refining site selection and increasing the likelihood of sustained meadow development. The project also advanced restoration of *Zostera noltii*, improving understanding of transplanting approaches and the influence of tidal elevation and sediment dynamics on establishment success. While outcomes varied across sites and years, these efforts clarified the conditions under which *Z. noltii* is most likely to thrive and laid essential groundwork for future intertidal restoration. Although the project did not reach its stated ambition of planting 4 hectares of seagrass, data from areas under restoration (0.2 hectares of restored seagrass) at Drum Sands and Burntisland provide optimism that these aims will be met over time as lateral expansion continues.

Poor water quality remains a significant issue for seagrass in the Forth of Forth, with our data confirming these problems at local scales in existing seagrass and with consequences to associated biodiversity. Working in such a human-dominated and modified system has created challenges; however, across both species, the work has strengthened restoration capacity. It helped refine protocols for reproductive shoot collection, seed processing, storage, and deployment, and built a robust evidence base for scaling up restoration in Scotland. Even where survival was limited, each intervention generated valuable ecological and operational insights to inform adaptive management and future design. Identifying and safeguarding suitable areas for both species, while addressing stressors such as sediment instability and water quality, will be critical for long-term success. Continued multi-year monitoring remains essential for evaluating persistence and refining methods. Overall, the project marks a major step forward in restoring subtidal and intertidal seagrass habitats in the Firth of Forth. The successful germination of *Z. marina*, advances for *Z. noltii*, and improved understanding of species-specific requirements provide a strong foundation for future large-scale restoration and the long-term recovery and resilience of these ecosystems.

Acknowledgements

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Seagrass monitoring at Burntisland in October 2025

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1. Introduction: The Restoration Forth project

1.1. Restoring the ecological foundations of the Firth of Forth

In the pursuit of a more productive future for the Firth of Forth, the Restoration Forth project represents a landmark conservation partnership among charities, local community groups, and scientists, united by the ambitious goal of restoring vital marine habitats within the Firth of Forth. Project managed by WWF-UK, this initiative brings together a diverse range of expertise from organisations including Project Seagrass, the Royal Botanic Garden Edinburgh, The Ecology Centre, Heriot-Watt University, and the Marine Conservation Society. Central to the project's philosophy is a participatory, community-led approach that empowers coastal residents to actively restore and monitor their natural heritage. While the project's immediate objectives involve restoring 4 hectares of seagrass and 30,000 oysters, the broader ambition is to provide a nature-based solution to climate change while fostering a long-term behavioural shift toward coastal protection. With up to 1.3m people living in or near the shores of the Firth of Forth there is an urgent need to ensure long-term resilience to improve well-being.

The Firth of Forth was once a biological powerhouse, due to its staggering abundance of oysters, herring and many other fisheries (Smout 2011). Oysters have been exploited on the shores of the Forth for thousands of years (Noble et al. 2018) and have supported whole industries built on these abundant resources (Howard, McKay & Newton 1987; Thurstan et al. 2013). However, centuries of industrialisation and over-exploitation have severely degraded this ecosystem. The degradation began in earnest during the Industrial Revolution and has largely continued ever since. As Edinburgh and surrounding towns swelled, the estuary became a repository for untreated sewage and chemical runoff from coal mining, paper mills, and tanneries. This introduced heavy metals and reduced oxygen levels, suffocating benthic life. By the mid-20th century, the Forth's legendary oyster beds, once covering thousands of hectares, had been harvested to functional extinction (Thurstan et al. 2013). The loss of these "ecosystem engineers" reduced natural water filtration, increasing turbidity and habitat loss. Recent understanding of seascape scale ecology suggests the loss of oysters would have had significant implications for the functioning of the whole Firth of Forth (Preston et al. 2025). The collapse of the herring and spratt fisheries followed, driven by a combination of habitat destruction and intensive bottom trawling, which levelled the seabed's structural complexity (Howard, McKay & Newton 1987; Smout 2011). Today, the once-diverse multispecies fishery has been replaced by a simplified ecosystem, largely dominated by Nephrops (scampi) prawns (Howard, McKay & Newton 1987). While the water quality of the Forth of Forth has improved significantly in recent decades (Dobson et al. 2001), there still remain significant challenges with eutrophication, historic contaminants and ongoing sewage discharges (Metcalf et al. 2022; Cortis et al. 2024). The loss of structural biodiversity means the Firth remains a shadow of its historic productivity. The Firth of Forth

has been significantly altered by human activity. By the 1990s, more than 50% of the intertidal region had been claimed for agriculture, harbours, and industrial development. The construction of major infrastructure, such as the Forth bridges and regular dredging for vessel access, has caused considerable disturbance to coastal habitats. The Forth remains a human-modified 'Novel Ecosystem' (Hobbs, Higgs & Harris 2009) where invasive species and degraded habitats are the norm. A new analysis has also revealed that the sediments of the Forth have a significant anthropogenic origin (Wang et al. 2026).

1.2. Seagrass: The ecosystem engineers

Seagrasses are unique as the world's only marine flowering plants. In Scotland, the two primary species are *Zostera marina* (common eelgrass) and *Zostera noltii* (dwarf eelgrass). A third species, *Ruppia maritima*, is also present but tends to favour inner-estuary and lagoon environments, rather than marine shorelines. These plants are often described as "ecosystem engineers" because they fundamentally alter their physical environment to create rich, biodiverse habitats. By stabilising sediments with their extensive root networks and reducing hydrodynamic forces through their blades, they facilitate the deposition of organic material and create a sheltered environment for a wide variety of fish, invertebrates, and molluscs. Furthermore, seagrass meadows provide an indispensable winter food source for migratory birds of international importance, such as Wigeon and Brent Geese.

Evidence for long-term seagrass change in the Firth of Forth reveals a history of significant fragmentation and decline. Records of *Zostera spp.* date back to 1863, with the earliest specific record of *Z. marina* documented at Burntisland in 1823. While historical maximum distributions remain unknown, the UK has experienced a century-long decline of up to 92% in seagrass meadows (Green et al. 2021), a trend reflected in the Forth's current restricted intertidal distribution (Thomas 2020). The estuary's physical and biological landscape has been fundamentally altered, impacting seagrass persistence. Industrial land claims for agriculture and ports removed more than 50% of the intertidal region by the 1990s, while infrastructure projects such as the Forth Bridges and ongoing dredging introduced chronic habitat disturbance. We will never know the true historic extent of seagrass in the Firth of Forth. However, we do know that herbarium specimens of seagrass from the Firth of Forth from the late 1800's were much longer in shoot length than those recorded now (Freeman 2020). As light availability is a strong driver of increasing seagrass shoot length (Bertelli et al. 2021), it is reasonable to expect that the clarity of the waters of the Firth of Forth was previously much higher, allowing more prolific seagrass extent and growth.

1.3. The long-term path to recovery

The restoration of the Firth of Forth seascape is recognised as a "decadal journey" rather than a short-term fix. Ecosystem restoration requires a long-term perspective; before large areas can be created, success should be defined by the establishment of mature populations that, over longer periods, can lead to larger, self-propagating meadows. Restoration Fourth

argues that water quality in the estuary is gradually improving but requires much greater momentum, presenting a vital opportunity for recovery. The restoration work led by Project Seagrass expands existing fragmented areas and is intended to support degraded meadows and enhance their self-propagation potential. By strengthening the integrity of these depleted populations, the project aims start to build resilience against the ongoing threats of climate change and biodiversity loss.

A cornerstone of the project is the empowerment of local communities. By engaging diverse people through volunteering and school programs, the project aims to develop and consolidate the skills required for community-led seagrass restoration. This participatory approach ensures that the project is not just a scientific exercise but a social one, building connections between people and their coastal environment. The goal is for coastal communities to eventually plan and execute their own restoration efforts, guided by local community hubs. To maximise the chances of success, the restoration efforts were focused on sites where seagrass presence has been confirmed or historically reported, such as Tynninghame (Belhaven Bay), Drum Sands, and Burntisland (Pettycur). These sites enable testing of seeding and planting methods tailored to the dynamic intertidal environment of the Forth.

2. Seagrass restoration and mapping in the Firth of Forth - methods

2.1. Restoration methods

2.1.1. Site selection in the Firth

Restoration Forth has three sites (1) Burntisland in Fife, on the northern shore of the Firth and Forth. (2) Drum Sands located in West Lothian, on the southern shore of the Firth and Forth. (3) Tynninghame in East Lothian on the estuary of the River Tyne. Each site (as described in table 1 and shown) was selected using a habitat suitability model combined with expert input from Dr Alastair Lyndon (Associate Professor at Heriot-Watt University) and senior scientists from Project Seagrass. Site choice was further refined using historical data, local site surveys and data submitted through the citizen science Seagrass Spotter app.

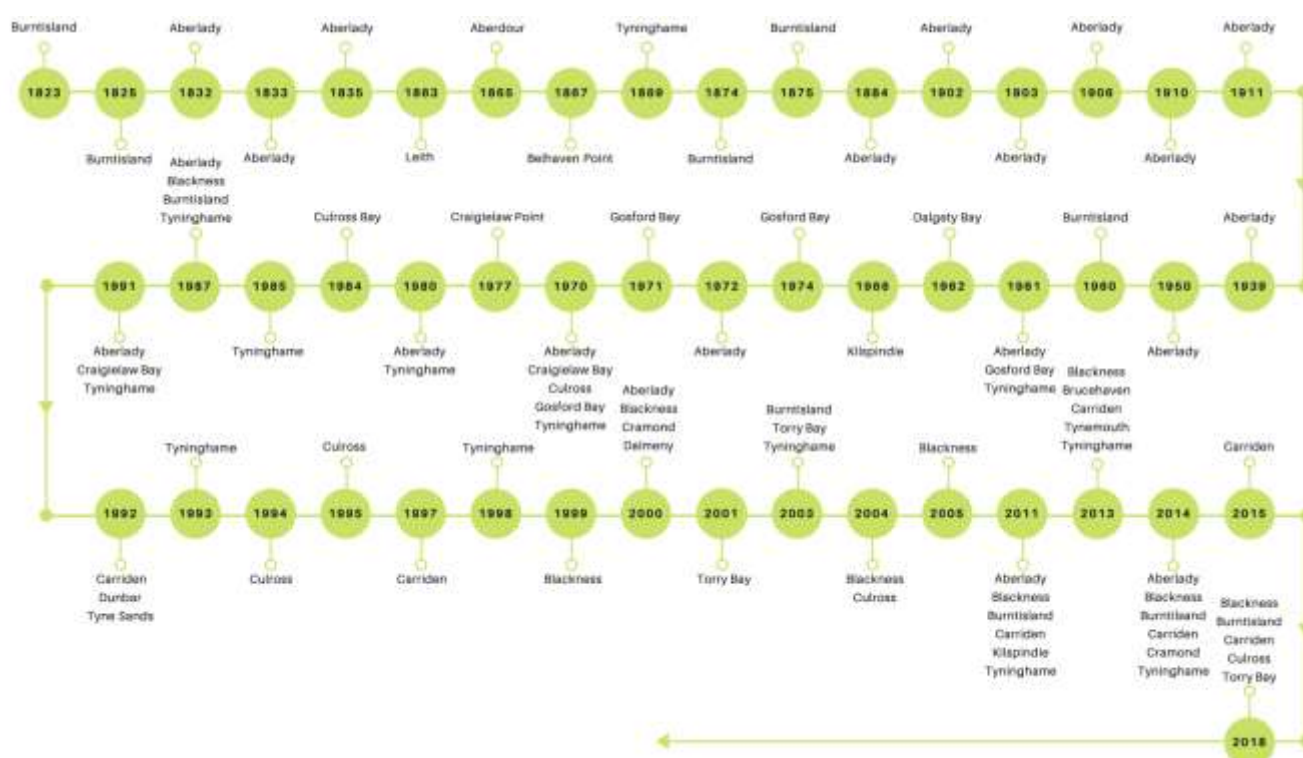


Figure 1. Timeline of *Zostera marina* records in the Firth of Forth. Between 1823 and 2018, there are 50 individual years which contain *Z. marina* records. Only records which had an exact year, rather than an estimated year, recorded are included (Thomas 2020).

Table 1. chosen sites with description, and *Zostera* knowledge in 2022.

Site name	Site Description
Burntisland	Located in sheltered intertidal sandflat extending from the railway line to Black Rock. Depth and wave exposure increase in the west towards Burntisland. <i>Zostera marina</i> and <i>Zostera noltii</i> are found in a patchy and mixed seagrass meadow high in the intertidal zone. Greater density and frequency of <i>Z. marina</i> are found towards the west of the site, where more tidal pools are present.
Drum Sands	Comprises a large sand and mudflat. In the southeast, adjacent to the mouth of the River Almond, a large <i>Z. noltii</i> meadow is located. <i>Z. noltii</i> grows on elevated sediment in the flat, while tidal pools are often bare sand or mud. The meadow extends, with gaps, until the outlet of Cockle Burn. <i>Z. marina</i> was not found at this site despite Dr Alastair Lyndon reported sightings of <i>Z. marina</i> in the past.
Tynninghame	Located within John Muir Country Park. It lies in the Tyne estuary receiving freshwater inflow. On the landward side of Sandy Hurst, there are mudflats in the lower and saltmarsh in the higher intertidal zone. The sediment is sandy to muddy within the estuary. The estuary is sheltered from wave exposure through a sandbar and rocks. Seagrass is present on the seaward side of Sandy Hirst.

It was identified that Burntisland and Tynninghame were best suited for *Zostera marina* restoration alone. Drum Sands was identified as suitable for *Z. marina* restoration, and later for *Z. noltii* restoration; thus, the following methodology is separated by species planted.

2.1.2. *Zostera marina*

2.1.2.1. *Zostera marina* seed collection

Seeds were collected in Orkney in 2022, 2023, and 2024. In 2024, additional seeds were collected from the Firth of Forth and the Moray Firth for use in a seed source experiment. Prior to collection, all donor seagrass meadows were surveyed to evaluate overall meadow condition and reproductive capacity. Measurements of shoot density, shoot morphology, and the density of reproductive shoots per square metre were undertaken in accordance with Nature Scot's licensing agreement. Seeds were harvested manually by selectively removing only reproductive shoots. Each shoot was carefully snapped at the connecting node to detach it from the rhizome, thereby avoiding damage to the root system and minimising disturbance to the donor bed.

Seed collection was conducted primarily by snorkelling, although in 2022 a combination of snorkelling and wading was used. Wading was subsequently discontinued because snorkelling resulted in less disturbance and lower risk of damage to the donor meadow.

Harvesting was carried out by Restoration Forth staff; however, in 2023 a small number of trained local volunteers assisted with collections on several days during the harvesting window. Snorkel sessions were scheduled around low tide to reduce water depth and improve access to the seabed while maintaining careful, low-impact harvesting practices.

Orcadian seeds were collected from Finstown and St Peter's Pool in August 2022 (approximately 50,000 seeds) and again in July–August 2023 (approximately 130,000 seeds). In July–August 2024, most seeds were collected from Mill Sands, a newly used donor site (approximately 61,620 seeds). Seeds from the Firth of Forth were collected in August 2024 at Burntisland (approximately 810 seeds), while those from the Moray Firth were harvested in September 2024 (approximately 22,120 seeds).

2.1.2.2. Seed processing

Initially, seeds were processed in a mobile Seagrass Processing Unit (SPU) based in Kinghorn (56°03'44.71"N, 3°10'47.32"W). The unit comprised six 120-L tanks used to process reproductive shoots and promote natural seed ejection. All reproductive shoots collected in 2022, and a subset from 2023, were processed in this facility. Shoots were placed in suspended plastic mesh containers within each tank, allowing ejected seeds to pass through the mesh and settle on the tank bottom for subsequent recovery and separation from plant material.

An electric generator supplied aeration to maintain oxygen levels, with an air stone positioned beneath each container to create upwelling and promote mixing. Tank water was fully replaced weekly, with partial changes between full exchanges to maintain water quality and protect seed viability. During water changes, shoots were gently stirred to dislodge and release retained seeds. Wastewater was screened through a fine-mesh sieve to capture invasive non-native species (INNS) and recover any uncaptured seeds. Collected seeds were stored separately in one of the tanks. Wastewater was then treated with Virkon Aquatic for 24 h prior to disposal. Processing occurred from August to early November, or until all material had been hand-processed with volunteer assistance.

In 2023, the Lobster Hatchery (North Berwick) joined the project to expand processing capacity. Four 500-L tanks were installed, and this facility became the primary processing site for the remainder of the project, while the SPU was retained for residual processing and community engagement. Processing methods were consistent between facilities, although the Lobster Hatchery used UV filtration for wastewater treatment. Seeds were overwintered in a seed-storage tower at ~4 °C and salinities >32 ppt to prevent premature germination. During winter 2024, seeds were stored at a bespoke facility in South Wales at the project Seagrass nursery in Pendine. This was chosen as the system was already operational and was able to guarantee a far more consistent environment for the seeds.

2.1.2.3. *Zostera marina* planting

Planting *Zostera marina* in 2023

In March 2023, the first Restoration Forth planting trial was conducted following the experimental design shown in Fig. 2. At each site (Burntisland, Drum Sands and Tynninghame) 6,500 seeds were planted using Dispenser Injection Seeding (DIS). A method of seagrass planting that uses modified caulking guns to inject seagrass seeds directly into the sediment. Media for the seeds within the caulking guns was created using local site-specific sediment, which was mixed with seawater to a consistency of 'Greek yoghurt'. The firing mechanism on each caulking gun was amended to limit the quantity of material ejected when fully squeezed. Caulking guns were calibrated to inject four seeds per injection. Holes were drilled on the lateral side of the tips, and tip ends were blocked. This was performed so that material was ejected out the side rather than out the end. This alteration was made to avoid material loss, as the seed mud mixture would not remain in the sediment when injected through the tip end. The 2023 plots were established to investigate the effects seed density and injection depth had on germination and seedling survival. The results from these trials were to guide future planting efforts by determining the treatment that yielded the best germination results.

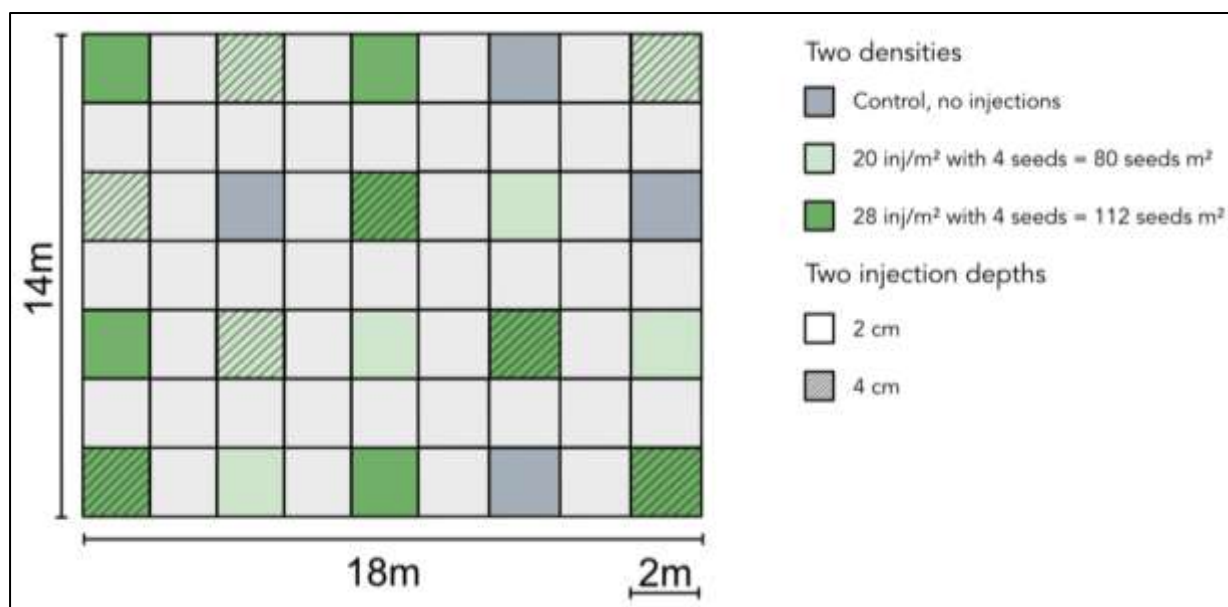


Figure 2. Experimental plot established in 2023 to test four seagrass injection treatments differing in planting density and depth. The treatments included: 20 injections m⁻² at 2 cm depth (n = 4), 20 injections m⁻² at 4 cm depth (n = 4), 28 injections m⁻² at 2 cm depth (n = 4), and 28 injections m⁻² at 4 cm depth (n = 4). Plot was repeated at each three Restoration Forth sites.

Planting Zostera marina in 2024

In March 2024, a revised planting approach was developed following sediment deposition observed across the 2023 experimental plots, which resulted in previously submerged areas becoming elevated above the sediment surface over the course of the year. To reduce the risk of plot burial or desiccation, circular plots (4 m² each) were established within existing tidal pools. Plots were organised into blocks of eight, with four planted and four left unplanted to monitor natural recovery alongside active restoration. Each planted plot received 1,000 *Zostera marina* seeds (250 seeds m²), inserted to a depth of 3 cm within a 1 m radius planting area. At Burntisland and Tynninghame, four blocks (32 plots total; 18 planted) were established, while Drums Sands contained eight blocks (64 plots total; 32 planted).

Planting Zostera marina in 2025

In March 2025, planting occurred at Drum Sands and Burntisland. Further restoration efforts at Tynninghame were ceased as no germination was recorded. Drum Sands contained three 25 m² plots and one 8 m² plot and BI contained two 25m²plots one experimental plot.

2.1.2.4. Restoration plots

Approximately 64,000 *Z. marina* seeds were planted across both sites using the DIS method, in which locally collected sediment is mixed with seeds and injected 3 cm into the substrate using a caulking gun. Each 25 m² plot was fully planted at a target density of 330 seeds m². Fig 3 illustrates the plots with their seed count in Drum Sands at Burntisland. Planting was undertaken by staff and volunteers, uneven seed distribution occurred due to caulking gun misfires, resulting in high seed concentrations in the initial rows and little to no seed deposition in later rows. Although guns were recalibrated and faulty equipment removed between planting sessions, uniformity did not improve, and cartridge counts were not recorded, preventing the exact seed densities from being known for each plot. An additional 8 m² plot at Drum Sands (Fig 3, B) and a 9 m² plot at Burntisland (Fig 3, C) were planted due to left-over seeds.

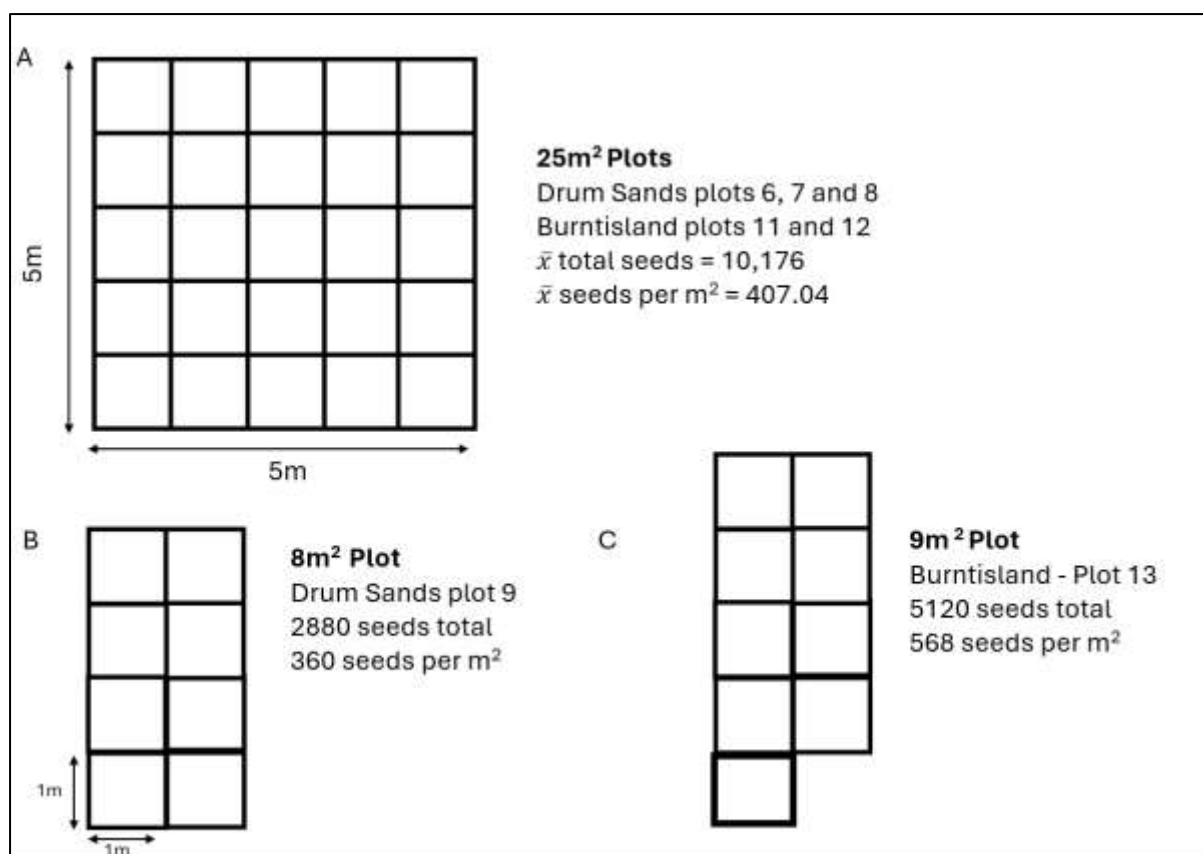


Figure 3. Plots planted at Drum Sands and Burntisland in March 2025. The included five 25 m² plots (three at Drum Sands and two at Burntisland), one 8 m² plot at Drum Sands, and one 9 m² plot at Burntisland.

2.1.2.5. Seed source experiment planting

In March 2025 at Burntisland, to investigate whether the source location of *Z. marina* seeds influences seedling survivability and restoration success. Four seed origins were tested: Tankerness and Finstown (both subtidal meadows in Orkney), the Moray Firth (intertidal meadow), and Burntisland (local intertidal meadow). This experiment was separated from the main restoration trial. The seeds were planted in the format seen in Fig 4. Due to a small number of seeds harvested in Burntisland, only the upper half of the box was planted in. Each seed source was planted 5 times within the plot.

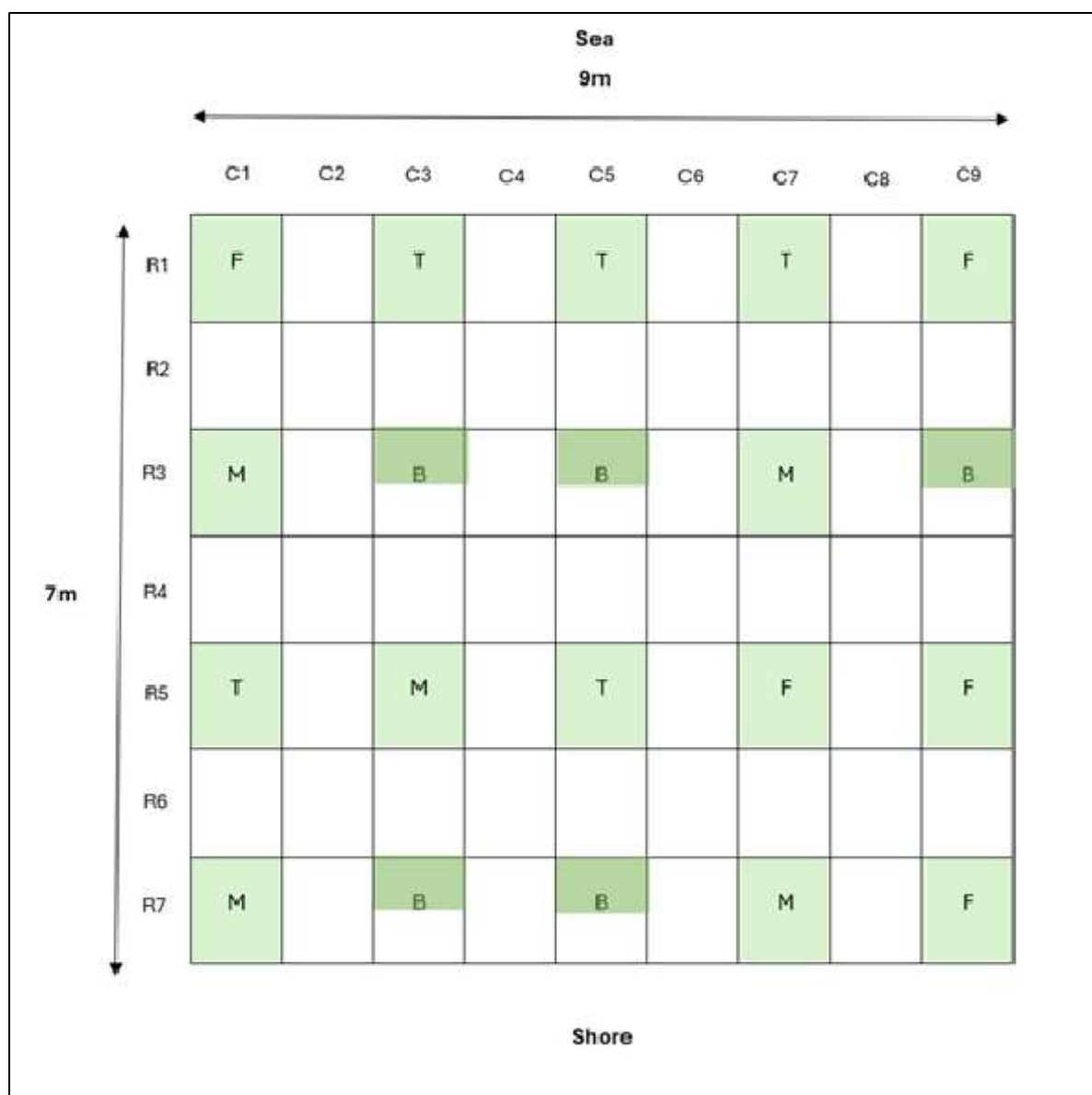


Figure 4. Planting layout of the Burntisland experimental plot, with each letter representing the original *Zostera marina* seed source used in the trial. T = Tankerness (n=5), F = Finstown (n=5) (both Orkney), B = Burntisland (Firth of Forth) (n=5), M = Moray (Moray Firth) (n=5). B plots are only half coloured due to limited seeds thus only the top half has been planted.

2.1.2.6. *Zostera marina* monitoring

The *Z. marina* planting plots established in 2023 were monitored in June, August, and November 2023 across all three sites. No seedlings survived into the subsequent year, and monitoring of these plots was therefore discontinued.

The *Z. marina* restoration plots established in 2024 at all three sites were monitored in June and August 2024. However, at Tynninghame all plants had died by August, and monitoring at this site was consequently discontinued. Monitoring at Drums Sands and Burntisland continued in November 2024 and February 2025. All three sites were revisited for a final monitoring survey in October 2025. During this final survey, all plots planted in 2024 were

consolidated into larger monitoring units. At Burntisland, four larger plots encompassed the original 32 smaller plots; however, only three of these larger plots were monitored because the location data for one plot had been lost. At Drums Sands, ten larger plots encompassed the original 64 smaller plots, while at Tynninghame five larger plots encompassed the original 32 smaller plots. This consolidation was undertaken to capture any potential seed movement beyond the original planting boundaries. Monitoring of the *Z. marina* restoration plots established in 2025 was conducted in October and November 2025.



Figure 5. Seed processing unit painted by Kinghorn Primary School students and by Donna Forrester. The SPU was used to store and process seeds for the Restoration Forth Project.

2.1.3. *Zostera noltii*

2.1.3.1. *Zostera noltii* seed collection and processing

Zostera noltii seed collection and planting were carried out at Drum Sands using nearby donor meadows. Seeds were harvested and planted at the same site within 24 hours. This site-specific approach was implemented to avoid the secondary introduction of invasive nonnative species (INNS) associated with moving seagrass material or sediment between restoration locations. In this context, “donor” refers to the nearby natural seagrass meadow

from which circular sediment cores containing live *Z. noltii* were removed to supply planting material for restoration plots.

2.1.3.2. *Zostera noltii* planting

Zostera noltii cores were collected and transplanted on the same day to minimise stress to the plants. The donor meadow at Drum Sands was divided into a grid, and only one core per square metre was taken from areas with seagrass cover greater than 70%. The Donor material was collected using 10 cm diameter cores inserted to a depth of 15 cm, ensuring the rhizomes, roots, surrounding sediment and leaf shoots were retained. The boundaries of the donor grid were marked with GPS and 50 cm bamboo canes. Each donor hole was also marked with a bamboo cane for future monitoring.

All cores were transported carefully and planted at the recipient site, located approximately 100 m from the donor area. A corresponding 15 cm deep, 10 cm diameter core was removed at each recipient position before inserting the transplant. At the recipient site, a grid of 1 × 1 m squares was set-up to test four treatments comparing clustered and non-clustered planting patterns. Fig 6 illustrates the four treatments and the number of cores needed for each treatment.

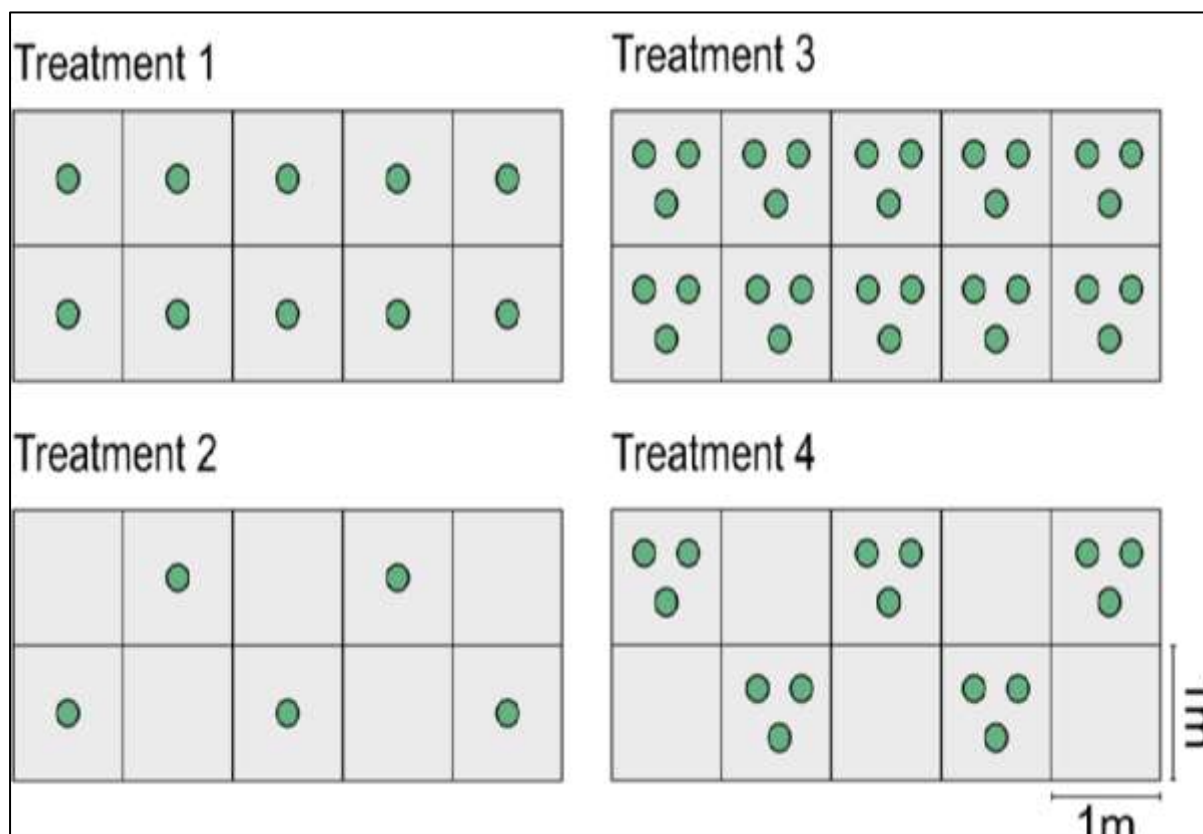


Figure 6. *Zostera noltii* experimental plot design planted in 2023. Treatment 1 = 10 cores, Treatment 2 = 5 cores, Treatment 3 = 30 cores, Treatment 4 = 15 cores.

The transplanting in Drum Sands took place in May 2023 during the low tide. Each treatment was replicated four times thus, 240 cores of *Z. noltii* were transplanted. In November 2023

during low tide the experiment was repeated with all four treatments replicated twice. The total amount of cores transplanted was 120. Thus, a total of 360 cores were transplanted in the in 2023.

In May 2024, *Z. noltii* was transplanted at Drum Sands using the two treatments identified in previous trials as the most successful: Treatment 1, consisting of 10 individually planted cores, and Treatment 4, consisting of 15 cores arranged in clumps of five groups of three (see Fig. 6). Both treatments were replicated 16 times. Thus, a total of 400 cores were transplanted in 2024.

2.1.3.4. *Zostera noltii* monitoring

Monitoring of *Z. noltii* transplants at Drum Sands was conducted over the three years of the study. In 2023, monitoring occurred in August and November, following both a summer and a winter planting undertaken that year. In 2024, monitoring was conducted in May and September. In 2025, monitoring was again carried out in May and October revisiting the 56 plots.

In October 2025, a donor hole recovery assessment was conducted to evaluate the extent to which sediment cores removed during 2023 and 2024 *Z. noltii* transplants had healed. A total of 46 donor holes were surveyed. GPS coordinates previously collected for each donor used for navigation. Surveyors located each donor hole using the DGPS, photographed the site, and recorded whether the hole had fully healed (i.e., infilled with sediment and/or recolonised by seagrass), partially healed or remained open. After assessment, any associated bamboo markers were removed.

2.1.4. Nutrient data collection and processing

To understand the background environmental conditions at seagrass meadows and seagrass restoration sites in the Firth of Forth, seagrass tissue nutrient analysis was conducted. This was alongside collection of epifaunal biodiversity data as part of a UK wide study, the results of which have implications for seagrass in the Firth of Forth (Hope et al. 2026). The tissue nutrient assessment was comparable with wider national datasets, we employed the standardised bioindicator approach established by (Jones and Unsworth 2016). Our field sampling strategy involved deploying 0.25 m² quadrats within the intertidal of each study site to capture representative variability. Within these quadrats, we harvested all seagrass material to measure shoot density and simultaneously sampled motile epifauna to assess biodiversity metrics, specifically species richness (S) and abundance (N). This followed methods developed within the *Zostera* Experimental Network (Reynolds et al. 2018).

In the laboratory, we prioritised rigorous cleaning protocols to ensure that analyses reflected the seagrass tissue itself rather than surface flora and fauna. Leaves were rinsed in fresh water, and epiphytes were carefully scraped from the leaf surfaces to prevent bio-fouling from skewing isotope signatures. Following this, the tissues were oven-dried at 60°C and ground to a homogeneous powder for analysis via Continuous Flow Isotope Ratio Mass

Spectrometry (CF-IRMS). This allowed us to measure Total Nitrogen (%N), Carbon (C), and stable isotope ratios to determine nutrient status and potential pollution sources (Jones et al. 2018). In addition to this tissue data collection, environmental data was logged on Hobo light and temperature loggers in the field. These loggers are notoriously difficult to keep logging over long periods of time, however some data was collected in 2023, 24 and partially in 2025 at Burntisland and Drum sands.

2.1.5. Statistical analysis and data visualisation

2.1.5.1. Analysis of 2024 planting data

Statistical analyses and data visualisations were performed on R version 4.5.1. To quantify and compare total shoot abundance for *Zostera marina* planted in 2024, monitoring data from all survey events were collated across sites. Data from Tynninghame, Burntisland, and Drum Sands were merged based on the final plot configuration surveyed in October 2025 that encompassed the circle plots planted. Spatial alignment of monitoring records was performed in QGIS (version 3.40.11 “Bratislava”; 2026). Field observations were joined to GIS-derived plot boundaries using plot identifiers and sampling event IDs. For each survey, total shoot count per plot was calculated. This was then illustrated in a line graph highlighting the total number of shoots at each site over time.

Burntisland was the only site where *Zostera marina* established from the 2024 plantings. Plot areas were assigned based on measurements from GIS (Plot 2: 366.53 m²; Plot 3: 433.67 m²; Plot 4: 382.79 m²). Plot 1 was excluded due to a corrupt GPS as exact location of the plot could not be found. This broad area contained seagrass, but it was not clear from the locations whether this was planted or natural. Total shoot counts were converted to shoot density (shoots m²) by dividing observed shoot numbers by the corresponding plot area. Shoot densities were aggregated by date; the resulting time series showed changes in mean shoot density over the monitoring period, with standard error bars indicating variability among the plots.

2.1.5.2. Analysis of 2025 seed source experiment

Variation among seed sources was first examined using boxplots of percent cover, shoot count, mean leaf length, and mean leaf width to summarise trait distributions. To assess multivariate patterns, these four traits were standardised and analysed using Principal Component Analysis (PCA) implemented with the `rda()` function in the `vegan` package in R. The PCA was based on the correlation matrix, and eigenvalues were used to quantify variance explained by each component. To test for differences in multivariate composition among groups, a permutational multivariate analysis of variance (PERMANOVA) was conducted using the `adonis2()` function with Euclidean distance as the variation measure. Site scores were plotted on the first two principal axes with trait loadings shown as biplot vectors to illustrate relationships among traits and seed-source groupings.

2.1.5.3. Analysis of *Zostera noltii* data

Monitoring data from five survey periods (August 2023, November 2023, May 2024, September 2024, and May 2025) were imported from Excel and merged in R using a unique plot identifier that combined treatment size (5, 10, 15, or 30 cores), replicate, transplantation season and year, and quadrat number.

Treatments were analysed as categorical factors and plots were grouped into transplantation sessions (Summer 2023, Winter 2023, Summer 2024). Initial planted area (znca0) was standardised according to treatment size (78.54 cm² for 5- and 10-core plots; 235.62 cm²).

Seagrass cover at each survey was quantified using a 5 × 5 grid quadrat, with percent cover (zncp) calculated as $(znsq / 25) \times 100$; where only percentage values were recorded, these were used directly. Patch area at each survey (zncax) was calculated as zncp multiplied by quadrat area and change in area (delta) was defined as zncax – znca0.

Magnitude of change (mag) was expressed as the fold-change in area relative to initial planting (zncax / znca0). For each transplanting session, treatment, and survey date, mean patch area and mean magnitude of change were summarised with standard errors (SE = SD/√n). Individual plot trajectories were visualised using line graphs, and treatment-level summaries were presented as mean ± SE bar plots.

The total potential site area was calculated by multiplying the number of treatment plots (n = 62) by the area of each plot (10 m²), giving 620 m² (0.062 ha). Actual seagrass cover was calculated for each plot by multiplying percentage cover by plot area (10 m²), then totalling across all plots to give 380 m² (0.038 ha).

2.1.5.4. Analysis of donor hole recovery

Donor hole recovery was assessed during monitoring in October 2025. A subsample of donor holes was surveyed and classified into three recovery categories: fully healed (Y), partially healed (P), or not healed (N). This was visualised in a boxplot.

2.2.6. Seagrass drone mapping

We analysed the spatial distribution of seagrass (*Zostera noltii*, *Zostera marina*) across Drum Sands, Burntisland and Tynninghame at the close-out of the Restoration Forth project. Each site was surveyed with drone imagery at the beginning and end of the project period, in 2023 and 2025. Here, we use a combination of drone imagery and field-collected ground-truth data to estimate the extent of seagrass present.

2.2.6.1. Mapping methods

To map the extent of seagrass at the Drum Sands, Burntisland, and Tynninghame sites (Figure 17), we combined drone surveys with field-collected ground-truth data. We then used machine-learning models to classify the drone pixels by the presence or absence of seagrass, and validated the classification using the collected ground-truth data.



Figure 7. Location of the Restoration Forth sites at Drum Sands, Burntisland, and Tynninghame in relation to Edinburgh and the Firth of Forth.

2.2.6.2. Collection of ground-truth data

We visited each Restoration Forth site (Drum Sands, Burntisland, Tynninghame) to collect ecological data within seagrass restoration plots and transects during October and November of 2025 (see section 2.1.2.6). We used an EMLID Reach RS3 (Emlid Tech Kft, Budapest, Hungary) DGPS system to collect accurate (< 2 cm horizontal error) geospatial data at each site. To support the classification of drone imagery, we used the DGPS system to collect ad-hoc ground-truth points at Drum Sands ($n = 115$). For each ground-truth point, we noted the absence or presence of seagrass, and visually assessed whether seagrass present was sparse, moderate, or dense in coverage. At Tynninghame, we found no seagrass and did not collect any ground-truth points.

Additionally, we used the DGPS to collect points that represented the corners bounding each restoration plot, the start and end point of each restoration transect, and the location of any *Zostera noltii* transplant core donors – where identifiable. These data enabled us to assess the predicted presence of seagrass in drone imagery against the areas where restoration work had taken place. Due to number and spatial complexity of plots across the sites, the data showing the location of plots are presented as a series of maps in the supplementary materials of this report (Figures S1-S4).

2.2.6.3. Elevation profiles over Zostera noltii restoration transects at Drum Sands

When collecting ecological data on *Zostera noltii* transplants at the western end of the Drum Sands site, we observed sediment accretion along each transect where *Z. noltii* was present. To evidence the accretion of sediment, we collected a series of DGPS points along the transect, at approximately 5 cm intervals, with a further three points beyond each end of the

transect. We then averaged the points outside of the restoration transect to get a base elevation and compared this with the elevation profile of the points within the transect.

2.2.6.4. Collection of drone imagery

Drone surveys were conducted with a DJI Mavic 3 Multispectral (DJI, Shenzhen, China). Surveys were carried out at 50 m above-ground-level, using a lawn-mower flight path, with a minimum of 75% front- and side-overlap between images. At each site, we designed the survey area-of-operations to cover all GPS points collected over the project's history, then expanded the survey area by a margin of at least 50 m to ensure no peripheral distortion during orthomosaic processing. At Burntisland, we were not able to achieve a margin to the north of the site, due to the presence of a live rail line where overflight was prohibited by legal and safety constraints. We targeted survey days with low wind speeds (< 12 m/s), where cloud cover was either full (100% cover) or absent (0% cover), following the recommendations of Nahirnick *et al.* (2019).

The RTK GPS unit on the drone was connected to an EMLID Reach RS3 base station for the duration of each drone survey, to enable accurate geolocation of the collected imagery. Additionally, ground control points (GCPs) were placed within the area-of-operations and surveyed with DGPS, to facilitate validation and subsequent correction of the drone orthomosaic's geolocation in post-processing.

2.2.6.5. Processing of drone imagery

To create a pair of orthomosaics for each site, we processed RGB and multispectral imagery through Pix4D Fields (Prilly, Switzerland). For each orthomosaic, we selected the 'Accurate' processing pipeline, the sky condition representing that present at the time of the survey (clear, overcast), and enabled radiometric correction based on the drone's incident-light-sensor. We set the GSD of each orthomosaic to 5 cm. As Pix4D fields does not take account of GCPs during orthomosaic processing, we could not enter the GCP location data to the processing pipeline.

To check the quality of the orthomosaics produced, we visually inspected and compared them with satellite and field-collected DGPS data in QGIS (version 3.4.11; Grüt, Switzerland). We observed minimal artefacts within the orthomosaics, though some distortion was present within a 50 m buffer of the orthomosaic edge. When assessing correspondence between each GCPs visual location and its DGPS location, we noticed discrepancies in all orthomosaics (2–3 m error). At Burntisland and Tynninghame, we were able to correct this discrepancy by using the georeferencing functionality within QGIS to reproject the orthomosaic. However, at Drum Sands the georeferencing process further distorted the imagery. Drum Sands was the site with the largest area, where GCPs were most constrained to the sandy shoreline, in proximity to tall trees. Ultimately, we decided that the correspondence between DGPS points and seagrass patches in central area of the orthomosaic was close enough (< 0.5 m) to enable use of the imagery in subsequent analyses.

To create a single orthomosaic that would support the analysis of seagrass extent, we combined the RGB and multispectral orthomosaics for each site using the terra package in R (version 1.7.55). We then calculated the normalised difference vegetation index (NDVI) for each site by calculating the standardised difference between the red ($650\text{ nm} \pm 16\text{ nm}$) and near-infrared ($860\text{ nm} \pm 26\text{ nm}$) bands, and appended this as a new band within the orthomosaic. Finally, we clipped the resulting orthomosaic at each site to an area-of-interest for all subsequent processing steps.

2.2.6.6. Comparison with drone imagery from project initiation

Drone imagery was collected by Project Seagrass at each of the Restoration Forth sites in 2023, before restoration efforts began. Surveys were conducted using a Wingtra WingtraOne Gen1 fixed-wing UAV, carrying a Sony RX1 RGB camera. The surveys were conducted at more than 60 m above ground level, with a lawn-mower flight pattern. Imagery was processed through Pix4D fields to an RGB orthomosaic. Due to: (a) the lack of a multispectral sensor and (b) differences in survey area and flight plan between 2023 and 2025 drone imagery (see section 2.3), we were not able to conduct geospatial analyses of change based on the imagery. Instead, we present visual comparisons between the imagery at each site in 2023 and 2025.

2.2.6.7. Creation of model training data

To create data for training a model to predict the presence and absence of seagrass, we first visually assessed the false-colour composite RGB and NDVI orthomosaic for each site. We identified four classes present within the orthomosaics at Drum Sands and Burntisland: seagrass, algae, rock, and bare sediment. At Tynninghame, we could not identify any areas of seagrass from the drone imagery, despite the aid of seagrass polygons created by Restoration Forth field staff during October monitoring with volunteers. As such, we could not proceed with creating training data for Tynninghame.

For Drum Sands and Burntisland, we manually created more than 1000 training points within each class, placing the training points only in areas where we could confidently identify the class of the pixel based on our field-experience and the pixel's appearance in RGB and NDVI imagery. During this process, we noted that the raised areas of sediment accretion we observed at the *Zostera noltii* transplants (see section 2.1.3.2) appeared light-grey in the RGB imagery, and were visually consistent with high-NDVI areas across Drum Sands.

2.2.6.8. Training of model

To train a model for the prediction of seagrass presence and absence at Drum Sands and Burntisland, we first grouped the training data for each site by class. For each class, we then sampled 1000 random points and extracted the spectral data and NDVI for each point from the corresponding orthomosaic. Next, we partitioned the data so that 20% of the points within each class were reserved for model validation. With the remaining points (80%; $n = 800$ per class), we trained a Random Forest classification model. We trained separate models

for the prediction of seagrass at Drum Sands and Burntisland, as the sediment and species present (*Z. marina*, *Z. noltii*) differed between the sites.

To select the most appropriate model for each site, we iterated through all possible combinations of model parameters (m-try, ntrees) and selected the model with the lowest out-of-bag error rate (Figure S5). We then used the selected model at each site to predict the classification of all drone pixels across that site and tested the classification's accuracy using the reserved validation data.

We attempted applying the selected Drum Sands and Burntisland models to the Tynninghame drone orthomosaic, however the models were overfitted to their respective sites and performed poorly for Tynninghame – incorrectly predicting extensive seagrass. Given that we observed minimal seagrass at Tynninghame in the field in 2025, and that models perform poorly, we do not present any classification for Tynninghame.

2.2.6.9. Predicting seagrass extent and calculating areal coverage

We applied the Random Forest model specific to Drum Sands and Burntisland to predict drone pixels containing seagrass at that site. We then used the withheld validation points for the site (see section 2.7) to calculate an accuracy rate for the predictions. Finally, we calculated the area of seagrass present at Drum Sands and Burntisland by summing the area of all 5 cm drone pixels where seagrass was predicted to be present.



Figure 8. Burntisland volunteers in March 2024 planting *Zostera marina* - Copyright Maverick Photo Agency



Figure 9. Drum Sands volunteers in March 2024 planting *Zostera marina* - Copyright Maverick Photo Agency.

3. Restoration and mapping in the Firth of Forth – results

3.1. Restoration results

Over the course of the project, accounting for all three sites, seagrass was planted over a total area of 1.65 hectares. Specifically, *Zostera marina* was planted over 1.041 hectares across Drum Sands, Burntisland and Tynninghame. *Zostera noltii* was transplanted from healthy donor meadows to restoration plots covering a total of 0.001 hectares at Drum Sands. At the conclusion of the project, we observed a total of 0.2 hectares of low-density planted seagrass (*Zostera noltii* restoration at Drum Sands, *Zostera marina* at Burntisland) that has survived over wintering and summer heatwaves. These are highly conservative estimates due to the challenges of quantifying seagrass as it spreads over time.

Figure 10 illustrates the typical areas of planted *Zostera marina* at Burntisland and *Zostera noltii* at Drum Sands observed during the restoration monitoring. These images show that areas previously devoid of seagrass now contain emerging, low-density, fragmented patches, demonstrating early establishment at both sites.

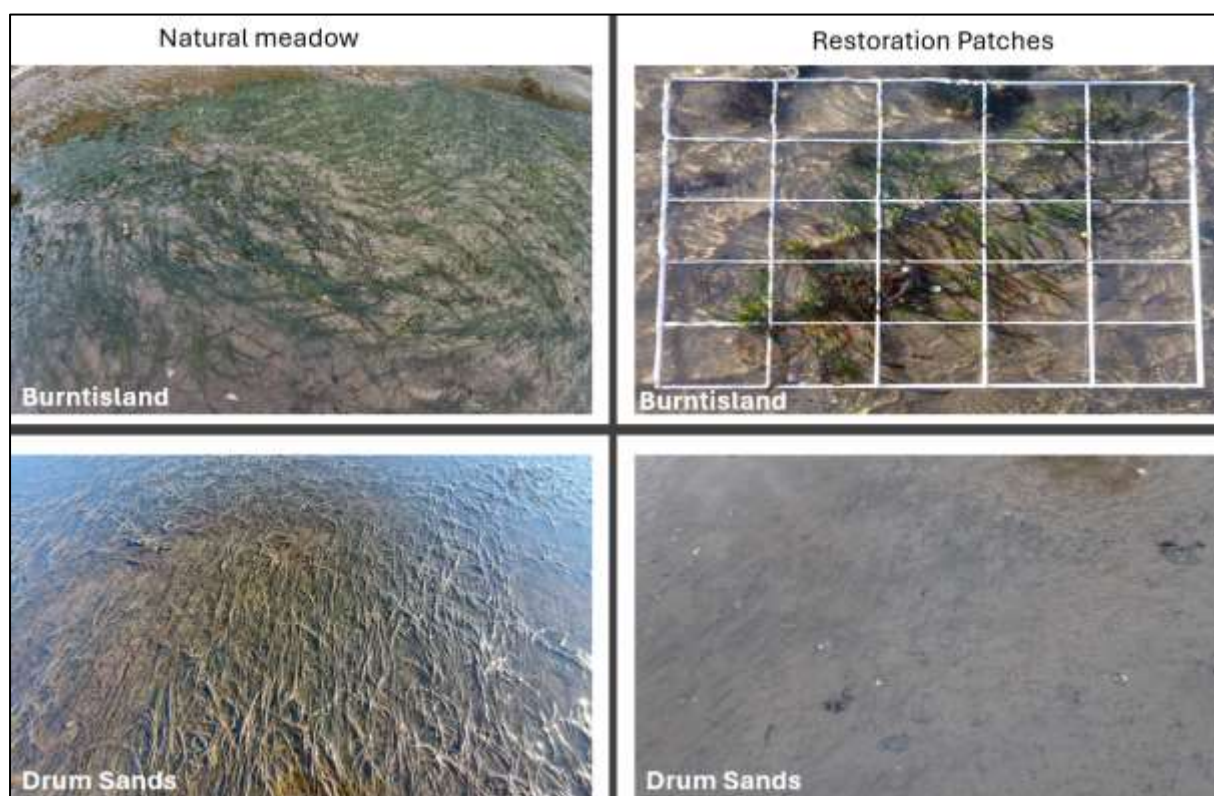


Figure 10. Natural meadow from Burntisland (*Zostera Marina*) and Drum Sands (*Zostera noltii*) compared with the restoration areas that were once void of seagrass.

Herein, we report on the outcome of each year and the method of seagrass restoration across the three Restoration Forth sites.

3.1.1. *Zostera marina*

3.1.1.1. Results for 2023

No seedlings were recovered at Burntisland in 2023, where sediment movement had buried the plots, and no seedlings were detected at Tynninghame. At Drum Sands, 19 shoots were recorded in one high-density, 4 cm injection-depth treatment. Overall, seedling establishment was unsuccessful, and no viable populations were formed in the Firth of Forth during the 2023 trial. The reasoning for the loss of the initial seedlings at Drum is thought to be poor light availability from sediments and nutrients after heavy rainfall (see section 2.23)

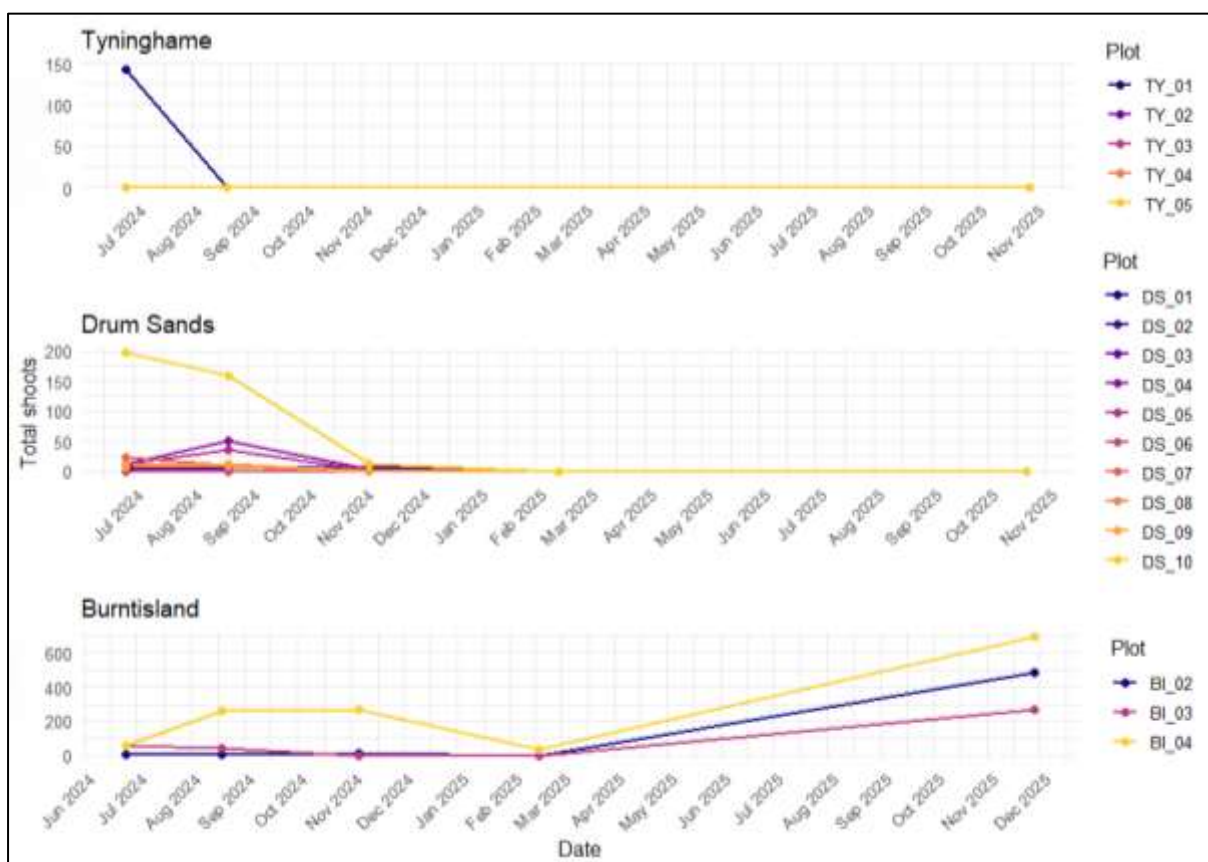


Figure 11. The total number of shoots from each site at each monitoring event from seagrass planted in 2024. Note: BI_01 plot location was lost due to a corrupt GPS, thus could not be monitored.

3.1.1.2. Results for 2024

Zostera marina restoration plots planted in March 2024 showed clear variation in shoot establishment and survival across sites and plots. Fig 11 illustrates the number of shoots at each restoration site. At Tynninghame, only plot one had shoots growing in July 2024. Each subsequent monitoring found no shoots at the site. At Drum Sands, plots 1 and 6 showed no shoot growth throughout monitoring. Plot 10 showed the strongest early establishment (196 shoots in June 2024), but all shoots were lost by winter. All other plots (2, 3, 4, 5, 7, 8,

and 9) displayed initial growth in 2024 but declined rapidly and failed to survive the winter, with zero shoots recorded in February 2025. At Burntisland, Plot 2 showed low shoot numbers in 2024 with none surviving the winter, but by November 2025 shoot count increased to 481. Plot 3 followed a similar pattern, with initial summer growth in 2024, winter loss, and recovery to 270 shoots in November 2025. Plot 4 showed the strongest survival, with steady summer growth in 2024, partial winter survival, and the highest recorded count had 691 shoots in November 2025.

Burntisland had the highest survival rate of the 2024 planting, with low density through 2024 and early 2025 before sharply increasing in November 2025 (Fig 12). The final mean shoot density at the site was low, with an average of 1.25 shoots m^2 in a total area of 0.17ha.

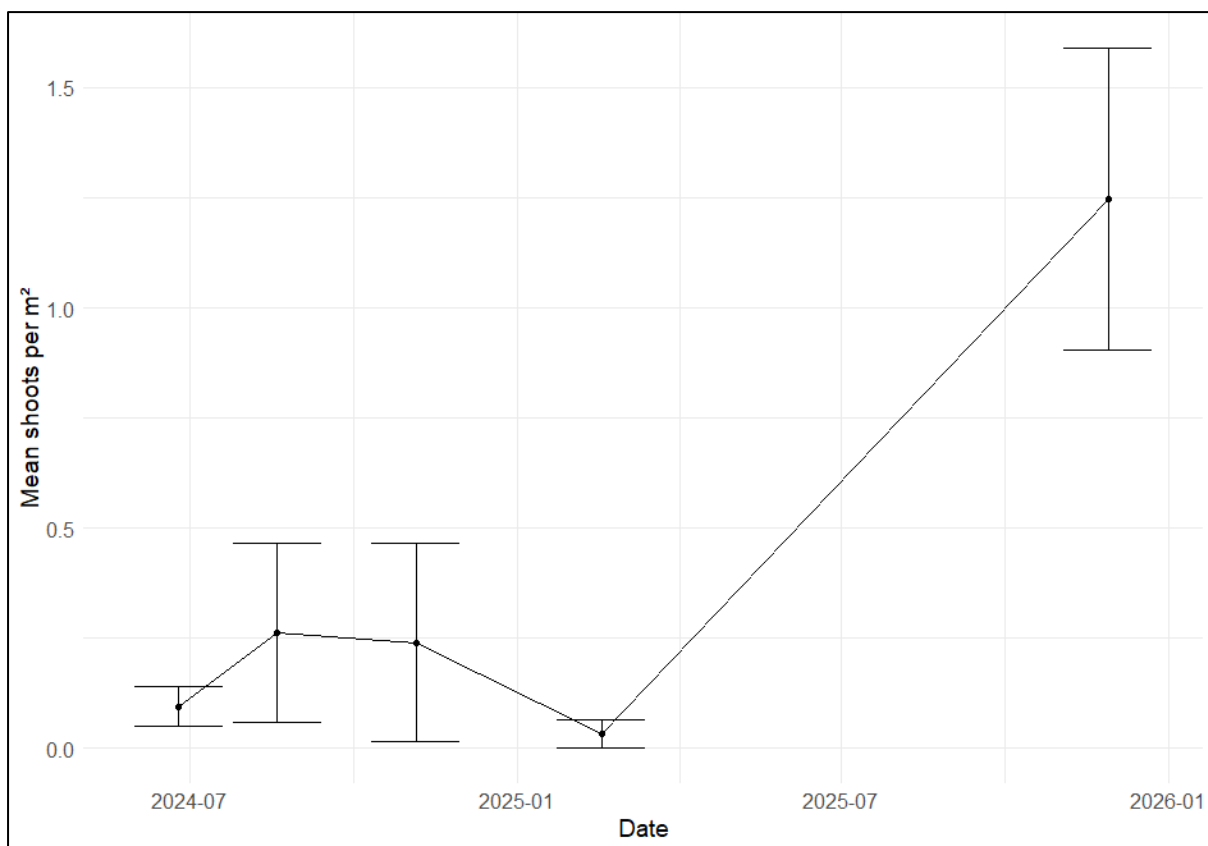


Figure 12. The average shoot density (*Zostera marina*) per m^2 in Burntisland restoration plots planted in 2024 throughout the monitoring period ($n=3$).

3.1.1.4. Results for seed source experiment

Plant performance varied among the four seed sources, with differences in percent cover, shoot production, and leaf morphology (Fig 13). Burntisland seeds produced the highest values overall, including the greatest percent cover and high shoot counts. Seeds from the Moray Firth had a low per cent cover but a high shoot count, which represents the morphology of shorter and narrower leaves. The Orcadian seeds from Tankerness and

Finstown had similar leaf morphology and number of shoots. However, Tankerness had a slightly higher per cent cover than Finstoun.

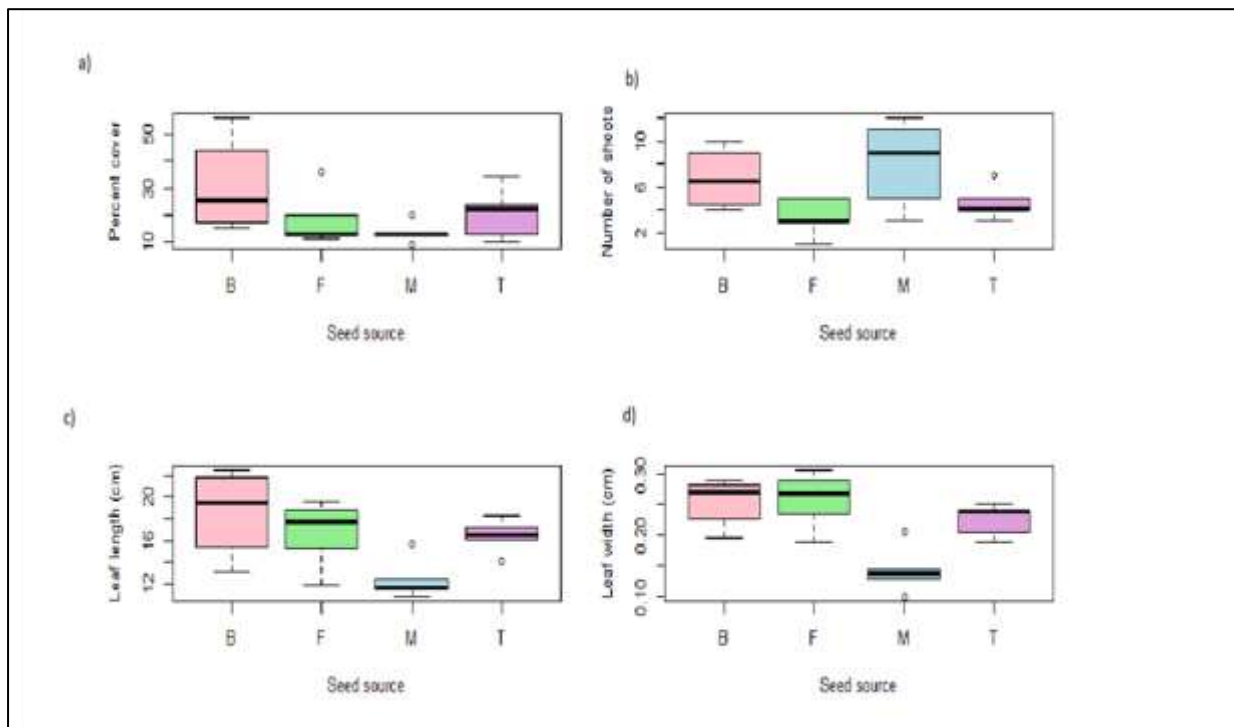


Figure 13. Comparison of shoot cover (a) shoot density (b) and leaf morphology leaf length (c) and leaf width (d) against seed sources across Scotland T = Tankerness, F = Finstoun, B = Burntisland, M = Moray. Each seed source had four corresponding plots see (Fig 4). *note Burntisland had less seeds planted due to lack of seeds.

Furthermore, a Principal Component Analysis (PCA, Fig 14) indicated that the first two principal components explained 81.8% of the total trait variation (PC1 = 52.5%, PC2 = 29.3%), with PC1 primarily associated with leaf size and PC2 associated with shoot cover and density. Although seed sources showed separation along these axes, a PERMANOVA showed no significant differences in overall trait composition among seed sources ($F = 1.97$, $R^2 = 0.28$, $p = 0.11$). Multivariate dispersion did not differ among seed sources (betadisper: $p > 0.05$).

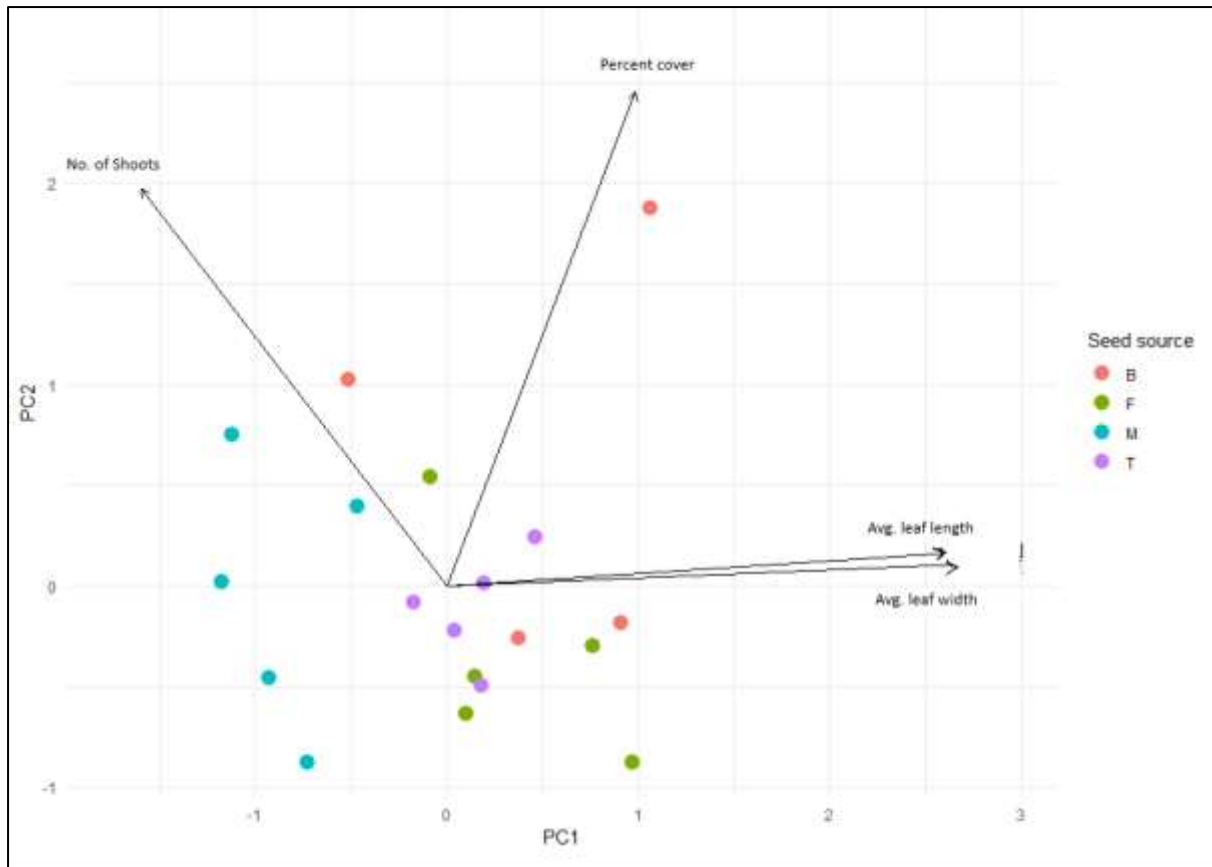


Figure 14. Principal Component Analysis of seagrass traits by seed source following planting in Burntisland in 2025. B = Burntisland, F = Finstown, M = Moray Forth and T = Tankerness

3.1.2. *Zostera noltii*

3.1.2.1. *Transplants over time*

Across all three transplanting sessions (Fig. 15 a–c), coverage of *Zostera noltii* increased between each monitoring period. In every graph, *Z. noltii* levels rose relative to the initial baseline (time 0), indicating ongoing accumulation over time. The greatest increase in *Z. noltii* was recorded in September 2024. Overall, each monitoring period demonstrated a clear upward trajectory in *Z. noltii* coverage, with September 2024 representing the peak increase across all patches and planting dates. From the total initial planting footprint of $\approx 6\text{m}^2$ of seagrass (760 x 10cm diameter cores), the total amount of *Z. noltii* within the 56 treatment plots covered 0.03ha. This is a 54 x increase in unit area. This is an absolute minimum estimate as we know that seagrass had expanded in area often way beyond the treatment plots and was proposed to be quantified using the drone, but this was not possible (see section 3.3). Further surveys in 2026 will be required to fill this data gap.



Figure 15. Increase in *Zostera noltii* patch area for the May 2023, November 2023, and May 2024 transplants, relative to the area in the preceding year. Bars represent mean $\pm$ SE. Grouped according to the number of planted cores, reflecting the four treatments defined in Fig 5 (5, 10, 15, and 30 cores).

3.1.2.2. Donor recovery

Donor hole recovery assessments in October 2025 showed that most holes within the subsample had healed. The subsample included 45 donor holes, representing 7% of the 640 holes created during 2023–2024 *Z. noltii* transplanting. Of these, 28 holes (62%) were fully healed (Y). 11 holes (24%) were partially healed (P). Six holes (13%) showed no signs of recovery (N) (Fig 16). This means that of the original $\approx 6\text{m}^2$ of seagrass collected, 4.5m^2 has recovered and 1.5m^2 remains damaged. To report our final restoration results, this damage figure will be removed from the gross change to yield a net restoration figure.

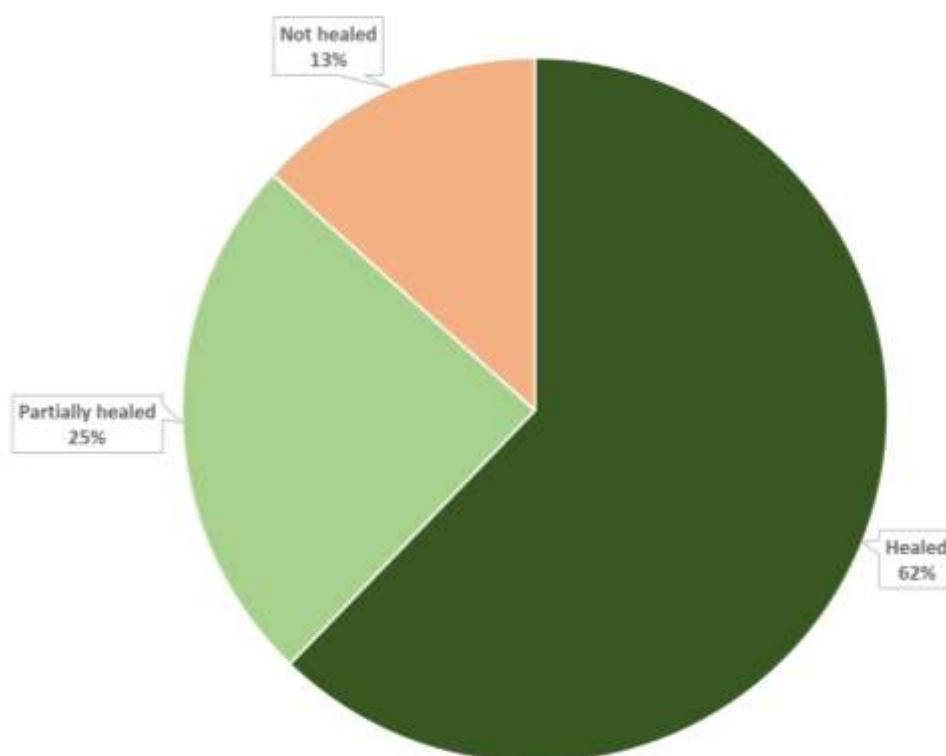


Figure 16. Recovery status of donor holes surveyed in October 2025, showing the number of sites that were healed, partially healed, or not healed ($n=45$)

3.2. Environmental data

Analysis of seagrass tissue nutrients revealed a stark contrast between the Firth of Forth and the cleaner, clearer reference sites, indicating excess nitrogen. Tissue nitrogen (Fig.17 and 18) levels at Burntisland and Tynninghame ranged from 2.8% to 4.0%, values that significantly exceed the threshold for pristine conditions ($<2.0\%$) observed in Orkney (and at other healthy *Zostera marina* sites in the Northern Hemisphere) (Duffy et al. 2022).

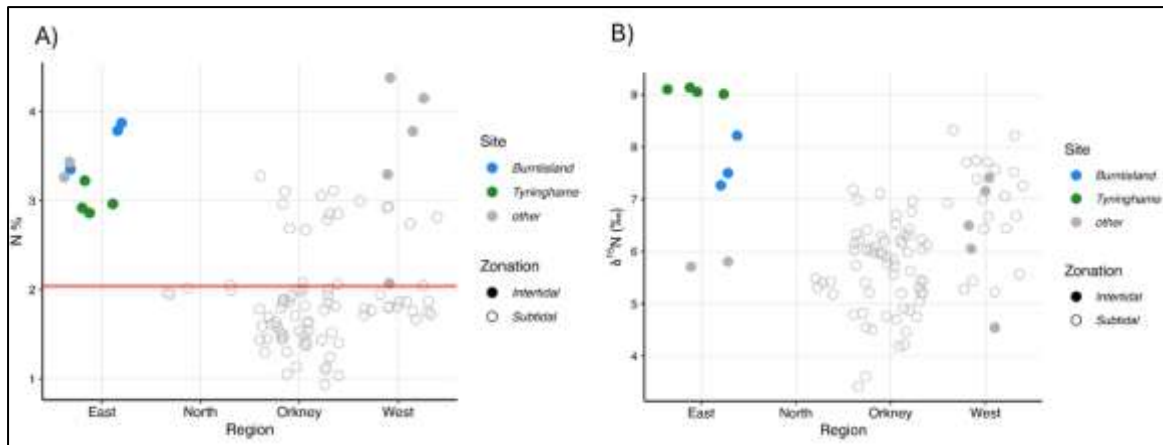


Figure 17. A) Spatial variability in leaf tissue nitrogen content (%N) of *Zostera marina* collected from intertidal and subtidal zones across four Scottish regions. The red line indicates the threshold for nitrogen repletion, with elevated values in the East region (Burntisland and Tynninghame) indicating nutrient enrichment relative to reference sites in Orkney and the West. B) Stable nitrogen isotope signatures ($\delta^{15}\text{N}$) in *Zostera marina* leaves from the Firth of Forth (East) compared to regional baselines. The enriched $\delta^{15}\text{N}$ values ($>7\text{‰}$) at Burntisland and Tynninghame are indicative of anthropogenic nitrogen sources, such as sewage or agricultural runoff, contrasting with lower values in Orkney and Western sites

This enrichment was further corroborated by the stoichiometric data, where the Forth sites displayed low C:N ratios of 12–16 (Fig 18). In ecological terms, a ratio below 14 indicates that the system is light limiting to growth, probably as a result of eutrophic conditions.

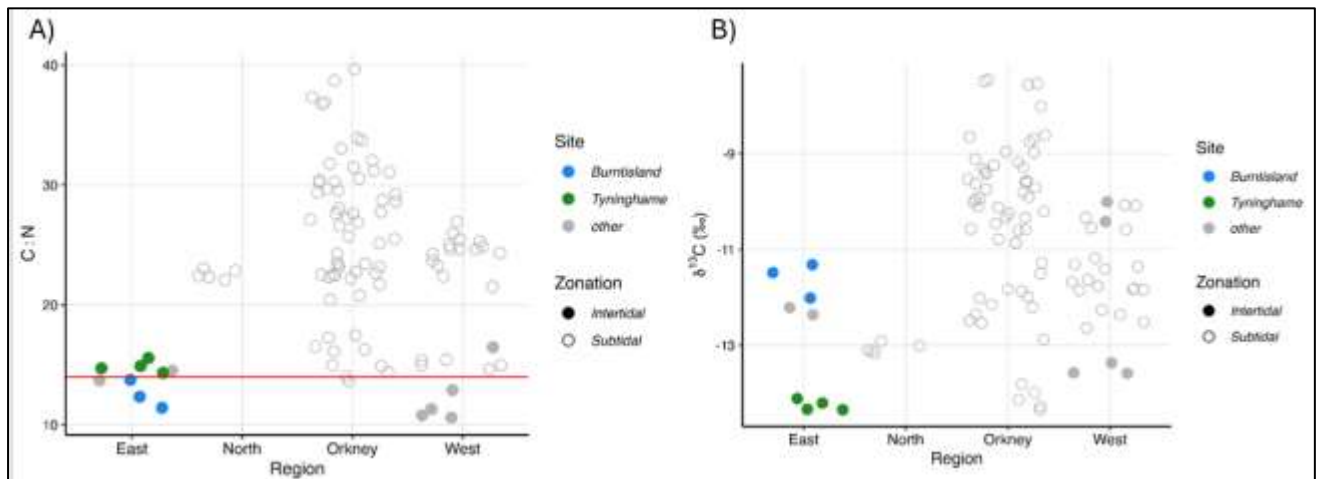


Figure 18. A) Stoichiometric Carbon:Nitrogen (C:N) ratios of *Zostera marina* leaf tissue. Low C:N ratios (<14) suggest light limitation, whereas higher ratios in Orkney and the West are indicative of high-light conditions typical of pristine seagrass meadows. B) Variation in stable carbon isotope ratios ($\delta^{13}\text{C}$) of *Zostera marina* leaves across Scottish survey sites. The distinct separation between Burntisland (enriched $\delta^{13}\text{C}$) and Tynninghame (depleted $\delta^{13}\text{C}$) within the East region reflects localised differences in photosynthetic rates, light availability, or inorganic carbon sources.

Crucially, the stable isotope analysis identified the origins of this excess nitrogen. The signatures at Tynninghame reached approximately 9‰, while Burntisland ranged between 7‰ and 8‰. In the context of UK coastal waters, values exceeding 8‰ are strongly linked to wastewater and sewage inputs, distinguishing these sites from the lower, more variable values in Orkney that reflect cleaner marine baselines or diffuse agricultural fertilizers. Furthermore, temporal data from Drum Sands indicated that this nitrogen stress is compounded by physical limitations; light intensity dropped below critical physiological thresholds in early autumn, coinciding with high seedling mortality (Figure 19).

These findings align with wider reports for the UK, where seagrass meadows in urbanised estuaries (like the Thames and Solent) often exceed global averages for tissue nitrogen (UK avg ~2.8%) due to inefficient sewage treatment and agricultural runoff. The Firth of Forth sites appear to follow this pattern of anthropogenic stress, contrasting sharply with the more pristine, nitrogen-limited meadows found in remote Scottish waters.

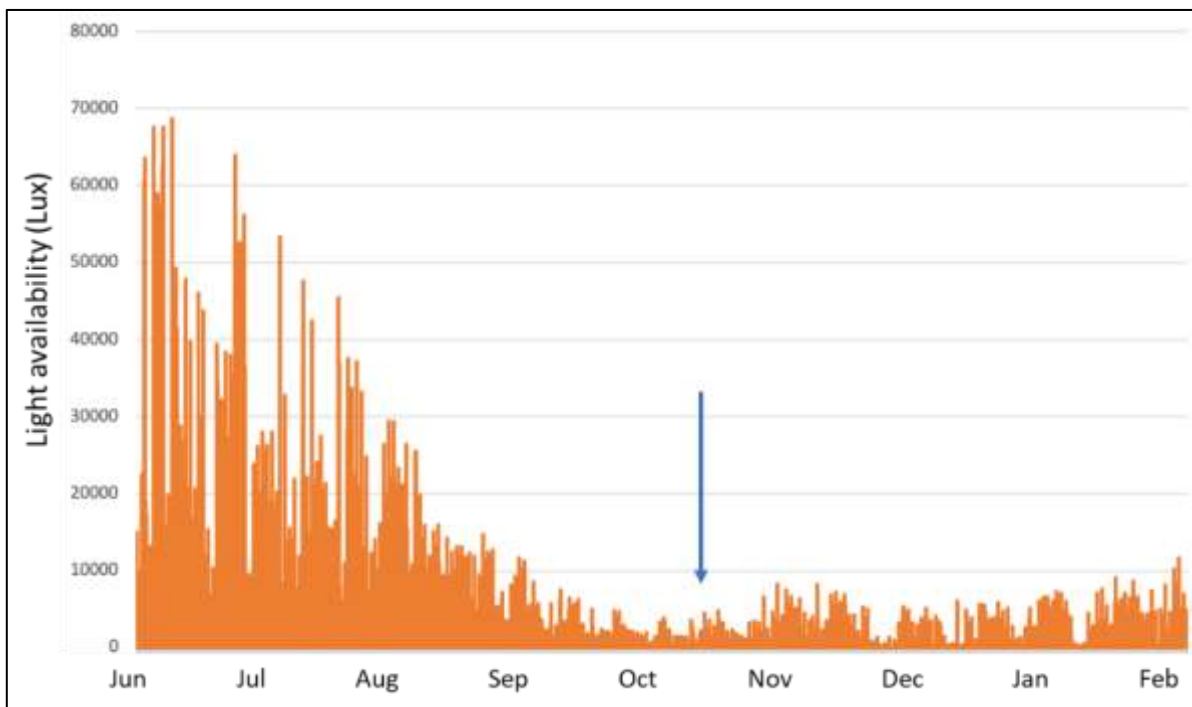


Figure 19. Temporal variation in benthic light intensity (Lux) at Drum Sands recorded using a deployable HOBO data logger. The blue arrow denotes the onset of critical light attenuation in early autumn, a period of reduced irradiance that coincided with the observed senescence and subsequent loss of the *Zostera marina* meadow.

The biodiversity data for Burntisland (Figure 20) paints a picture of a depauperate community, consistent with the poor environmental status indicated by the nutrient data. Burntisland displays one of the lowest species richness values ($S \approx 1.0$) among all surveyed UK sites. It is significantly lower than the reference sites in Orkney ($S \approx 1.8$) and Loch Craignish ($S > 2.0$), suggesting that the high nutrient load is excluding sensitive species. The suppression is even more evident in species abundance. Burntisland's abundance score ($N \approx 1.0$) is a fraction of that seen at high-biodiversity sites like Seaview on the Isle of Wight ($N >$

9.0) or the Fleet lagoon ($N \approx 5.0$). This indicates that the seagrass meadow at Burntisland supports a much sparser community of mobile invertebrates than healthier meadows.

This eutrophic state likely promotes epiphytic algal growth (algae growing on seagrass leaves), which can smother the seagrass and create hypoxic conditions at the leaf surface. While some opportunistic grazers thrive in such conditions, sensitive epifauna often decline. The data support Hope's (2025) conclusion that excess nitrogen acts as a stressor rather than a fertiliser for the associated animal community. The data from Hope (2025) validates the concern raised by the isotope analysis. The Firth of Forth seagrass is not just enriched; it is biologically impaired. The high Nitrogen / $\delta^{15}N$ levels are directly associated with minimal biodiversity where the meadow supports significantly fewer species and fewer individuals than equivalent meadows in Orkney or the West of Scotland.

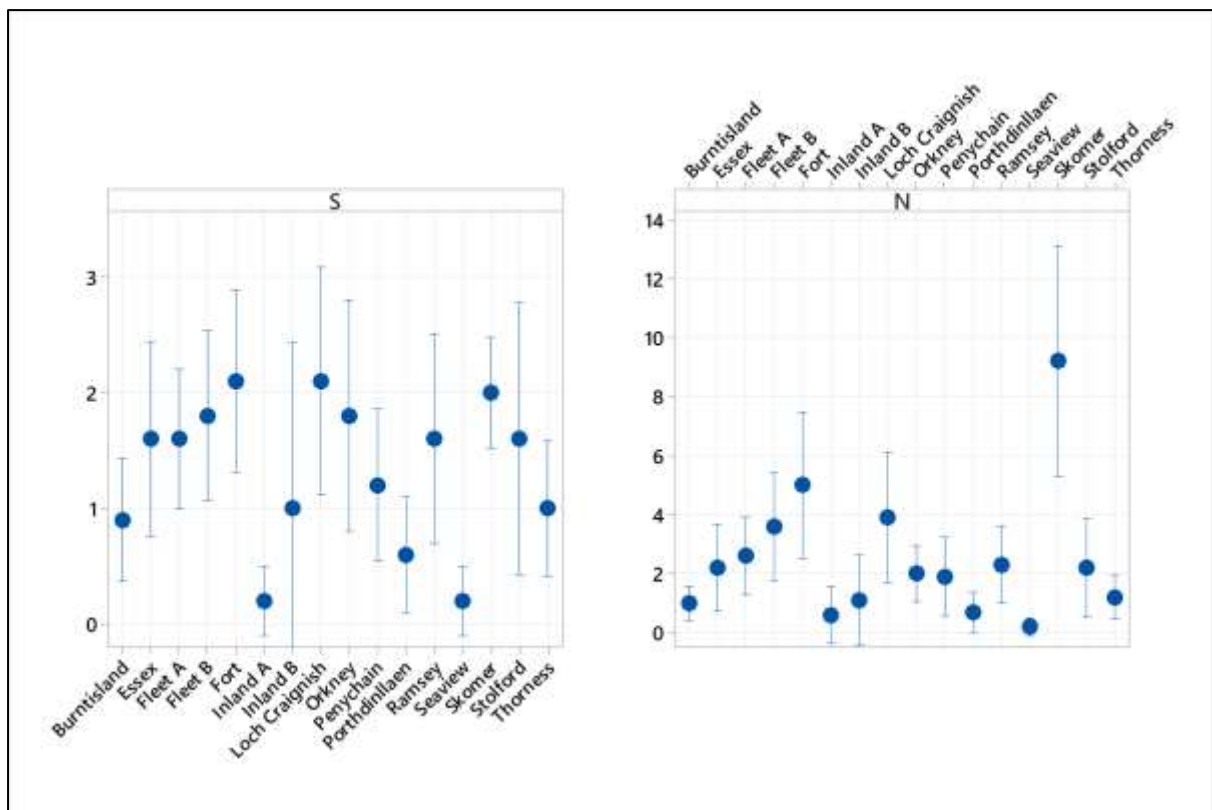


Figure 20. Comparative assessment of motile epifaunal biodiversity associated with *Zostera marina* meadows across United Kingdom study sites. Panels display (A) Species Richness (S) and (B) Species Abundance (N), showing suppressed biodiversity metrics at Burntisland relative to higher-diversity reference sites (e.g., Seaview, Loch Craignish). Data adapted from Hope (2025).

3.3. Seagrass mapping

3.3.1. Orthomosaics between 2023 and 2025

Visual examination of RGB orthomosaics generated from drone imagery captured in 2023 and 2025 provided an estimate of seagrass extent, showed areas of bait digging, and indicated significant sediment changes at two of the Restoration Forth sites. At all three sites, dense seagrass was visible in RGB imagery as areas dark green in colour (Figure 21: subsets 2, 4, 5, 6). Sparse seagrass was not apparent in the RGB imagery. Bait digging activity was visible as symmetrical light-grey stripes in imagery from 2025 at Burntisland (Figure 21: subset 3), but was not in proximity to visible areas of seagrass. Comparison between 2023 and 2025 imagery at Drum Sands indicated reduction in the extent of raised light-grey sediment in several areas across the site, with expansion and disorganisation of previously existent channels (Figure 21: subset 1). At Tynninghame, comparison of 2023 and 2025 imagery indicated significant change in sediment characteristics, from lighter brown sand or mud to a darker grey rocky matrix (Figure 21: subset 6). Additionally, coarse sediment at the base of the beach is reduced in 2025, the coastline is eroded, and seagrass visible in 2023 is no longer visible (Figure 21: subset 6). Unfortunately, 2023 imagery at Burntisland was poor in quality, preventing comparison with 2025 imagery.

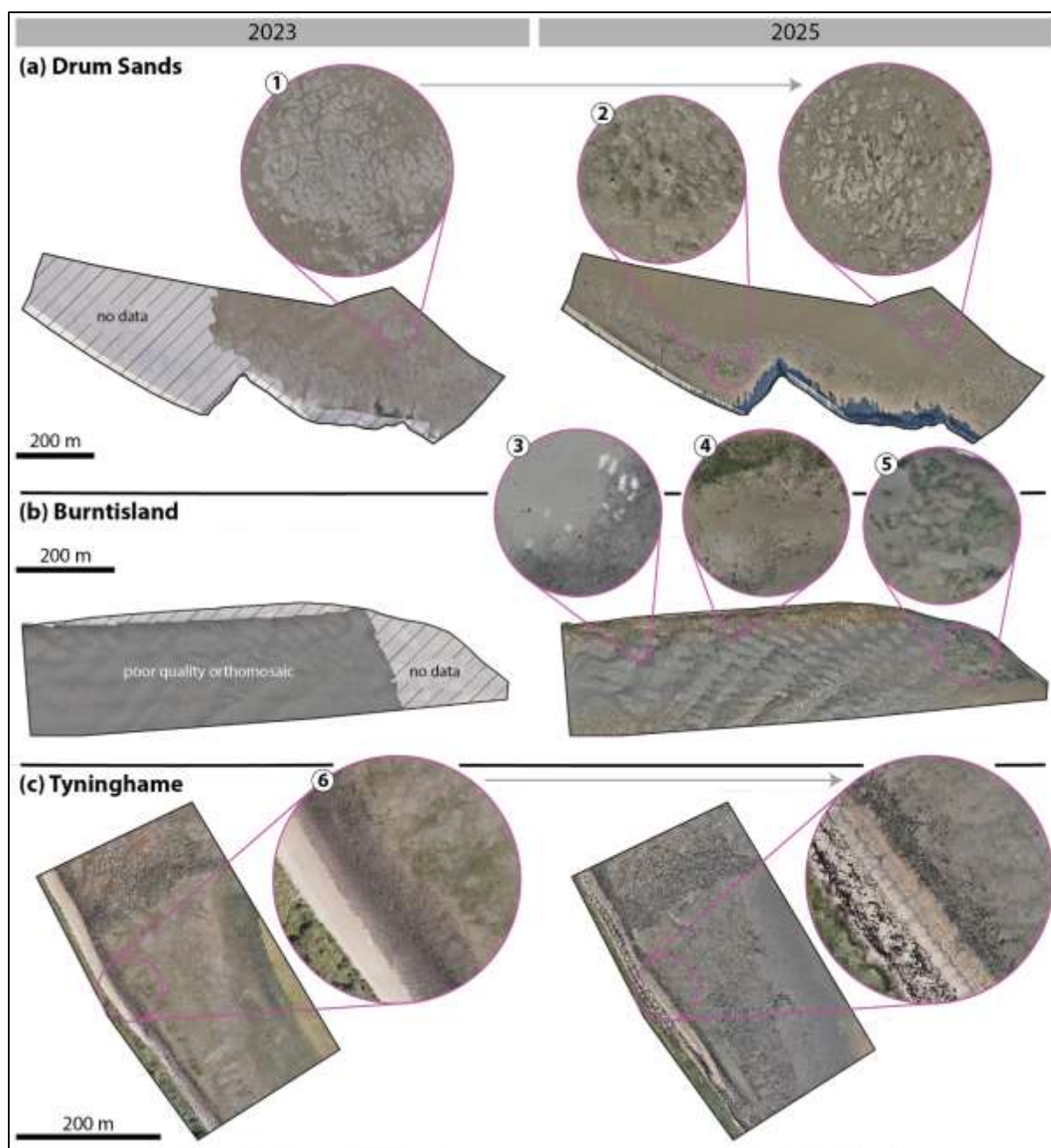


Figure 21. Orthomosaics generated from RGB drone imagery show seagrass meadows, disturbance features, and changes in sediment at Restoration Forth sites. **(a)** Drum Sands imagery from 2023 and 2025 indicates [1] erosion of areas of raised sediment associated with [2] *Zostera noltii* meadows. **(b)** Burntisland imagery from 2023 is poor quality, but 2025 imagery indicates [3] areas of bait digging as symmetrical light-grey features, and [4, 5] the presence of *Zostera marina* meadows in pools along the foreshore. **(c)** Tynninghame imagery from 2023 and 2025 shows [6] coastal erosion, significant change in sediment characteristics and loss of visible *Zostera sp.* meadow.

3.3.2. NDVI assessment

Visual assessment shows that extensive areas with mid-range NDVI values (0.4 – 0.6) at Drum Sands and Burntisland correspond with observations of seagrass from fieldwork

(Figure 22a, 22b). However, typically higher NDVI values at Tynninghame represent macroalgae of the *Fucus* genus (Figure 19c).

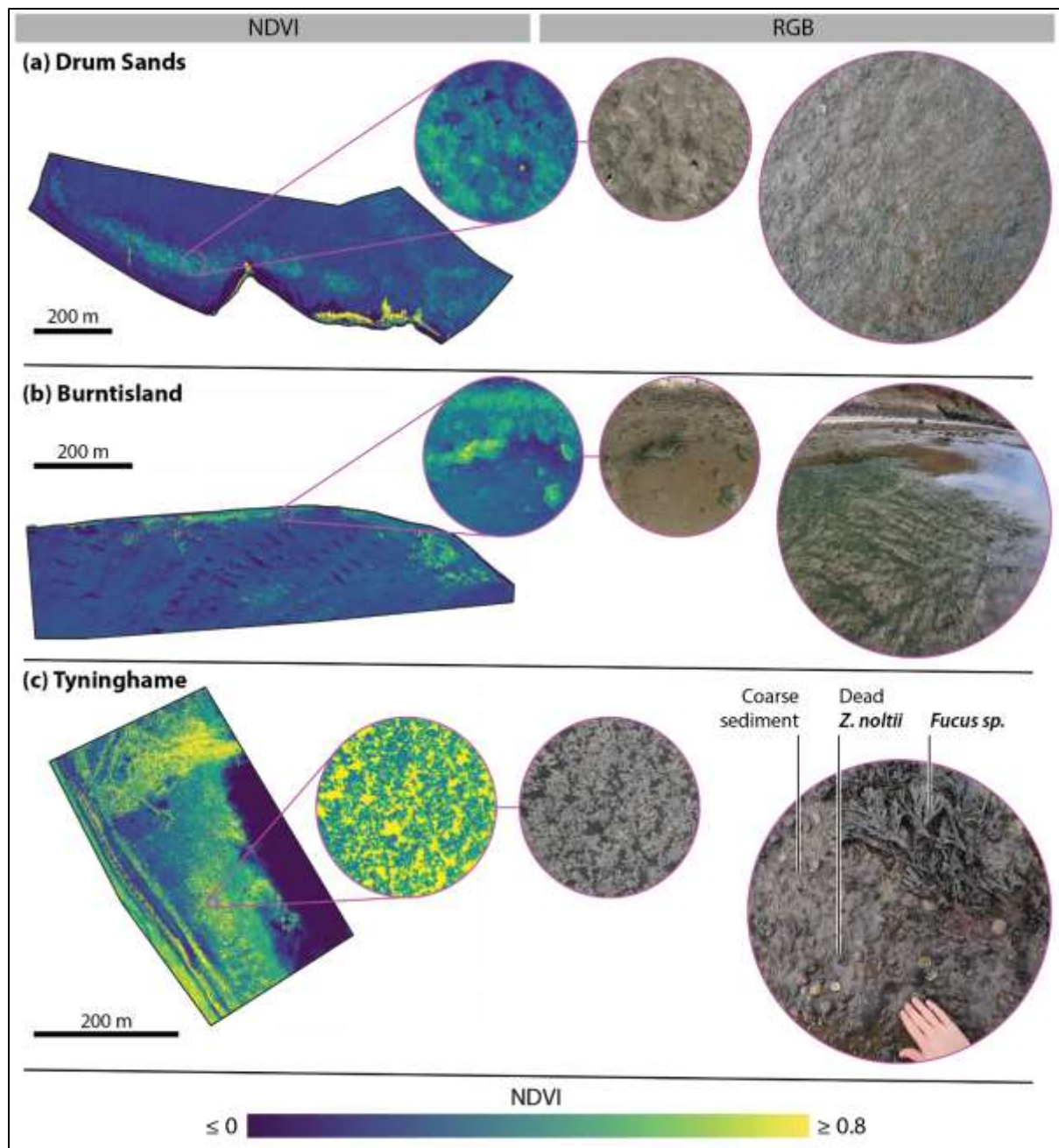


Figure 22. Mid-range NDVI values in 2025 drone imagery are indicative of seagrass at Drum Sands and Burntisland, but high values more typically correspond with macroalgae, particularly at Tynninghame. **(a)** *Zostera noltii* meadows at Drum Sands are represented by mid-range (0.4 – 0.6) NDVI values, which are present across large areas of the site. **(b)** *Zostera marina* meadows at Burntisland are represented by mid-range (0.4 – 0.6) NDVI values, which are present along the foreshore against concrete sea defences and at the eastern end of the site. **(c)** Higher NDVI values at Tynninghame (0.6 – 0.8) were representative of macroalgae, with rocky substrate and *Fucus sp.* present across much of the site.

3.3.3. Random Forest models

Random Forest models predicted the extent of dense *Zostera noltii* meadows at Drum Sands, and *Zostera marina* meadows at Burntisland (Figure 23). At Drum Sands, model accuracy using withheld training data was 98% and *Zostera noltii* was predicted across an area totalling 5 hectares (Figure 23a). However, when comparing model predictions with field observations of seagrass density, the model failed to predict sparse seagrass accurately (accuracy = 50%; Figure S6). As such, seagrass was inconsistently detected in areas of active restoration, although patches were predicted around some *Z. noltii* transplant transects (Figure 23a: subsets 1, 2). At Burntisland, model accuracy using withheld training data was 93% and *Zostera marina* was predicted across an area totalling 0.56 hectares (Figure 23b). We did not have sufficient ground-truth data from Burntisland to calculate accuracy within restoration areas. The model successfully delineated the extent of the *Zostera marina* meadow adjacent to the restoration plots (Figure 23: subset 3) and identified patches of *Z. marina* that were present in tidal channels seaward from the plots (Figure 23: subset 4).

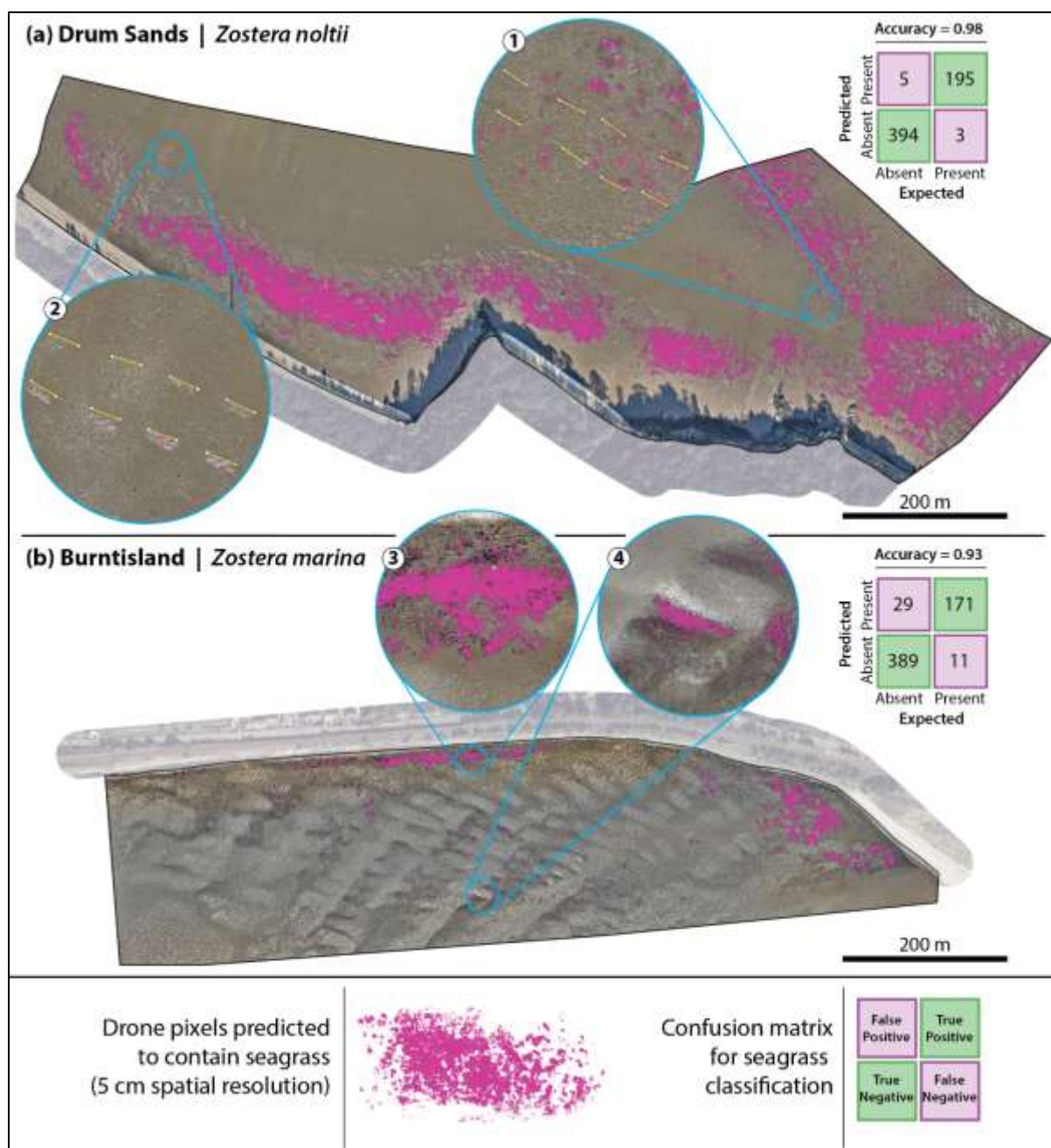


Figure 23. Random Forest models accurately predict the presence of seagrass within multispectral drone pixels at a 5 cm spatial resolution, indicating the extent of seagrass at the Drum Sands and Burntisland sites. **(a)** Predicted extent of *Zostera noltii* at Drum Sands, with confusion matrix indicating model accuracy. Sparse *Z. noltii* within transplant transects was detected by the model [1, 2], though with less accuracy than field-based monitoring (see sections 2.1.3.4 and 2.2.2). **(b)** Predicted extent of *Zostera marina* at Burntisland, with [3] meadow running parallel to areas of planting in shallow pools. The model indicated some [4] dense patches of seagrass further from the foreshore than anticipated.

3.3.4. Coastal erosion and change at Tynninghame

We observed near-total loss of seagrass at the Tynninghame site between project inception in 2023 and closeout monitoring in 2025, concurrent with erosion of the coastline and change

in sediment characteristics (Figure 24). Areas of seagrass visible in 2023 drone imagery (Figure 24a), known from field observations to contain both *Zostera noltii* and *Zostera marina*, were significantly reduced in area when mapped with handheld GPS by groups of volunteers in October 2025 (Figure 24b). Experience of Restoration Forth Seagrass Officers indicates that the change in sediment and loss of seagrass may have been caused by large storm events, such as Storm Ciarán on the 1st to 2nd November 2023 (BBC News, 2024)

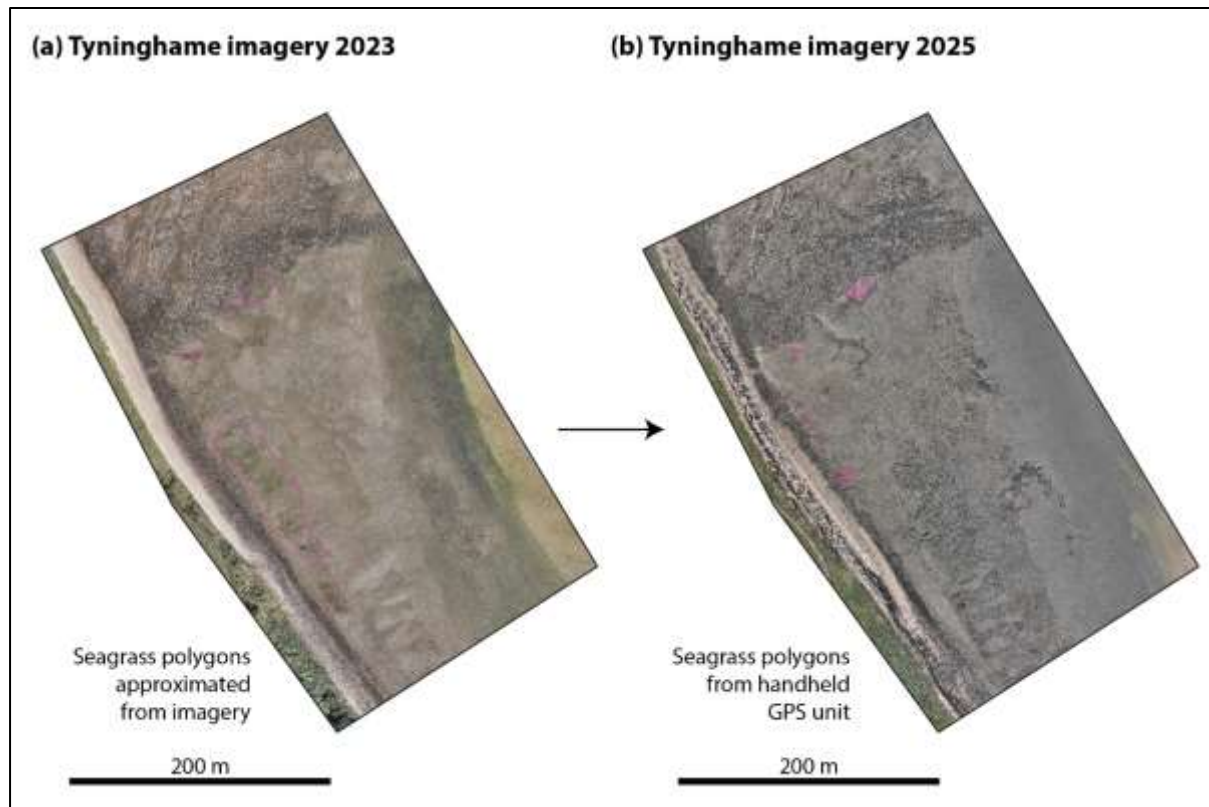


Figure 24. Imagery at Tynninghame from 2023 and 2025 shows the loss of seagrass meadows from the site, concurrent with a shift in sediment characteristics. **(a)** Areas of dense seagrass are visible in 2023 Tynninghame imagery as green, textured patches along the shoreline and the edge of the bedrock to the north. Coarse purple polygons represent approximate extent of seagrass and were created manually from visual inspection of imagery. **(b)** No seagrass is visible in 2025 imagery. Purple polygons represent extent of seagrass documented during a survey with volunteers in October 2025, using handheld GPS.

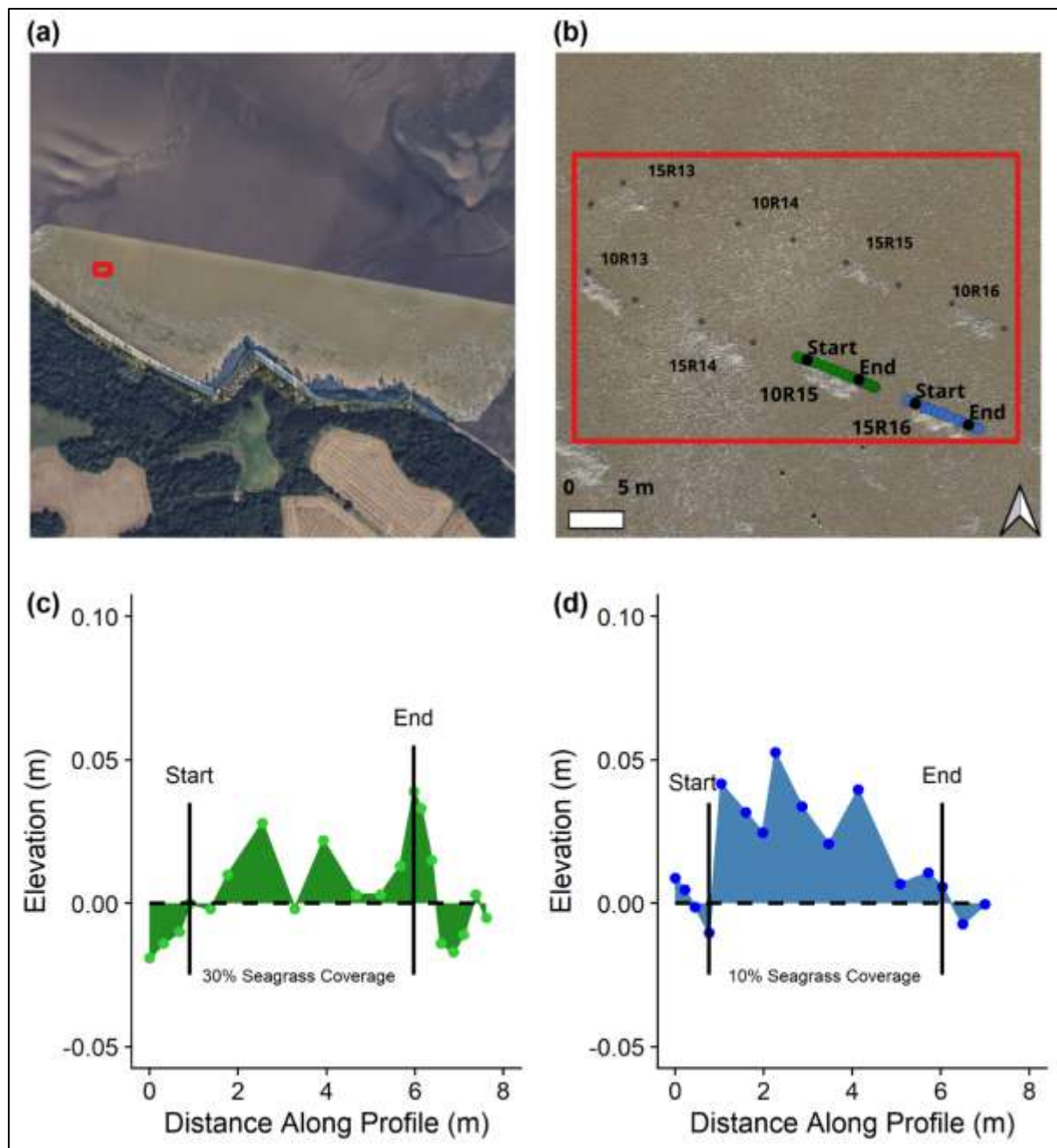


Figure 25. Drone imagery and DGPS elevation profiles through *Zostera noltii* restoration transects indicate accretion of sediment within the restored area. **(a)** Overview of the Drum sands site, where *Z. noltii* restoration efforts were conducted (red box). **(b)** Drone imagery over *Zostera noltii* restoration transects, with lighter-coloured sediment indicating accretion within most transects. **(c, d)** Elevation profile of plots 10R15 and 15R16, where the base elevation is the average of six points taken beyond each end of the transect, and points within each transect showing generally raised elevation measurements. The overall percent coverage of seagrass from field monitoring is indicated below the elevation profile (see section 2.1.3.4).

3.3.5. Sediment accretion by *Zostera noltii*

Drone imagery and DGPS elevation measurements taken along *Zostera noltii* transects at Drum Sands indicate that growth of transplanted *Z. noltii* cores (see section 2.1.3.) caused sediment fixation, resulting in accretion within most plots in a subset of the site (Figures 25a, 25b). Drone imagery recorded lighter-coloured sediment coincident with the location of each *Zostera noltii* transect, corresponding with field observations of raised sediment along restoration transect (Figure 25b). Elevation measurements within plots 10R15 and 15R16 show that average sediment height is greater within each plot than outwith the plot (Figures 25c, 25d; 10R15: + 3.4 cm; 15R16: + 2.9 cm).

3.4. Community engagement in seagrass restoration

Between 2022 and 2025, Restoration Forth staff ran numerous engagement events specifically linked to seagrass, including general engagement, adult learning, and citizen science events, to raise awareness about the project and provide training opportunities to upskill local volunteers (Figure 26). The engagement strategy from 2023 onward focused on providing greater opportunities for adult learning and citizen science to train volunteers in seagrass data collection, seed processing, and planting elements of the project. This decision was to strengthen the conservation and restoration capabilities of the local community members in the Firth of Forth. Three outputs which exemplify the community efforts towards strengthening the work conducted by Restoration Forth staff members include the creation of spatial maps of the seagrass meadows at Belhaven and Burntisland and the 'how to' videos explaining fixed transect monitoring and spatial mapping.



Figure 26. Restoration Forth volunteers participating in common eelgrass restoration trials over the project’s three years (top left – Restoration Forth funders Aviva and Scottish Power Foundation representatives with MP Douglas Chapman, bottom left – volunteers and staff after planting trials 2023, top right – Esther demonstrating how to plant, bottom right – volunteers at Drum Sands 2025).

4. Discussion

4.1. Seagrass Restoration

The Restoration Forth project represents a critical strategic intervention aimed at reversing the long-term historical loss of seagrass extent in the Firth of Forth and leading the way in seagrass restoration in Scotland. By attempting to restore 4 hectares of marine habitat through community-linked action in an industrial human-dominated estuary, this study provides essential insights into the complexities of re-establishing and extending populations of *Zostera marina* (common eelgrass) and *Zostera noltii* (dwarf eelgrass) within the Firth of Forth. The results from the 2022–2025 period highlight a dichotomy of success between the two species, influenced heavily by environmental stressors, planting methodologies, and site-specific conditions.

The restoration of *Z. marina* using seeds encountered significant hurdles across all three study sites. The 2023 trial was largely unsuccessful, with no viable populations formed. This was in spite of initial successes at Drum Sands, with high emergence rates, creating great

excitement across the project. The limited resilience of the plants to the low light availability encountered in the latter part of 2023 (due to the heavy rain exacerbating poor water quality) meant that all but a couple of the plants died at Drum sands. Physical environmental factors appeared to be the primary drivers of this failure at Burntisland and Tynninghame; sediment movement completely buried the plots, while Tynninghame yielded no seedlings. This underscores *Z. marina*'s vulnerability to environmental change, particularly those that affect the physical structure of the seabed that is highly modified and mobile across the Firth of Forth (Wang et al. 2026). We also speculate that the coastal erosion at Tynninghame is over time, making the locality less suitable for seagrass (see section 3.3), even though historic records of its large extent do exist (Thomas 2020).

In 2024, despite a revised approach to planting *Z. marina* seed, success remained localised. Seed was planted in targeted tidal pools in areas observed to be stable over time to avoid desiccation and burial (Boese, Robbins & Thursby 2005). While Drum Sands and Tynninghame experienced low seedling emergence, rapid declines and almost total winter mortality by February 2025, Burntisland emerged as the most successful site for this species. The plots planted in Burntisland in both 2024 and 2025 still contained seagrass clumps by Dec 2025. At Burntisland, there is surviving seagrass over a good area (0.17ha) but at a low density (1.25/m²) comprised of clumps, not continuous seagrass. The survival of other seagrass at these sites indicates that these may proliferate after winter dieback; however, it's important to recognise that the seagrass at Burntisland is among the most nutrient-rich of any site surveyed in the UK (Hope 2025), with strong epiphytic algal cover likely limiting rapid growth.

The collection of *Zostera marina* seeds from Orkney was guided by advice from NatureScot. The strategy aimed to leverage the robust and extensive seagrass meadows of the north to jumpstart restoration in the Firth of Forth. However, this move introduced a significant environmental mismatch, seeds were harvested from the crystal-clear, subtidal depths of Orkney and transplanted into the sediment-heavy, low-light intertidal zones of the Forth. *Zostera marina* shows extensive adaptation to varied environments (Boyé et al. 2022), with such adaptation leading to the creation of environmental ecotypes (Ruesink et al. 2022). The donor seeds collected from Orkney were likely genetically adapted to stable, deep-water conditions, they were potentially ill-adapted to survive the fluctuating temperatures and limited light availability characteristic of the Forth's muddy shoreline. Poor survival of *Z. marina* plants in 2023 ultimately led to permission from NatureScot to collect a small number of seeds locally in the Firth of Forth and in the Moray Firth.

An experimental plot at Burntisland in 2025 investigated whether the origin of *Z. marina* seeds (Orkney, Moray Firth, or local) influenced restoration success. While seed source accounted for 28% of the variation in structural traits, the results did not indicate a significant overall difference in traits or density across seed sources. It's likely that overall variability within populations may have masked any differences between populations. This monitoring was conducted in the autumn, further analysis of these populations will be

required in 2026 in order to reach a decision on their viability after winter storms and light stress (Govers et al. 2015).

In contrast to the planting of *Zostera marina* seeds, the intertidal *Z. noltii* restoration at Drum Sands showed a clear upward trajectory but was conducted at a much smaller scale. Utilising a transplanting method (sediment cores that included both sediment and plant material) rather than seeds enabled the project to avoid introducing invasive non-native species (INNS) and to maintain the integrity of the rhizome and root systems. The proximity of sufficient seagrass to serve as a local donor facilitated the use of locally adapted plants.

The data indicate that the *Z. noltii* area increased consistently across all monitoring periods following the initial 2023 and 2024 plantings. Peak increases were observed in September 2024, with continued expansion through 2025. Transplants were conducted in both spring and winter, with no observed differences between the two time points. By the end of the three-year trial, at least 0.032 ha planting area (this is a net estimate removing the impact of any remaining slow recovery from transplant collections – see discussion below) was successfully covered by seagrass. This is at least a 54 x increase in area to that originally planted. This high success rate suggests that transplanting mature cores of *Z. noltii* is an effective method for the Firth of Forth's intertidal zones. The core method was chosen in preference to seed-based methods because the ability to collect sufficient numbers of *Z. noltii* seeds was constrained by their small size and proximity to the sediment. The relative success of these plantings of *Z. noltii* relative to *Z. marina* also reflects the niche of this species. *Z. noltii* is a low-light-adapted species (Erftemeijer & Lewis 2006) requiring a high shelter (typical of Drum Sands) that is able to deal with the increased desiccation stress of the upper intertidal (Azevedo, Dias & Lillebø 2016).

A vital component of ethical restoration is ensuring that donor sites are not permanently degraded and understanding the overall net benefit of the restoration rather than just considering the change at the restoration site. For *Z. noltii*, in which 760 cores were removed over two years ($\approx 6\text{m}^2$), a 2025 assessment indicated that 62% of donor holes had fully healed and 24% were partially healed. Only 13% showed no signs of healing after two years. We conclude that upto 1.5m^2 of seagrass was damaged through collection but with a benefit of a further 323m^2 .

These findings suggest that the regulated removal of donor material, specifically taking only one core per square metre in high-density areas, has a minimal long-term impact on the health of the donor meadow. Within our overall estimates of the effect of restoration, we have included this data to provide a net estimate of increase rather than just a gross one.

4.2. Seagrass Mapping

Remote sensing with drones captured the extent of seagrass meadows at Restoration Forth project sites, in addition to disturbances such as bait digging, coastal erosion, and changes in sediment characteristics. Visual assessment indicated that mid-range NDVI values were

representative of seagrass at Drum Sands and Burntisland, while typically higher NDVI values at Tynninghame represented widespread *Fucus sp.* macroalgae. Random Forest models specific to each site were able to predict the presence of seagrass within drone pixels with a high degree of accuracy (> 90%), although comparison with ground-truth data indicated a high rate of false negatives in areas of sparse seagrass. In total, models predicted extensive *Zostera noltii* meadows at Drum Sands (5 ha) and smaller *Zostera marina* meadows at Burntisland (0.56 ha). At Tynninghame, coastal erosion and changes in sediment characteristics were concurrent with an almost total loss of the seagrass meadow that existed at the beginning of the Restoration Forth project. Conversely, at Drum Sands DGPS elevation measurements evidenced raised sediment in areas that *Zostera noltii* cores were transplanted into, indicating that sediment accretion had occurred. Overall, remote sensing successfully documented the extent of seagrass at each Restoration Forth site, in addition to capturing critical evidence of processes affecting the Forth's seagrass meadows and associated restoration efforts.

4.2.1. Simple drone imagery was an invaluable tool

High-resolution imagery from drones indicated the extent of intertidal seagrass meadows and captured information on associated environmental pressures, making drone surveys invaluable to our monitoring of the Restoration Forth project. At all three Restoration Forth sites it was possible to approximately locate seagrass meadows from their colour and texture in RGB imagery, though calculation and inspection of the NDVI provided a clearer visual indication of overall seagrass extent. Similarly, simple visual inspection of RGB imagery from the beginning and closeout of the project documented processes that impacted both the health of naturally occurring seagrass meadows and the success of restoration efforts. In particular, imagery showed that the Tynninghame site experienced significant coastal erosion and changes in sediment characteristics, concurrent with an almost total loss of seagrass. The aerial perspective and spatial continuity of data afforded by drone imagery makes the scale of change at Tynninghame visually striking, documenting change beyond what could have been achieved through field ecology data collection alone. We believe the changes at Tynninghame were driven by Storm Ciarán on the 1st to 2nd of November 2023, which damaged coastlines with a generally northerly aspect throughout East Lothian (BBC News, 2024). Drone imagery also documented changes in sediment characteristics at Drum Sands and evidence of bait digging at Burntisland, although effects on seagrass extent were less apparent. To move beyond generalised statements about broad-scale change at the site, more sophisticated analyses of drone imagery are required, in combination with more extensive and targeted field data collection. Nonetheless, this work reinforces the value of simple visual inspection of drone imagery for monitoring of intertidal seagrass restoration.

4.2.2. Generation of orthomosaics is difficult in intertidal environments

While simple visual inspection of orthomosaics was valuable across the Restoration Forth sites, more sophisticated analyses rely on standardised data collection, which is extremely

challenging in nearshore intertidal environments. Firstly, the drone platform and sensor we used changed between project initiation and closeout. As a result, we gained multispectral bands, the ability to calculate the NDVI, and the ability to apply robust Random Forest classification models to the data. However, changes in the area surveyed by each drone platform and differing sensor spectral characteristics meant we were unable to perform analyses of change between imagery pairs (see Figures 18a, 18b). Secondly, best-practice guides recommend the use of ground-control-points (GCPs) to enable accurate orthorectification and geolocation of drone orthomosaics during processing and validation (e.g., Assmann *et al.*, 2019). At the three Restoration Forth sites, placement of GCPs was limited by sediment characteristics (difficulty moving in mud) and the rate of tidal encroachment during the survey period. This was most pronounced at Drum Sands, with GCP placement constrained to a narrow band of beach bordered by tall deciduous trees, which affected DGPS accuracy when surveying the GCPs and hindered drone imagery orthorectification. In turn, the accuracy of model validation with field-collected ground-truth points may be affected by a spatial mismatch between each point and its respective classified drone pixel. Finally, it would be extremely difficult – or impossible – to standardise spectral reflectance between imagery pairs collected in different tidal conditions. Variance in tidal height and drying time will drive differences in the spectral characteristics of a surface, even when the surface itself has not changed, and this could lead to spurious results in analyses based on spectral data. For the aforementioned reasons, we were unable to conduct sophisticated drone-imagery analyses of change over time at Restoration Forth sites, and we recommend that further work be undertaken to resolve these.

4.4.3. Random Forest models

We were able to delineate the extent of seagrass at Restoration Forth sites by applying Random Forest models to high-resolution (5 cm) multispectral drone imagery, but model insensitivity to sparse seagrass prohibited monitoring of growth in restoration plots. While it is well established that multispectral drone imagery supports classification of intertidal seagrass extent with a high degree of accuracy (e.g., Duffy *et al.*, 2018; Nahirnick *et al.*, 2019; Simpson *et al.*, 2024), there is less evidence for the successful classification of sparse seagrass in areas of active restoration. Areas of restoration pose additional challenges - where seagrass is being restored percent coverage and biological characteristics are likely to be inconsistent across space and time. Seagrass within restoration areas is also inherently limited in extent, in-turn limiting the volume of representative training data that can be supplied to machine-learning models. For the Restoration Forth sites, we overtrained classification models to maximise accuracy on naturally occurring seagrass at each site, which likely further impeded model accuracy in restoration areas. Despite these issues, Random Forest models detected some restored seagrass at our Drum Sands site (see Figure 20: subsets 1 and 2). Increasing the accuracy of drone imagery for the monitoring of seagrass restoration requires more extensive and systematic ground-truth data collection,

and efforts to establish a minimum spatial resolution for the consistent detection of sparse seagrass.

3.4.4. Sediment accretion in areas of restored *Zostera noltii*

Through field observations and DGPS measurements we established that some *Zostera noltii* transplant transects have accreted sediment over the duration of the Restoration Forth project, in line with studies of sediment fixation in *Zostera noltii* meadows (e.g., Widdows *et al.*, 2008). Sediment accretion in areas populated by *Zostera noltii* may explain the extensive light-grey sediment colouration in drone imagery of Drum Sands, which is concurrent with mid-range NDVI values (0.4-0.6) and model predictions of seagrass extent. Importantly, this observed difference in sediment colouration may – in cases where it is caused by sediment fixation – be a reliable proxy for the presence of *Zostera noltii*. Indeed, it is likely that this visual characteristic is inherently accounted for in the Random Forest classification for Drum Sands. Sediment accretion, or binding, in seagrass meadows provides important ecosystem services (Widdows *et al.*, 2008; do Amaral Camara Lima *et al.*, 2023) and detection of this phenomenon in drone imagery would be a novel use-case that we recommend as a focus of future research.

5. Project Conclusions and Recommendations

5.1. Restoration outcomes and overall success

The restoration activities delivered meaningful and encouraging progress towards rebuilding seagrass in the Firth of Forth. Across the programme, a total of 1.65 hectares of seagrass were planted, establishing multiple restoration plots that provided clear evidence of successful establishment under challenging conditions. Although this did not expand to the initially anticipated 4 hectares, and significant unsuccessful plantings occurred, the work demonstrated that seagrass restoration is feasible in this dynamic and challenging environment and that carefully targeted interventions can achieve measurable ecological gains.

Within a confirmed restored area of 0.17 hectares, *Zostera marina* reached a mean shoot density of 1.25 shoots m⁻², indicating that planted patches can survive and persist beyond initial deployment. These surviving units represent important nuclei for future recovery, as even small patches can generate positive ecological feedbacks by stabilising sediments, reducing local hydrodynamic stress, and gradually improving water clarity (Maxwell et al. 2016). Such processes increase the likelihood of natural expansion over time, meaning that the true restoration impact may continue to grow beyond the lifetime of the project.

Particularly notable was the strong performance of *Zostera noltii* at Drum Sands that by the end of the project covered an area of 54x its original planted area, demonstrating that large-scale intertidal restoration is highly viable in the region. This success provides a practical and scalable pathway for future work and highlights the importance of species selection aligned with local environmental conditions. Meanwhile, the incremental but consistent gains observed for *Z. marina* at Burntisland show that, with continued refinement of methods and site targeting, further recovery is achievable.

Overall, the restoration programme successfully established functioning seagrass patches covering an area of just over 0.2ha. Although they remain at low density and very patchy, these achievements have advanced planting and monitoring methodologies, and created a robust evidence base to guide future restoration in the Forth. Rather than representing an endpoint, these achievements provide a strong foundation for long-term but slow meadow recovery and expansion.

5.2. Supporting evidence from mapping and monitoring

Drone-based mapping documented the wider seagrass landscape and confirmed the continued presence of substantial meadow areas at Drum Sands (5 ha) and Burntisland (0.56 ha) by project close-out in 2025. While sparse seagrass within restoration plots remained difficult to detect remotely, field monitoring confirmed establishment and survival at several planting sites. This demonstrates that restoration success can occur at spatial scales below

the detection limits of current remote sensing approaches, reinforcing the value of detailed ecological monitoring alongside aerial surveys.

At Tynninghame, significant erosion and sediment change resulted in near-total loss of the pre-existing meadow during the project period. Although this limited restoration outcomes at that site, the observations provide valuable insight into the dominant physical drivers shaping seagrass persistence in the Firth, allowing future restoration to be better targeted towards more stable environments.

5.3. Key lessons from restoration activities

The programme highlights several positive and actionable lessons:

- Small, well-established patches can survive and provide nuclei for future meadow expansion.
- Intertidal restoration, particularly with *Zostera noltii*, offers a reliable and scalable restoration pathway.
- Targeted site selection and adaptive monitoring significantly improve restoration outcomes.
- Even modest early successes generate ecological feedbacks that can enhance long-term recovery potential.

These lessons reinforce that restoration success in dynamic coastal systems should be evaluated not solely by hectares restored within a short timeframe, but by the establishment of resilient patches capable of persisting and expanding naturally.

5.4. Strategic recommendations for future restoration

- **Build on established restoration nuclei**
Continue planting around surviving patches to promote gradual expansion and meadow coalescence over time.
- **Adopt a connectivity-led restoration model**
Create multiple small, resilient patches that serve as stepping stones between existing meadows, strengthening ecological connectivity and resilience.
- **Prioritise resilient species and facilitative approaches**
Expand the use of *Zostera noltii* in suitable intertidal zones and explore its role in facilitating longer-term recovery of *Z. marina*.
- **Focus on environmental suitability and adaptive site selection**
Use lessons from site performance to prioritise areas with stable sediments, favourable light conditions, and lower exposure to disturbance.
- **Maintain sustained monitoring and stakeholder collaboration**
Continued scientific monitoring, combined with collaborative action to improve water quality and reduce local pressures, will be critical to ensuring that early restoration gains translate into long-term meadow recovery.

5.3. Final synthesis

The restoration activities undertaken through the programme demonstrate clear and tangible progress towards seagrass recovery in the Firth of Forth. While expansion to large continuous meadows will require sustained effort and improving environmental conditions, the successful establishment of resilient patches, the strong performance of intertidal restoration, and the development of refined methodologies collectively represent a significant and optimistic step forward. These outcomes provide both proof of concept and a practical roadmap for scaling up restoration and achieving lasting seagrass recovery across the region.



Muddy habitats in the Firth of Forth make restoration challenging.

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5. Appendices and supplementary materials

5.1. Maps of plot locations at Restoration Forth project sites



Figure S1. Location of all *Zostera marina* planting at the Drum Sands project site.

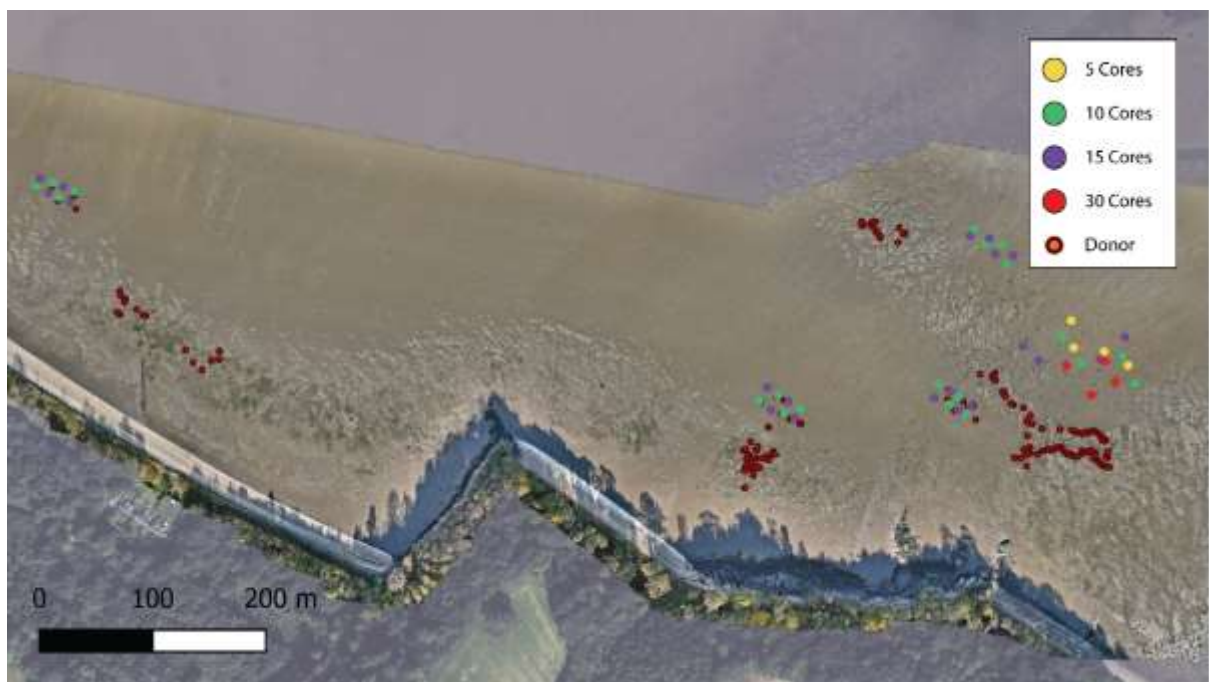


Figure S2. Location of all *Zostera noltii* planting at the Drum Sands project site.



Figure S3. Location of all *Zostera marina* planting at the Burntisland project site.



Figure S4. Location of all *Zostera marina* planting at the Tynninghame project site.

5.2. Selection of Random Forest model parameters at Drum Sands and Burntisland

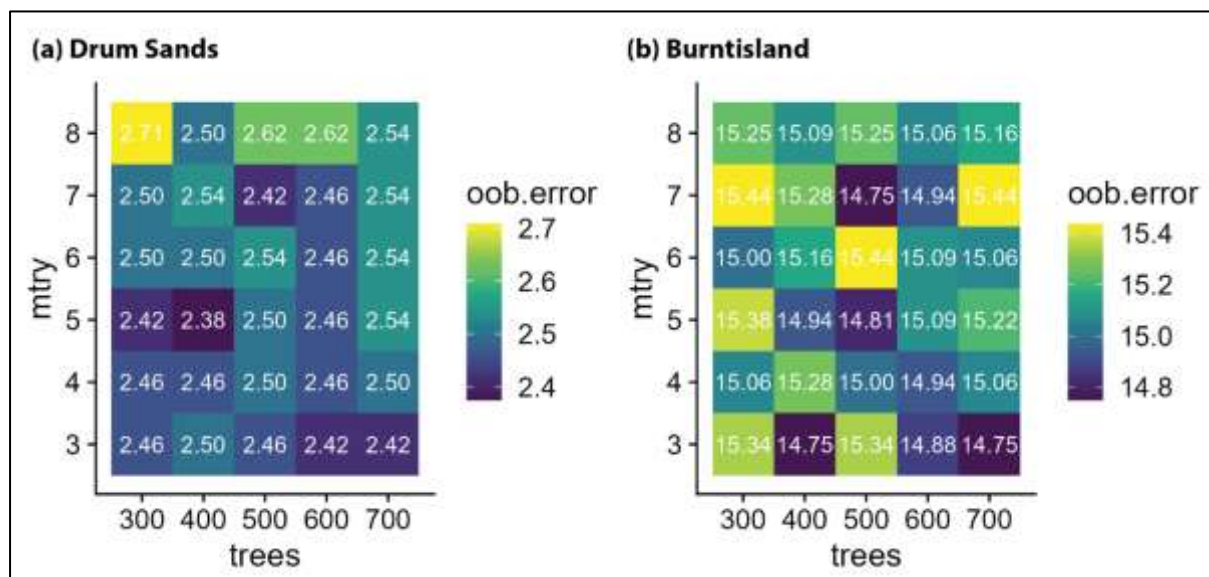


Figure S5. Iterative testing of Random Forest model parameters (m-try, ntrees) for **(a)** Drum Sands, and **(b)** Burntisland. Numbers show the out-of-bag error associated with each parameter combination.

5.3. Issues with model predictions at Drum Sands

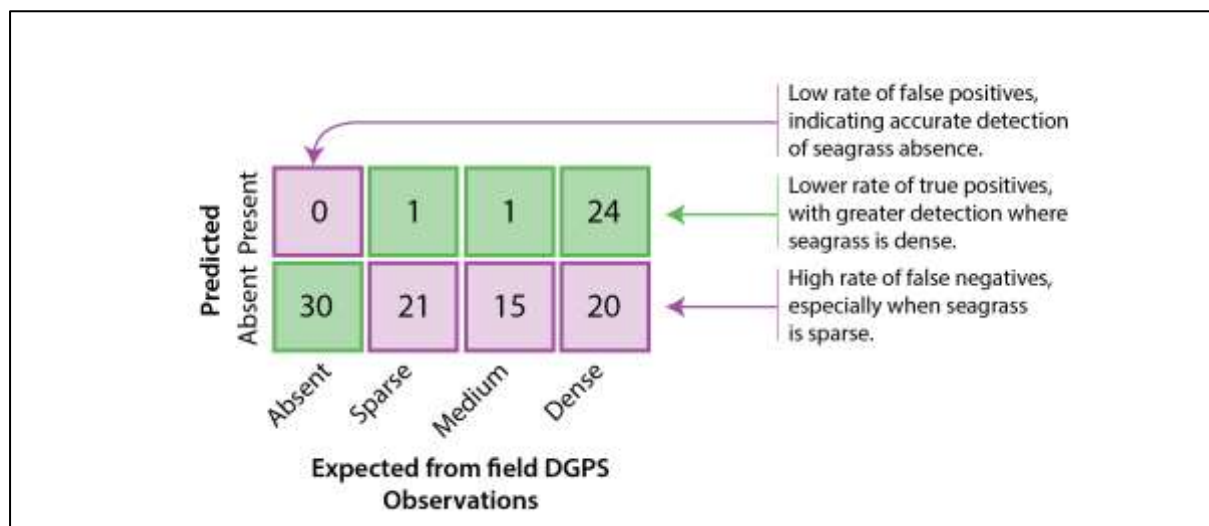


Figure S6. Comparison of Drum Sands model predictions with observations using DGPS in field indicates issues with model accuracy, or with orthorectification of drone imagery. If orthorectification of imagery is correct, then the accuracy of predicted seagrass extent at Drum Sands is lower than anticipated. In particular, the model struggles to predict *Zostera noltii* that is not dense. Green represents predictions that are correct, purple represents predictions that are incorrect.