MARGHERITA SCALA

RESTORATION OF SEAGRASS MEADOWS IN RIA FORMOSA: OVERCOMING AND ASSESSING HUMAN IMPACT

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Declaração de autoria de trabalho

Declaro ser a autora deste trabalho, que é original e inédito. Autores e trabalhos consultados estão devidamente citados no texto e constam da listagem de referências incluída.

Margherita Scala

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Resumo

As pradarias marinhas constituem ecossistemas costeiros de elevado valor ecológico, desempenhando funções cruciais como a fixação de carbono, a estabilização de sedimentos, a filtragem de nutrientes e a provisão de habitat para diversas espécies marinhas. Entre as espécies mais dominantes no intertidal europeu está a *Zostera noltei*, cujas populações têm vindo a regredir substancialmente nas últimas décadas, sobretudo em consequência da intensificação das pressões antropogénicas. A Ria Formosa, um sistema lagunar costeiro no sul de Portugal, constitui um exemplo paradigmático dessa realidade, sofrendo os efeitos cumulativos de atividades como a navegação recreativa, o uso de âncoras e boias de amarração, a construção civil e a apanha de bivalves.

O presente estudo teve como principal objetivo avaliar a eficácia da transplantação de *Zostera noltei* como estratégia de restauro ecológico num trecho degradado de pradaria, situado na zona intertidal da Ria Formosa, fortemente impactado pela construção de uma nova ponte. Ao longo de sete meses, foram monitorizados, em simultâneo, os locais recetores (transplantados) e os locais dadores (de onde foram retiradas as unidades transplantadas), com base na recolha de dados de campo e em imagens captadas por drone. Foram analisados quatro parâmetros principais: percentagem de cobertura, área vegetada, densidade de tufos, altura da folha e taxa de crescimento.

Os resultados preliminares, com base nas primeiras monitorizações (M1 a M9), indicavam uma tendência positiva de aumento na cobertura de *Z. noltei* nas unidades transplantadas. No entanto, com a inclusão dos dados finais, esta tendência deixou de ser estatisticamente significativa (p = 0.1142), revelando apenas um aumento médio de 5.6%. Paralelamente, observaram-se declínios consistentes na densidade de tufos e na altura das folhas, também sem diferenças significativas ($p > 0.99$). Esta informação sugere que, apesar de alguma recuperação localizada, o sucesso do restauro foi limitado à escala do prado como um todo.

Os dados obtidos por drone confirmaram esta conclusão, evidenciando um decréscimo constante da cobertura vegetal ao longo do período de estudo, desde 18.5% no início até cerca de 8% no final. Esta discrepância entre as observações de pequena escala (quadrículas) e os dados de larga escala (UAV) realça a importância de integrar diferentes níveis de monitorização em projetos de restauro ecológico. A análise da variância da taxa de crescimento diário entre unidades transplantadas revelou períodos de crescimento sincronizado (Janeiro–Março), alternando com momentos de forte variabilidade, sobretudo em Outubro e Novembro. Os testes ANOVA não mostraram efeitos significativos da identidade do patch ou do plot (p >

0.9), sugerindo que as diferenças não são explicadas pela distribuição espacial das unidades mas antes por fatores ambientais e antrópicos externos.

No caso dos locais dadores, os resultados foram substancialmente mais positivos. A análise do crescimento ao longo do tempo revelou um aumento estatisticamente significativo da cobertura de *Z. noltei* entre MD1 e MD7 ($p = 0.0001$), com uma diferença média de 14%. A comparação entre técnicas de colheita (com ou sem reposição de sedimento) não demonstrou diferenças significativas na taxa de crescimento diário ($p = 0.516$), o que sugere que ambas as metodologias são válidas, não comprometendo a integridade do prado dador. A baixa variabilidade nas taxas de crescimento ao longo do tempo também confirma a resiliência destas zonas, que não foram expostas a perturbações fixas, como boias ou estruturas de ancoragem.

A componente do estudo relativa às boias de amarração revelou impactos negativos claros. As áreas com boias (Buoy sites) apresentaram valores consistentemente inferiores de cobertura, densidade e altura das plantas, quando comparadas com áreas de controlo (NoBuoy sites). Apesar de não ter sido observada uma correlação significativa entre a distância à âncora e os parâmetros estruturais das plantas, o efeito negativo foi evidente nas zonas mais próximas da amarração, confirmando observações de estudos anteriores (e.g. Paling et al., 2003; Unsworth et al., 2017). Estas estruturas mecânicas, ao gerarem distúrbios contínuos e localizados, limitam fortemente a capacidade de regeneração das pradarias afetadas.

A análise global dos resultados levou à consideração do conceito de limiar ecológico. Mesmo após a remoção de um fator de stress significativo (neste caso a plataforma de construção), a persistência de pequenas perturbações, como a atividade náutica e recreativa, ou a pesca artesanal, poderá ter impedido a recuperação total do ecossistema. Quando um sistema ultrapassa um certo ponto crítico, pode entrar num estado degradado que se mantém estável e resistente à recuperação, mesmo que os fatores iniciais de perturbação sejam eliminados. No caso em estudo, é possível que a combinação de vários fatores de stress de baixa intensidade tenha mantido o sistema num estado subótimo, abaixo do limiar necessário para se regenerar de forma natural.

A comparação entre os locais transplantados e os locais doadores reforça esta ideia. As pradarias doadoras, não expostas a distúrbios permanentes, conseguiram recuperar de forma rápida e eficaz após a extração. Já as pradarias transplantadas, embora tenham recebido um esforço ativo de restauro, permaneceram vulneráveis, sem sinais de estabilização ecológica a longo prazo.

Este trabalho evidencia a complexidade dos processos de restauro ecológico em ambientes costeiros altamente impactados. Mostra que não basta remover as grandes fontes de

perturbação; é fundamental identificar e controlar também os fatores crónicos e difusos que, acumulando-se ao longo do tempo, impedem que os ecossistemas ultrapassem o limiar necessário à sua recuperação. O sucesso de projetos futuros dependerá da capacidade de integrar múltiplas escalas de monitorização, de escolher cuidadosamente os locais para transplante e de adotar uma abordagem que considere o conjunto de pressões existentes.

Palavras-chave: *Zostera noltei*, pressões antropogénicas, impacto das bóias, resiliência ecológica

Abstract

Seagrass meadows are vital coastal ecosystems providing numerous ecological services, yet they are increasingly threatened by anthropogenic pressures. This study investigates the effectiveness of a transplantation-based restoration strategy for *Zostera noltei* in the intertidal SE1 meadow of Ria Formosa (Portugal), an area recently impacted by construction activities, mooring infrastructure, and recreational boating. Over a seven-month period, both donor and receiver sites were monitored using field surveys and drone-based imagery to assess seagrass percentage cover, shoot density, shoot height, and daily growth rates. Initial quadrat-level data indicated promising trends in percent cover, however, inclusion of the last three monitorings revealed that no statistically significant increase had occurred ($p = 0.1142$). Both shoot density and height declined, suggesting limited restoration success at the meadow scale. This was corroborated by UAV-derived monitoring, which documented a decline in total cover from 18.5% to approximately 8%, underscoring the importance of integrating multiscale assessments. Variance in daily growth rates among receiver units revealed both synchronized and heterogeneous growth periods, with no significant influence of plot or patch identity (ANOVA, $p > 0.9$). In contrast, donor sites exhibited consistent recovery with a significant increase in percent cover ($p = 0.0001$) and no impact from sediment refill treatments. Mooring assessments revealed significantly reduced seagrass structure in buoyed areas compared to controls, with no correlation between distance from anchor and vegetation parameters. These findings suggest that chronic, small-scale disturbances may suppress recovery below ecological thresholds, even after removal of major stressors. This study highlights the importance of comprehensive disturbance management and multiscale monitoring to improve restoration outcomes and long-term resilience of seagrass ecosystems.

Keywords: *Zostera noltei*, anthropogenic disturbance, mooring impact, ecological resilience

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1. Introduction

1.1. Theme justification

Seagrasses are essential habitats in marine ecosystems, providing crucial services that support biodiversity and ecosystem stability. These rooted, flowering plants thrive in marine environments, forming extensive underwater meadows that play a fundamental role in maintaining the health of coastal ecosystems (Borum et al., 2004). Seagrass meadows are particularly significant in carbon sequestration, acting as long-term carbon sinks that mitigate climate change by storing carbon in their biomass and sediments (Fourqurean et al., 2012). Additionally, seagrasses contribute to marine food webs by exporting organic material, such as detritus (Heck et al., 2008), and enhance water quality by trapping sediments, filtering pollutants, and recycling nutrients (Duarte et al., 2017; Orth et al., 2006a). Their dense root and shoot systems help to reduce wave energy and water currents, minimizing coastal erosion and acting as a natural barrier to protect shorelines (Duarte et al., 2017).

Despite their importance, seagrass meadows have been extensively degraded globally due to direct and indirect human pressures (Raven, 2006; Short et al., 2011; West & Glasby, 2022). Indirect pressures, such as reductions in salinity and increases in turbidity and nutrient levels, often stem from land-based disturbances like deforestation, agriculture, mining, industrial development, urbanization, and pollution (Foster et al., 2017; Kilminster et al., 2015). These alterations in water quality can severely affect seagrass health by limiting light penetration and promoting harmful algal blooms. At the same time, direct physical impacts, including damage from boat moorings (Glasby & West, 2018; Hastings et al., 1995; Walker et al., 1989), anchoring (La Manna et al., 2015; Milazzo et al., 2004), propeller wash (Bell et al., 2002), dredging (Larkum & West, 1990), and activities like harvesting and trampling (Butler & Jernakoff, 1999), can result in the complete removal of seagrasses. These pressures often interact, complicating the identification of specific drivers behind the degradation of seagrass meadows (Fonseca et al., 2013). Additionally, climate change exacerbates these issues by increasing sea levels, storm intensity, and ocean acidification, further stressing these ecosystems (Unsworth et al., 2019). The introduction and spread of invasive species such as *Caulerpa prolifera* and *Asparagopsis armata* add an additional layer of stress, outcompeting native species and destabilizing ecosystem function (Cunha et al., 2013).

The Ria Formosa coastal lagoon, located in southern Portugal, provides a striking example of these dynamics. Local intertidal seagrass meadows, dominated by *Zostera noltei*, have been progressively degraded due to marina expansion, mooring pressure, and commercial clam harvesting (Cunha et al., 2009; Guimarães et al., 2012). In particular, the construction of a new bridge in the Faro Channel has intensified physical disturbance through piling, cables, and anchor deployment. Since the last pre-restoration survey in September 2023, a 66% reduction in cover has been recorded, corresponding to a loss of 383.81 m² within three months (CCMAR-CTS, 2024).

Given the extensive degradation of these seagrass meadows, restoration efforts are urgently needed to reverse the damage and promote the recovery of these vital ecosystems. This study tested the hypothesis that transplanting *Zostera noltei* into the intertidal SE1 meadow, heavily impacted by bridge construction, could promote vegetative recovery and restore key ecosystem functions. In addition, we assessed the influence of anchoring buoys on meadow health, evaluated recovery trajectories of donor areas subjected to different post-extraction techniques, and examined whether persistent anthropogenic stressors might hinder the restoration potential of the system.

1.2. The Seagrass Ecosystem

Seagrasses, the only angiosperms adapted to marine environments, evolved from terrestrial plants around 100 million years ago (Olsen et al., 2016). As rooted, flowering plants that form extensive underwater meadows, they play a fundamental role in maintaining the health of coastal ecosystems (Borum et al., 2004). Despite their vital yet fragile nature, they thrive in shallow embayments and estuaries across the globe, with Antarctica being the only exception (Cullen-Unsworth & Unsworth, 2016; Orth et al., 2006b). Recognized as bioindicators of water clarity and eutrophication, seagrasses are included in the European Water Framework Directive (WFD: 2000/60/EC) and the Marine Strategy Framework Directive (MSFD: 2008/56/EC) (Krause-Jensen et al., 2005).

Currently, over 70 seagrass species have been described, classified into four families and 13 genera. Their estimated global coverage ranges from 177,000 to 600,000 km² (Waycott et al., 2009), with the most recent high-confidence estimate being 160,000 km² (McKenzie et al., 2020). However, based on light availability and seagrass light requirements, their potential distribution could extend up to 4.3 million km², approximately 26 times larger (Gattuso et al., 2006). Fully submerged, seagrasses complete their entire life cycle in seawater, occupying habitats from the mid-intertidal zone to depths exceeding 50 meters, depending on water clarity and light penetration (Duarte, 1991).

Seagrasses exhibit a suite of adaptations that enable them to flourish in dynamic coastal environments. Their thin, ribbon-like leaves maximize light absorption, buoyancy, and gas exchange, while high chlorophyll concentrations within epidermal cells enhance survival in turbid waters, a trait shared with many shade-tolerant terrestrial species (Dennison, 1987). The root–rhizome system anchors plants in often unstable sediments while simultaneously facilitating nutrient uptake. Additionally, a network of air spaces, or lacunae, enables efficient oxygen transport from leaves to belowground structures, mitigating the metabolic costs of anaerobic respiration (Armstrong, 1980; Hemminga & Duarte, 2000).

Under optimal conditions, seagrasses exhibit high productivity, with biomass accumulation rates comparable to those of agriculturally significant crops (Duarte & Chiscano, 1999). Seagrasses act as an ecosystem engineer that alters its surrounding environment by affecting hydrodynamic energy, modifying sediment properties and providing a three-dimensional structure that constitutes shelter, nursing ground, feeding ground, spawning ground, substrate and food for herbivores, detritivores, and microorganisms (Gilby et al., 2018; Hemminga & Duarte, 2000; Polte & Asmus, 2006). Aboveground tissues also function as a substrate for epiphytic organisms, enhancing overall ecosystem productivity by as much as 35% (Brush & Nixon, 2002). Seagrass meadows rank among the most productive ecosystems on Earth (Barbier et al., 2011; Costanza et al., 1997), delivering a broad range of services valued at approximately US$3.8 trillion annually (Costanza et al., 1997). They are particularly significant in carbon sequestration, functioning as long-term carbon sinks that store carbon in both their biomass and underlying sediments (Fourqurean et al., 2012). By exporting organic material such as detritus, these meadows support broader marine food webs (Heck et al., 2008), and they enhance water quality by trapping sediments, filtering pollutants, and recycling nutrients (Duarte et al., 2017; Orth et al., 2006b). Their role as nurseries for commercially important fish and shellfish underscores both their economic and ecological importance (Beck et al., 2001; Nordlund et al., 2016).

Beyond their importance in carbon fluxes, seagrasses have a mechanical influence on local hydrodynamics. By increasing bottom roughness and elevating the benthic boundary layer, their canopies generate drag on waves and currents (Fonseca et al., 1982; Fonseca & Cahalan, 1992). This process dissipates energy, thereby attenuating wave action and current strength (De Boer, 2007; Fonseca et al., 1982; Infantes et al., 2012), which can contribute to coastal protection (Ondiviela et al., 2014; Paul, 2018). The dense root and shoot systems further minimize coastal erosion by stabilizing sediments and reducing resuspension (Duarte et al., 2017; Potouroglou et al., 2017). Taken together, these functions enable seagrass meadows to act as a natural barrier that safeguards shorelines from degradation (De Boer, 2007; Hansen & Reidenbach, 2012; Infantes et al., 2022).

1.3. Threats

Historical episodes highlight the vulnerability of seagrass ecosystems. Notably, the "wasting disease" of the 1930s resulted in the loss of approximately 90% of eelgrass (*Zostera marina* L.) populations across Europe and North America (Short et al., 1988). This catastrophic event triggered geomorphological and biological shifts, including substrate destabilization, beach erosion, and alterations in macroinvertebrate assemblages (Den Hartog, 1987; Rasmussen, 1977). Fauna dependent on seagrasses, such as waterfowl (*Branta bernicla*) and shellfish (*Argopecten irradians*), also suffered significant population declines (Thayer et al., 1975). These historical events underscore the fragility of seagrass habitats and foreshadow the challenges they continue to face.

Despite their ecological importance, and value, seagrass meadows have been extensively degraded worldwide due to both direct and indirect human pressures (Raven, 2006; Short et al., 2011; West & Glasby, 2022). Over the last century, about 29% of the global seagrass area has been lost (Orth et al., 2006b; Waycott et al., 2009), and although the rate of decline has slowed in certain regions, it remains ongoing in many areas (de los Santos et al., 2019; Dunic et al., 2021). Anthropogenic factors, such as eutrophication, coastal development, and the overfishing of top predators, are primary contributors (Orth et al., 2006b). Indirect pressures, including reductions in salinity and increases in turbidity and nutrient concentrations, often arise from land-based disturbances like deforestation, agriculture, mining, industrial development, urbanization, and pollution (Foster et al., 2017; Kilminster et al., 2015). These water-quality alterations can severely impair seagrass health by restricting light penetration and fostering harmful algal blooms. While seagrasses can assimilate nutrients under moderate loads, excessive agricultural runoff and wastewater inflows further amplify algal blooms, epiphyte overgrowth, and macroalgae proliferation (Burkholder et al., 2007; Dennison et al., 1993; McGlathery, 2001), thereby reducing light availability and exacerbating meadow decline.

Eutrophication is among the most severe of these stressors (Airoldi & Beck, 2007; Worm & Lotze, 2006) driven by excessive nitrogen and phosphorus inputs from agricultural fertilizers and wastewater (Nixon, 1995). Elevated nutrient levels promote the overgrowth of phytoplankton, epiphytic algae, and macroalgae (Hauxwell et al., 2001; Liu et al., 2009), which compete with seagrasses for light and nutrients, reduce oxygen availability, and ultimately smother seagrass leaves (Cabaço et al., 2005). Climate change, including raising water temperatures, increasing the frequency of extreme climate events (De Fouw et al., 2016; Fraser et al., 2014; Short et al., 2016), and ocean acidification (Unsworth et al., 2019), further aggravates these issues by imposing additional stress on already compromised habitats. The spread of invasive species such as Caulerpa prolifera and Asparagopsis armata compounds these challenges, displacing native species and disrupting ecological balances (Cunha et al., 2013).

Physical alterations to coastal environments worsen the situation. Structures like harbors, docks, breakwaters, and beach stabilization projects disrupt sediment dynamics (Cardoso et al., 2004), often leading to local burial events during storms or runoff episodes that severely damage seagrass meadows (Cabaço et al., 2007). Dredging and sedimentation from altered land catchments also destabilize the seabed, smothering root–rhizome systems (Pasqualini et al., 1999). Meanwhile, direct physical impacts, including damage from boat moorings (Glasby & West, 2018; Hastings et al., 1995; Walker et al., 1989), anchoring (La Manna et al., 2015; Milazzo et al., 2004), propeller wash (Bell et al., 2002), dredging (Larkum & West, 1990), and harvesting or trampling (Butler & Jernakoff, 1999), can result in the complete removal of seagrasses. These activities fragment meadows and uproot entire patches (Bell et al., 2002; Uhrin & Holmquist, 2003), making it difficult to isolate specific drivers of degradation due to their interactive and cumulative effects (Fonseca et al., 2013).

Such fragmentation is well documented in Posidonia meadows along the Corsican coast, where trawling, anchoring, and even bomb blasts have produced a patchy landscape marked by unvegetated channels and isolated sand patches within vegetated areas (Pasqualini et al., 1999; Sánchez-Jerez et al., 1999). Collectively, these threats compromise the ecological integrity of seagrass meadows (Orth et al., 2006b) and diminish the vital ecosystem services they provide (Costanza et al., 1997; Waycott et al., 2009). Understanding the mechanisms driving these impacts and their spatial extent is essential for designing targeted conservation strategies and mitigating further losses (McGlathery et al., 2012; Unsworth et al., 2019).

1.4. Restoration effort

Seagrass restoration efforts began in the 1940s, in response to the “wasting desease”, pioneered by Addy (1947). After substantial additional losses during the latter half of the twentieth century, mostly due to eutrophication and overfishing, the field rapidly expanded through methodological advancements and large-scale projects (Fonseca, 2011; Fonseca et al., 1998; Lewis, 1987; Phillips, 1960). Over the past two decades, further research and larger restoration trials have been undertaken (e.g., Leschen et al., 2010; McGlathery et al., 2012; Orth et al., 2020; Tan et al., 2020; Ward & Beheshti, 2023), resulting in guidelines tailored to specific regions (e.g., Gamble et al., 2021; Moksnes et al., 2021; van Katwijk et al., 2009). Seagrass restoration is now highlighted as a critical component in rebuilding marine life during this century (Duarte et al., 2020) and has been proclaimed essential to achieve ambitious global recovery targets (Buelow et al., 2022). In the European Union, seagrasses and their restoration are explicitly mentioned in the proposed Nature Restoration Law, which, if passed, would require Member States to restore 30% of degraded seagrass areas by 2030.

Despite recognition of its importance, seagrass restoration (and coastal restoration in general) remains in its infancy. Success rates are often low, around 37%, and costs can be prohibitively high (Bayraktarov et al., 2016; van Katwijk et al., 2016). Still, seagrass has been planted in high-energy environments, such as Australia (Wear et al., 2010), Portugal (Paulo et al., 2019), the United Kingdom (Unsworth et al., 2019), and Tanzania (Wegoro et al., 2022), illustrating the feasibility of different methodologies and underscoring the context-dependence of restoration outcomes (van Katwijk et al., 2016). Consequently, experiences gained from one project or location may not always be applicable elsewhere (Paling et al., 2003; van Katwijk & Hermus, 2000).

Nevertheless, several general factors for successful restoration have emerged. First, it is critical to confirm that seagrass historically occurred at a site (i.e., the habitat is suitable) and to address the cause of its disappearance (e.g., poor water quality) before starting any restoration. Additional key considerations include site selection, optimal transplantation timing, planting techniques, and nutrient availability (Pansini et al., 2022; van Katwijk et al., 2009). While the most effective approach varies by location and is ideally tested at a small scale first, a few general principles have been identified. For instance, larger restoration plots can help trigger self-reinforcing mechanisms and/or spread risks across multiple areas (van der Heide et al., 2007; van Katwijk et al., 2016), although contrasting findings exist (Matheson et al., 2023; Mourato et al., 2023). Incorporating known biotic and abiotic interactions can also enhance outcomes, such as pairing seagrass with bivalves (Gagnon et al., 2021; Meysick et al., 2020) or employing artificial structures to stabilize sediments (Temmink et al., 2020).

1.5. Restoration methods

There is no single "one-solution-fits-all" approach for seagrass restoration. A technique that proves effective in one location may fail in another, emphasizing the importance of context when selecting restoration methods, sites, and assessment protocols (Tan et al., 2020; van der

Heide et al., 2021; van Katwijk et al., 2016). Numerous factors—including target species, restoration goals, and prevailing anthropogenic stressors—need to be carefully evaluated before initiating a project. Broadly, restoration methods can be divided into two main categories: transplanting adult shoots and seed-based restoration.

Transplanting adult shoots is a widely used technique, though success varies (Bastyan & Cambridge, 2008; Statton et al., 2012; van Katwijk et al., 2016). Within this category, two primary approaches exist: sediment-free and sediment-based methods. In sediment-free transplantation, seagrass rhizomes (typically close to the sediment surface) are excavated, and excess sediment is shaken off before transport (Davis & Short, 1997). Minimally invasive tools are recommended to reduce damage to the donor bed, and a minimum of one apical rhizome meristem per planting unit (PU) is advisable to promote successful establishment (Fonseca et al., 1998). Plants are then placed directly into the substrate (sprig method) or anchored using various devices, such as U-shaped metal staples, bamboo skewers, rods, or boulders (Costa et al., 2022; Curiel et al., 2003; Wegoro et al., 2022).

Sediment-based methods involve leaving the rhizomes and associated substrate largely intact. Sod transplantation, for example, uses intact sections of seagrass, making it particularly effective for hard, compact substrates or large seagrass species with deeper roots (Costa et al., 2022; Mourato et al., 2023). This technique typically requires only shovels and large containers for sods, minimizing disturbance to the donor bed. Plug transplantation uses devices such as pipes or sod pluggers to extract smaller units of seagrass plants and their sediment (Wegoro et al., 2022), then replanting these plugs into similarly soft, cohesive sediments. This approach can be impractical for larger species, as the risk of leaf shearing increases (Calumpong & Fonseca, 2001).

Seed-based restoration, on the other hand, holds promise for achieving large-scale outcomes, especially in low-energy environments where seeds can settle and germinate with minimal predation pressure (Mourato et al., 2023). Recent studies illustrate the potential of this approach. Gräfnings et al. (2024), for instance, documented the successful recruitment of *Zostera noltei* from seeds, with 184 individual patches emerging from 7,000 seeds over three months. This corresponds to a recruitment rate of 2.6%, which improved in absolute terms at higher seed densities, although the efficiency of seed use declined as more seeds were added per injection.

Once restoration is underway, regular monitoring is crucial to assess short- and long-term success. In the first year following transplantation, at least three monitoring events are recommended, followed by biannual assessments in subsequent years (Calumpong & Fonseca, 2001). These evaluations focus on three main categories. First, seagrass parameters include survival of planting units, areal coverage, and shoot density (Calumpong & Fonseca, 2001). Second, disturbance parameters capture the impact of animals, currents, and wave activity (Calumpong & Fonseca, 2001). Third, water quality indicators measure total suspended solids, sedimentation rate, temperature, salinity, and sometimes nutrient concentrations (Dennison et al., 1993). Changes in absolute area, percentage cover, vegetated area (calculated by multiplying total area by percentage cover), shoot density, and leaf length provide a comprehensive picture of meadow health and expansion (Curiel et al., 2021; Gräfnings et al., 2024; Mourato et al., 2023; Paulo et al., 2019; Suykerbuyk et al., 2016).

1.6. Seagrasses in Ria Formosa

Ria Formosa is Portugal's largest coastal lagoon, situated along the southern coast of the Algarve region between the cities of Faro and Tavira. Covering approximately 18,400 hectares, this lagoon system includes extensive salt marshes, mudflats, channels, and barrier islands stretching over 60 km of coastline. Its dynamic environment—shaped by tidal movements, sediment deposition, and seasonal changes—drives the lagoon's evolving morphology and habitat distribution (Guimarães et al., 2012). The intertidal zone, which is exposed during low tide and submerged at high tide, supports seagrass meadows dominated by *Zostera noltei*. In contrast, the subtidal zone, remaining underwater even at low tide, is mainly colonized by *Zostera marina* and *Cymodocea nodosa* (Cunha et al., 2009). This unique geomorphology, marked by expansive sandbanks, tidal inlets, and shallow waters, creates optimal conditions for seagrass growth, making these meadows among the most productive ecosystems in the region (Cunha et al., 2013).

Renowned for its high biodiversity, Ria Formosa is a critical nursery ground for numerous fish and invertebrate species, several of which are economically important for local fisheries (Erzini et al., 2022). The seagrass meadows, in particular, offer essential habitat and shelter for juvenile fish, mollusks, and crustaceans, thereby supporting the lagoon's food web (Heck et al., 2008). Moreover, the lagoon is recognized as a wetland of international importance under the Ramsar Convention and serves as a pivotal stopover site for migratory birds traveling along the East Atlantic Flyway. Seagrasses in Ria Formosa contribute significantly to carbon sequestration and nutrient cycling (Fourqurean et al., 2012). Their dense root systems stabilize sediments, reducing erosion and improving water clarity, which in turn enhances the lagoon's overall water quality and productivity. By trapping sediments and nutrients, these meadows help prevent coastal eutrophication, thereby maintaining the ecological balance of the region (Duarte et al., 2017).

Despite its ecological significance, Ria Formosa is under considerable pressure from human activities that have led to the degradation of its natural habitats, especially the seagrass meadows. Practices such as dredging, marina expansion, and the proliferation of boat moorings have physically disturbed the lagoon's seabed, altering both hydrodynamics and sediment composition (Figueiredo da Silva et al., 2004). Increased sediment resuspension, coupled with reduced light penetration in turbid waters, has impaired seagrass growth and contributed to the loss of meadow coverage. Clam harvesting, another prevalent activity in the region, has also harmed seagrass beds; the mechanical methods used to collect clams disrupt the substrate and weaken these ecosystems (Cabaço et al., 2005). Additionally, infrastructure projects, such as constructing a new bridge in the Faro channel, have altered local sediment dynamics and water circulation patterns, affecting nearby seagrass meadows (CCMAR-CTS, 2024). These changes have been linked to significant losses of *Zostera noltei*, particularly in the intertidal zone. Notably, monitoring by the Centro de Ciências do Mar do Algarve (CCMAR) identified severe impacts on the intertidal meadow SE1, where approximately 66% of its total area (over 383 m²) was lost between September and December 2023 (CCMAR-CTS, 2024) (Table 1.1). This decline was primarily attributed to the stationary platform used during the bridge's construction, whose pillars, anchoring cables, and prolonged shading depressed local sediments and reduced light availability.

Table 1.1: Evolution of the SE1 seagrass meadow in the south-east area over time. The meadows have fragmented over time. Only the species *Z. noltei* is shown in this table, as it was the species with the most complex changes.

<table>
<tr><th></th><th>October 2021</th><th>September 2023</th><th>December 2023</th></tr>
<tr><td rowspan="2">South-east</td><td rowspan="2">SE1
(2960 m²)</td><td>SE1
(462,55 m²)</td><td>SE1
(158,74 m²)</td></tr>
<tr><td>SE2
(300 m²)</td><td>SE2
(237,98 m²)</td></tr>
</table>

1.7. Hypothesis and Objectives

The main hypothesis of this study is that transplanting *Zostera noltei* in the intertidal zone, in the area affected by the construction of the new bridge in the Faro channel, will facilitate the recovery of seagrass meadows and their associated ecological functions. The specific aims of this thesis are to:

- Assessing the effects of the existing anchoring buoys and boats on the health of seagrass meadows.
- Evaluate the best recovery techniques for donor seagrass meadows, monitoring these areas to ensure sustainable transplantation practices.
- Test the effectiveness of these strategies in reversing human-induced degradation, restoring the natural state of the seagrass meadows, and enhancing the resilience of the Ria Formosa ecosystem.

2. Material and Methods

2.1. Study site

The Ria Formosa lagoon stretches over 60 km along the southern Algarve coast between Faro and Tavira and covers an area of approximately 18,400 hectares. It is characterised by a mosaic of intertidal mudflats, salt marshes, tidal channels, and barrier islands, shaped by dynamic sedimentation and tidal processes (Guimarães et al., 2012). The intertidal zone is primarily colonised by *Zostera noltei*, while the subtidal region supports *Zostera marina* and *Cymodocea nodosa* meadows (Cunha et al., 2013). The lagoon's shallow morphology and high sediment turnover create suitable conditions for seagrass development but also expose the meadows to physical disturbance from human activities.

Transplantation was carried out in the intertidal zone of the SE1 meadow (Figure 2.1), adjacent to the remaining patches of *Z. noltei* affected by the bridge construction in the Faro Channel. The goal was to restore at least 100 m² of vegetated area using sods from a healthy donor meadow. The planting layout overlapped partially with surviving seagrass beds to maximise integration with the existing habitat. Site selection was constrained to intertidal elevations similar to the donor meadow, ensuring ecological compatibility and minimising transplant stress.

The SE1 site is located at approximately 37°02′52″N, 7°59′46″W and is characterised by shallow depths (<1 m at low tide), sandy-muddy substrate, and moderate tidal energy. The area remains exposed to frequent human disturbances, including clam harvesting, recreational anchoring, and daily boat traffic crossing under the nearby bridge. Despite the removal of the construction platform, several mooring buoys and small vessels persist in the area, creating continuous physical stress on the recovering meadow.

Following general restoration guidelines (Paling et al., 2003; van Katwijk et al., 2016), the donor site was selected based on environmental similarity, health status, and species composition. NW1 (Figure 2.1), located at approximately 37°04′00″N, 7°59′50″W, was identified as the optimal source area. This site lies within the same intertidal range as SE1 and exhibits similar sediment type and exposure conditions. Previous monitoring by CCMAR-CTS (2024) had shown this meadow to be persistent and resilient, with no signs of mechanical disturbance and consistent shoot density across years.

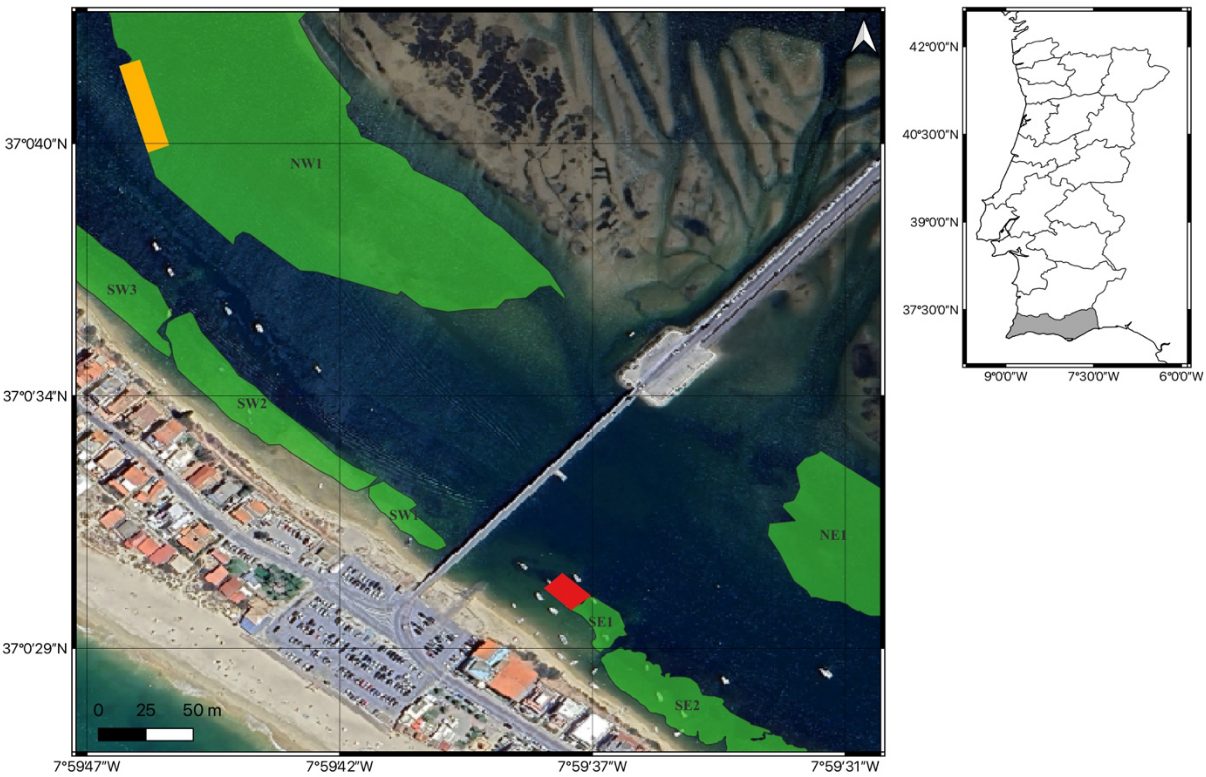

Figure 2.1: Study area in the Ria Formosa lagoon (southern Portugal), showing the location of seagrass meadows and experimental plots. Green polygons indicate the main natural seagrass meadows (NW1, NE1, SW1–SW3, SE1–SE2), while the red and yellow rectangles represent the donor and receiver plots, respectively, used for transplantation experiments. The receiver site (red) is located in a high-disturbance area near the bridge, exposed to heavy boat traffic and mooring infrastructure. The donor site (NW1, yellow) is located in a more protected area with limited anthropogenic impact. The inset map shows the regional location of the study area within southern Portugal.

2.2. Harvest Method

The harvesting technique used in this project involved the collection of intact sods, recognized as the most effective method for *Zostera noltei* transplantation (Mourato et al., 2023). This approach enhances the species' self-facilitation process while minimizing disturbance to the donor meadow (Costa et al., 2022). Sods, consisting of undisturbed roots, rhizomes, and leaves, were collected from the donor meadow NW1 in sections approximately 35 × 35 × 5 cm using a shovel and a linear sampling approach to minimize the impact on the healthy seagrass (Figure 2.2). The sods were immediately placed in buoyant plastic trays (1 m × 0.5 m) and quickly transferred to the transplant site by boat within 30 minutes to preserve optimal seagrass conditions.

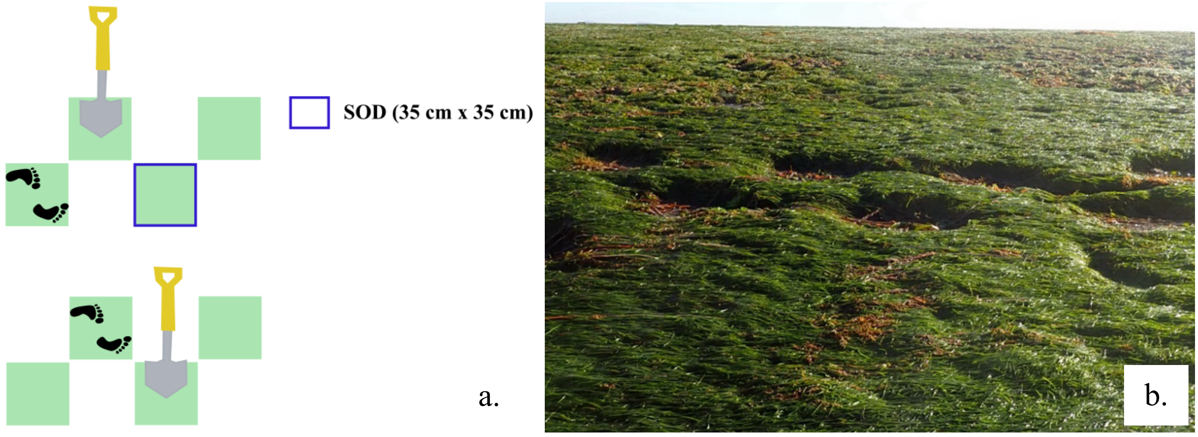

Figure 2.2: Sampling strategy at the donor site: a. A schematic representation of the sod extraction process for seagrass transplantation. b. The actual donor site after sampling.

2.3. Transplant Methodology

In the designated transplant area, ten 3 m × 3 m plots were established, arranged in a grid of three rows of three plots and one additional plot at the top, with a one-meter distance between each (Figure 2.3a). Each plot contained 25 seagrass sods, which were arranged in a chessboard pattern, alternating between seagrass and bare sediment to promote ingrowth in the open spaces (Costa et al., 2022) (Figure 2.3b). The transplant method involved excavating each spot to a depth of 5 cm using a shovel, allowing for proper placement of the sods. After positioning, sand was gently pushed over the rhizomes to cover them, ensuring the roots were not exposed (Mourato et al., 2023).

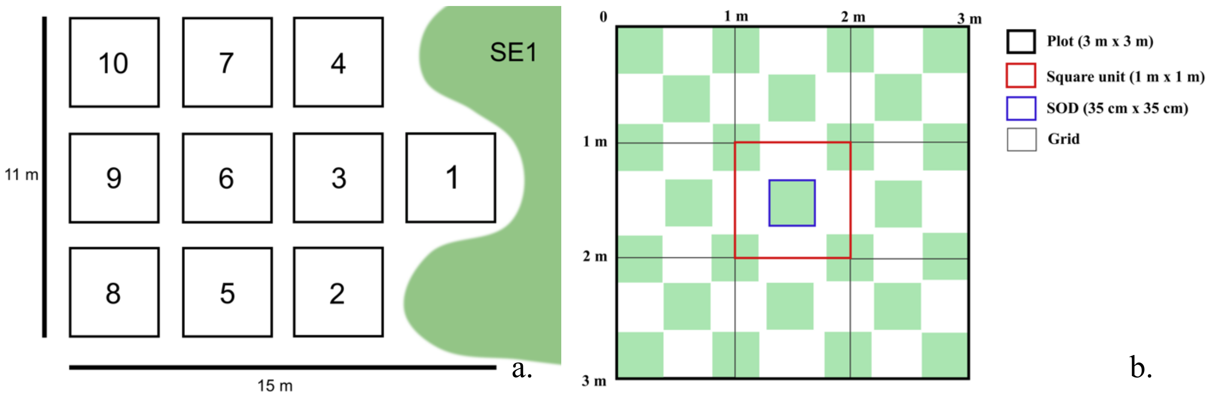

Figure 2.3: a. Transplant design in the area of Praia de Faro, Portugal. In green, the meadow SE1. In black, the 10 plots (9 m2 each) displayed in a grid pattern and separated by a meter between themselves. From east to west plots are numbered 1 to 10; b. Representation of the chessboard methodology. A grid was mounted at every meter (grey lines), creating nine square units (red outline) within each planting unit of 3 × 3 m (black outline). The sods (in green) were transplanted in a checkerboard pattern, separated by approximately 50 cm and with the corners in proximity.

2.4. Transplant Monitoring

Two complementary approaches were employed to monitor the performance of the transplantation effort. First, direct field-based assessments were conducted on twenty randomly selected planting units (sods) at the receiver site (Figure 2.4) and ten units at the donor site, supplemented by two undisturbed controls. These units were monitored at structured time intervals, beginning with an initial assessment one day post-transplantation, followed by weekly assessments during the first month and then monthly evaluations.

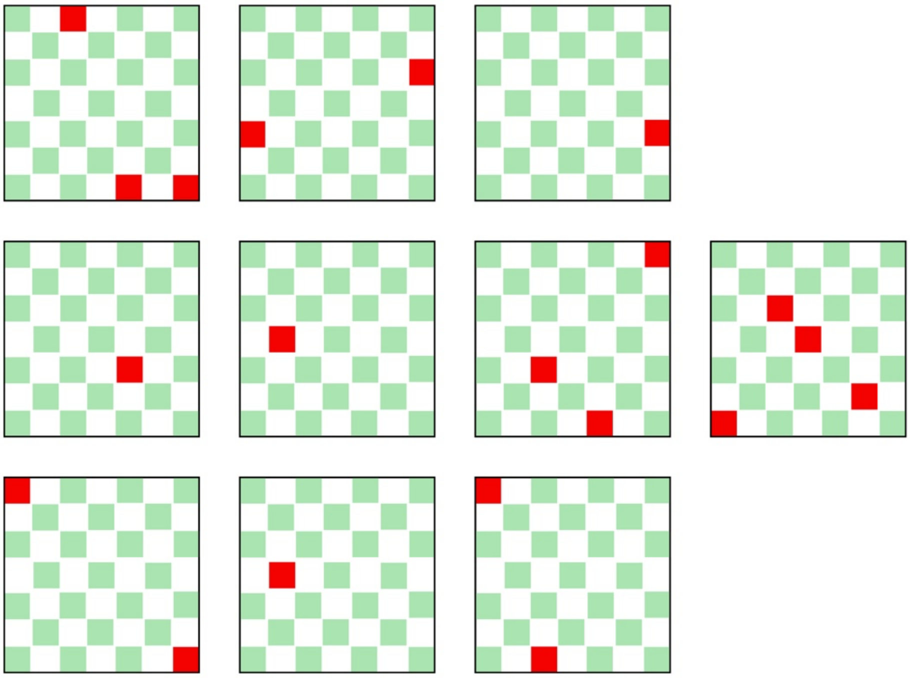

Figure 2.4: Spatial layout of the transplantation plots at the SE1 site. Each black-outlined square represents a 3 × 3 m plot composed of a grid of 50 × 50 cm planting units (sods), arranged in a chessboard configuration. Green squares indicate all transplanted units, while red squares represent the sods that were randomly selected for monitoring. A total of 20 planting units across 10 plots were monitored throughout the study to assess changes in percentage cover, shoot density, and shoot height over time.

Zostera noltei coverage was monitored at low tide using an Olympus TG7 camera and a 50 × 50 cm quadrat. The photographs were analysed with CPCe 4.1 software (Kohler & Gill, 2006), using 25 random points within each image to assess the presence of *Z. noltei*. For the transplanted patches also shoot density and canopy height was monitored. Shoot density, a key indicator of seagrass abundance (Short & Coles, 2001), was measured in the field by placing a 10 × 10 cm shoot-counting quadrat in the centre of each planting unit and counting all visible shoots. Canopy height was measured in situ using a metric ruler. The shoot length (cm) within the 50 cm x 50 cm quadrat was recorded and rounded to the nearest 0.5 cm, with a minimum of 10 shoots measured per quadrat to ensure accuracy.

In addition to these ground-based methods, drone-based aerial surveys were conducted to monitor the receiver sites. These surveys were carried out at quarterly intervals and served to assess overall patch development and horizontal expansion at a broader spatial scale. UAV-based monitoring allowed the detection of patch fragmentation and spatial dynamics that may not be captured through quadrat-based sampling. Drone assessments were limited to receiver areas, as the donor site was located within a protected airspace near an operational airport, which prohibited UAV flights.

2.5. Mooring and Boats Monitoring

Since one mooring (N 37° 0′ 29.64″, W 7° 59′ 36.90″) and an anchored boat (N 37° 0′ 29.52″, W 7° 59′ 37.08″) were located within the transplant area (Figure 2.5), additional monitoring were carried out around four moorings (N 37° 0′ 27.96″, W 7° 59′ 34.38″; N 37° 0′ 27.48″, W 7° 59′ 33.90″; N 37° 0′ 27.12″, W 7° 59′ 33.30″; N 37° 0′ 26.40″, W 7° 59′ 31.26″) in the healthy meadow SE2 at the same depth range (Figure . This allowed for the evaluation of the buoys' impact on both transplanted and healthy seagrass meadows.

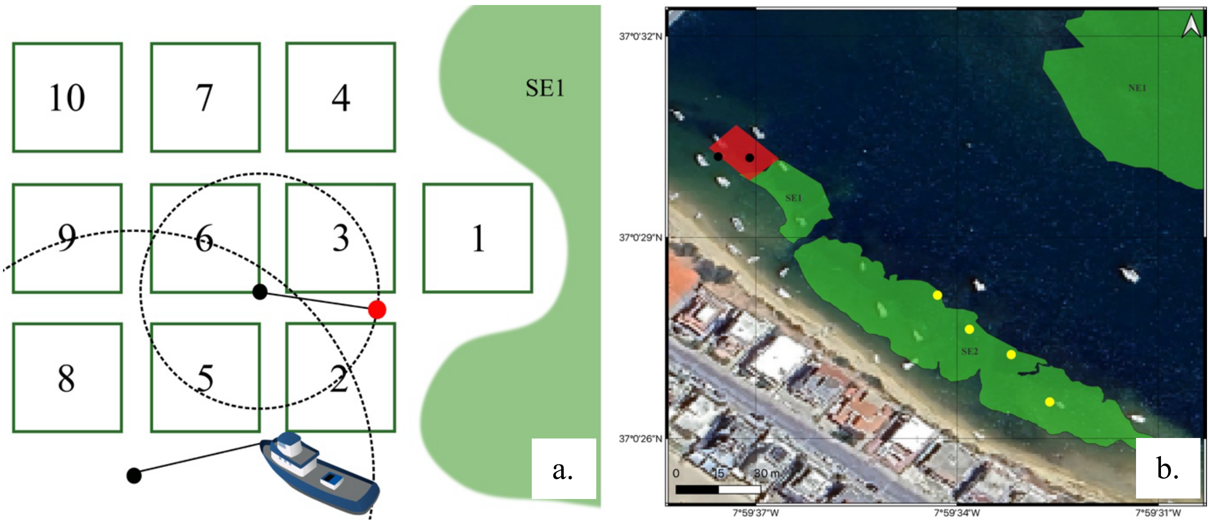

Figure 2.5: a. Transplant layout in the SE1 area of Praia de Faro (Portugal), illustrating the configuration of ten 3 × 3 m plots used for *Zostera noltei* transplantation. The red dot indicates the position of a buoy directly installed within the transplanted area, while the adjacent vessel is shown moored to the buoy system. Fixed anchoring blocks are marked in black, with dashed circles representing the estimated impact zone of the mooring cable's swing radius; b. Satellite map of the SE1 and SE2 meadows showing the location of the transplanted receiver site (in red), nearby intact meadows (green), and the positions of five monitored mooring systems (yellow dots) distributed along the southern edge of the Faro Channel.

2.6. Statistical analysis

A comprehensive set of statistical analyses was conducted using the R statistical software (R Core Team, 2023) to assess the performance and impact of seagrass restoration interventions. Paired-sample t-tests were performed to evaluate temporal changes in seagrass structural attributes between the initial and final monitoring events. These tests were applied at the level of individual planting units, where percentage cover, shoot density, and shoot height were directly measured in the field. Due to the limited number of observations available from drone-based monitoring, where only one value was recorded for each time point, it was not possible to apply standard inferential statistics such as paired t-tests or ANOVA, which require at least two observations per group to estimate within-group variability. Instead, a descriptive trend analysis was conducted to explore potential changes in percentage cover over time. A simple linear regression was applied to model the relationship between the monitoring date and the percentage of seagrass cover, providing an exploratory assessment of whether an overall increasing or decreasing trend could be observed during the study period. Although the statistical power of this model is limited by the small sample size, it offers a preliminary insight into the temporal dynamics of the restored area as captured by UAV-based data. The trend was visualized using a linear fit line and 95% confidence interval to aid interpretation. This dual approach allowed the detection of changes occurring within the micro-scale of individual sods and the broader-scale patterns across the entire transplant area. In addition, to explore variability in daily growth rate across space and time, a two-way ANOVA was performed using Plot and PatchID as fixed factors. Temporal variance in daily growth rate was also calculated to identify periods of synchronized or heterogeneous behaviour among patches.

In donor sites, the analysis focused on comparing the growth rate of donor units subjected to two treatments: Test 1, where sediment was used to refill the extraction holes, and Test 2, where holes were left open. A one-tailed Welch t-test was performed to test whether Test 1 resulted in higher growth than Test 2. Growth rate variance was also calculated across dates to detect periods of uniform donor response.

To assess the impact of mooring infrastructure on seagrass meadows, data were collected from four monitored sites with moorings (B1 to B4) and ten randomly selected control sites where no buoys were present. For each point, percentage cover, shoot density, and shoot height were measured across six monitoring events (M1 to M6), and the average value per parameter was calculated to represent a site-level metric. At buoyed sites, the distance from each sampling point to the mooring anchor (Length) was also recorded.

For each parameter, Pearson correlation tests were used to assess whether seagrass percentage cover, density, or shoot height increased with distance from the mooring anchor. To evaluate whether seagrass structure was significantly reduced in buoyed areas, Welch's two-sample t-tests were conducted under the one-tailed hypothesis that NoBuoy sites would exhibit higher values than Buoy sites. All analyses were applied independently to each ecological variable.

3. Results

3.1. Receiver Sites

The paired-sample t-test performed on seagrass percentage cover at the level of individual planting units between the initial (M1) and final (M12) monitoring events did not reveal a statistically significant increase ($p = 0.1733$), although the mean difference was +4.4%. Similarly, shoot density decreased significantly, with a mean difference of –9.7 shoots ($p = 0.9928$), while shoot height also declined with a mean difference of –4.01 cm ($p = 1$).

The two-way ANOVA performed on daily growth rates of percentage cover showed no statistically significant effects for Plot ($p = 0.975$), PatchID ($p = 0.962$), or the Plot × PatchID interaction. Individual ANOVAs at the plot level also indicated no significant differences among patches (all $p > 0.26$).

Temporal analysis of daily growth rate variance revealed heterogenity. The lowest variance values were observed on March 2025 (0.0072 $(\%/day)^2$), January 2025 (0.0836 $(\%/day)^2$), February 2025 (0.165 $(\%/day)^2$), December 2024 (0.262 $(\%/day)^2$), and June 2025 (0.269 $(\%/day)^2$). In contrast, the highest levels of growth variability were recorded on April 2025 (0.322 $(\%/day)^2$), May 2025 (0.388 $(\%/day)^2$), October 31, 2024 (1.89 $(\%/day)^2$), November 2024 (2.01 $(\%/day)^2$), November 2024 (3.32 $(\%/day)^2$), and October 21, 2024 (10.4 $(\%/day)^2$), (Figure 3.1).

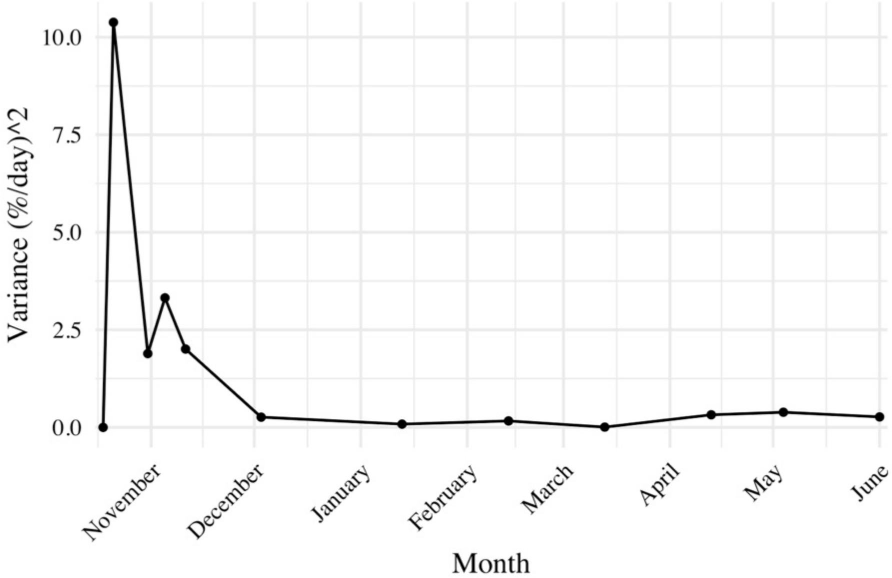

Figure 3.1: Temporal variation in daily growth rate variance among receiver patches. The graph displays the variance ($(\%/day)^2$) in seagrass daily growth rate across monitoring dates from October 2024 to June 2025.

Drone-based monitoring revealed a gradual decline in seagrass percentage cover over the course of the study period (Figure 3.2). Starting from approximately 18.5% in the initial posttransplant survey (November), percentage cover values steadily decreased across subsequent monitoring events, reaching a low of around 8% by April. Although statistical inference was not possible due to the limited number of observations, a simple linear regression suggested a negative trend in overall percentage cover, potentially indicating a reduction in transplant performance or meadow persistence at the receiver site over time.

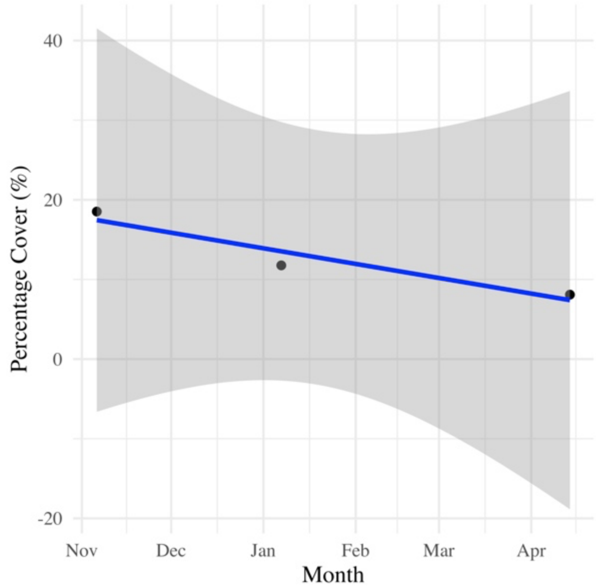

Figure 3.2: Temporal trend in seagrass percentage cover at the receiver site, as assessed through drone-based monitoring. Each point represents the percentage cover estimated for the transplant area during a given monitoring event. A simple linear regression (blue line) with 95% confidence interval indicates a declining trend over time.

3.2. Donor Sites

The paired-sample t-test comparing donor cover between MD1 and MD8 revealed a statistically significant increase ($p = 6.236e-05$), with a mean difference of +14.3% across all monitored donor patches.

The one-tailed Welch's t-test comparing daily growth rate between Test 1 (sediment refilled) and Test 2 (no refill) showed no statistically significant difference $p = 0.4232$), with mean rates of 0.0999 and 0.0851 %/day, respectively. The mean daily growth rate was 0.100 %/day for Test 1 and 0.085 %/day for Test 2. A one-way ANOVA also confirmed the non-significance of treatment effect ($p = 0.846$).

Variance in daily growth rate among donor units was lowest on 2024-10-28 (0.000 $(\%/day)^2$), 2025-03-12 (0.0025 $(\%/day)^2$), and 2025-01-17 (0.00712 $(\%/day)^2$), while higher variance values were recorded on 2024-12-16 (0.248 $(\%/day)^2$), 2024-11-25 (0.0417 $(\%/day)^2$), 2025-05-04 (0.0396 $(\%/day)^2$), and 2025-06-01 (0.0156 $(\%/day)^2$) (Figure 3.3).

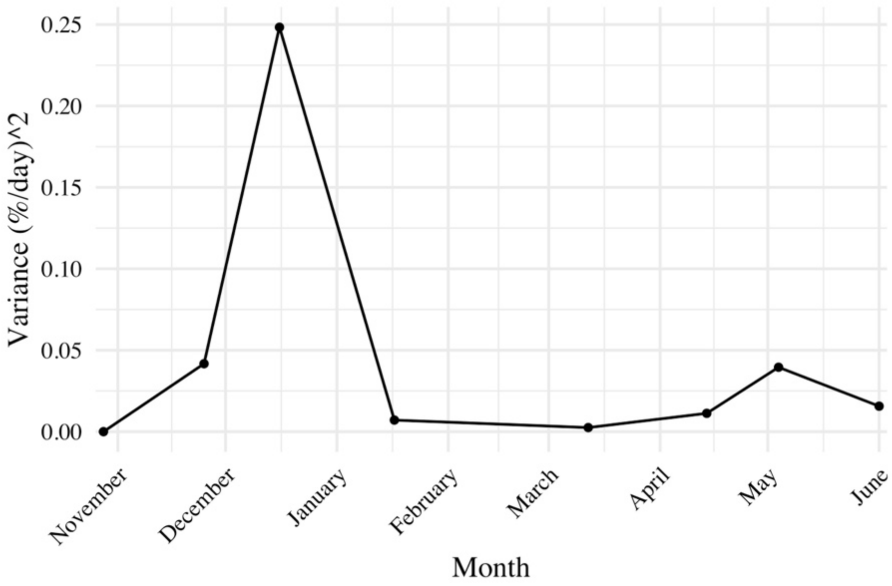

Figure 3.3: Temporal variation in daily growth rate variance among donor patches. The graph displays the variance ($(\%/day)^2$) in seagrass daily growth rate across monitoring dates from October 2024 to June 2025.

3.3. Buoy Impact Analysis

The Pearson correlation test did not identify a statistically significant relationship between mooring anchor distance and seagrass percentage cover ($p = 0.261$), nor with shoot density ($p = 0.650$), nor with shoot height ($p = 0.252$).

The Welch's t-test comparing buoyed and unbuoyed areas showed statistically significant differences for all three structural variables (Figure 3.4). Mean percentage cover was 40.34% in Buoy sites and 90.63% in NoBuoy sites ($p < 0.001$). Shoot density was 25.41 shoots/100 cm^2 in Buoy sites and 76.38 shoots/100 cm^2 in NoBuoy sites ($p < 0.001$). Mean shoot height was 21.11 cm in Buoy sites and 39.65 cm in NoBuoy sites ($p < 0.001$).

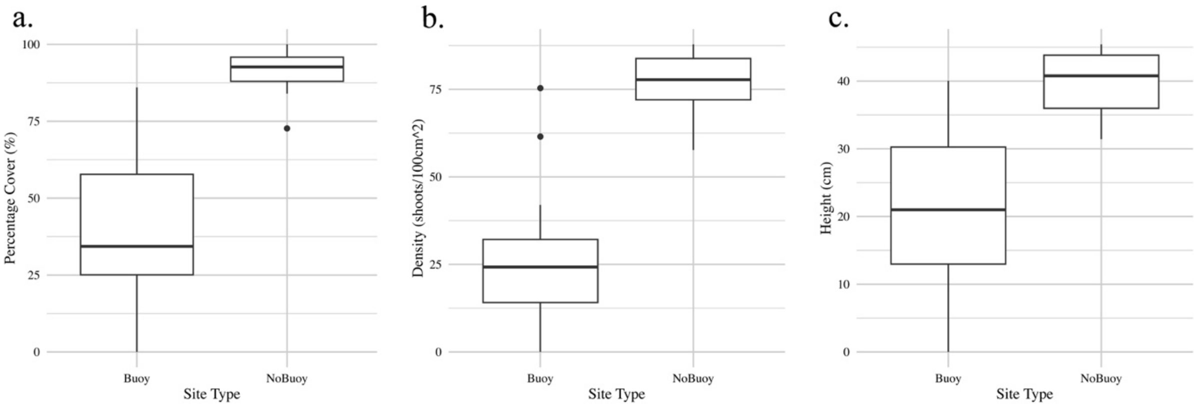

Figure 3.4: Boxplots showing seagrass structural attributes in sites with moorings (Buoy) and without moorings (NoBuoy). a. Mean percentage cover (%); b. Shoot density (shoots/100 cm^2); c. Shoot height (cm).

4. Discussion

The combined effect of minor but persistent disturbances appears to have maintained the transplanted meadow in a sub-threshold state, incapable of returning to a self-sustaining trajectory. This outcome underscores the complexity of restoring seagrass ecosystems in areas subject to chronic anthropogenic stress. Although some transplanted units exhibited localised improvement, the meadow as a whole failed to recover structurally or functionally. The absence of spatial coherence, along with continued degradation of key ecological indicators, suggests that the system remained ecologically unstable throughout the study period. The inability to restore structural integrity and spatial continuity across the restored area points to the presence of ongoing external stressors preventing the meadow from surpassing critical recovery thresholds.

The temporal evolution of seagrass structural parameters across all receiver sites was characterised by a consistent decline in shoot density and height, while percent cover exhibited moderate fluctuations. These trends allowed the identification of different conditions under which seagrass meadows in the study area appear to respond to environmental stressors. The observations suggest different possible scenarios. First, in healthy meadows, the presence of small-scale disturbances may lead to localised impacts but do not compromise the overall integrity of the meadow. When such disturbances are either temporary or sufficiently limited in spatial extent, the seagrass population can recover through natural growth and lateral expansion. Second, when a major disturbance is introduced (such as the physical impact caused by large-scale construction) the recovery of the meadow becomes unlikely, even in the presence of active restoration interventions, unless all additional sources of disturbance, including minor and chronic ones, are simultaneously addressed. In such cases, persistent low-intensity stressors may prevent the system from reaching the ecological conditions necessary for long-term recovery. To synthesise and represent these population trajectories under different disturbance regimes, a conceptual framework has been included (Figure 4.1).

Drone-based observations and quadrat-level data both documented consistent declines in structural metrics and a lack of coordinated recovery across monitoring events. These findings support the interpretation that meadow fragmentation persisted, likely due to the cumulative impact of low-intensity disturbances. This pattern is consistent with the concept of ecological thresholds, which posits that ecosystems under continuous stress may become locked in a degraded state unless all major and minor pressures are simultaneously mitigated (Carr et al., 2012; Maxwell et al., 2017; Scheffer et al., 2001).

One possible explanation for the limited recovery is that the removal of a single major stressor, such as the construction platform, was not sufficient to trigger meadow-wide regeneration. Despite the cessation of direct mechanical disruption, other persistent factors such as boat traffic, mooring systems, recreational use, and small-scale fishing activity likely continued to exert pressure on the transplanted area. These forms of diffuse disturbance may not cause immediate collapse, but they can prevent regeneration by maintaining the system below its recovery threshold over time (Unsworth et al., 2017).

Temporal variability in growth synchrony among planting units further supports the idea of fluctuating resilience. Periods of high spatial variance in daily growth rate suggest asynchronous responses among patches, possibly reflecting heterogeneity in microhabitat conditions or localized exposure to disturbance. Conversely, short intervals of low variance indicate that transient stabilisation may have occurred, although not sustained long enough to initiate full recovery. Such instability is characteristic of systems on the verge of a regime shift, where feedback mechanisms and external pressures interact to suppress resilience (Scheffer et al., 2001).

The comparison between Buoy and NoBuoy areas provided further evidence that mechanical disturbance from mooring systems remains a key limiting factor for seagrass structure. Lower values of seagrass cover, shoot density and height in moored areas, regardless of distance from the anchoring point, suggest that these systems exert localised yet persistent pressure on the sediment and plant community. While some studies have documented distance-dependent effects (Unsworth et al., 2017), our results indicate that the impact was concentrated near the source but sufficient to impair recovery across adjacent plots. These findings are in line with previous work demonstrating that physical damage caused by chains and anchors leads to degradation of the seagrass canopy and rhizome network (Hastings et al., 1995; Luff et al., 2019; Paling et al., 2003; Serrano et al., 2016).

The donor sites exhibited a markedly different pattern. Despite undergoing disturbance through sod removal, they demonstrated rapid and consistent structural recovery. This divergence from the receiver sites strongly suggests that the absence of permanent stressors, particularly fixed mechanical pressures such as moorings, played a critical role in maintaining the resilience of donor meadows. The observed recovery in both sediment treatments also indicates that sod extraction, when carefully executed, does not compromise the long-term integrity of donor meadows (Matheson et al., 2017).

The divergent performance between donor and receiver sites reinforces the notion that the ecological context, including the type, frequency, and duration of disturbance, is fundamental to restoration outcomes. In donor areas, where anthropogenic impacts were minimal or temporary, the ecosystem was able to absorb the transplant-related stress and return to pre-disturbance conditions. In contrast, the receiver meadow, even after transplantation efforts, remained in a degraded state, likely due to the continued influence of multiple overlapping pressures.

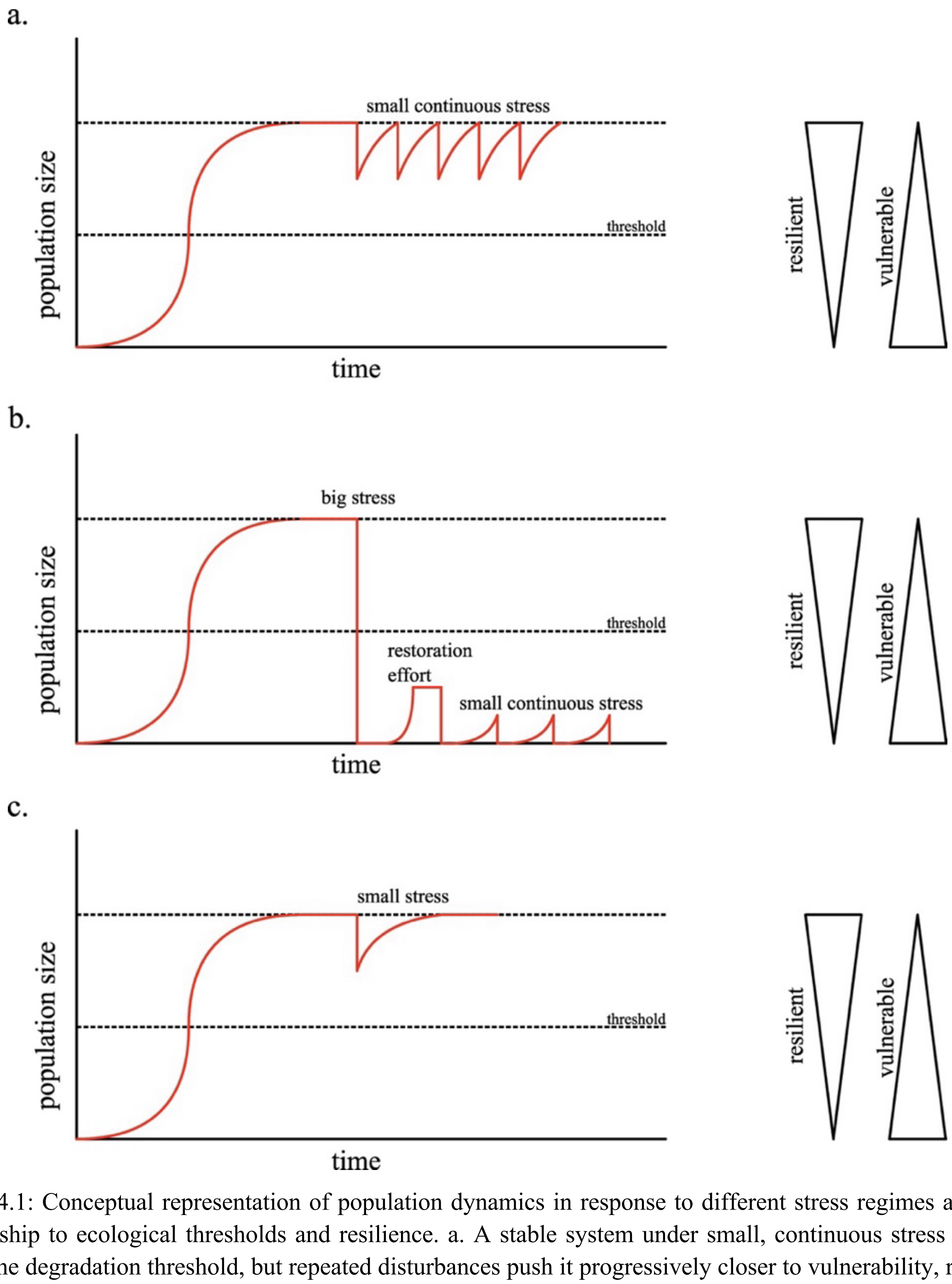

Figure 4.1: Conceptual representation of population dynamics in response to different stress regimes and their relationship to ecological thresholds and resilience. a. A stable system under small, continuous stress remains above the degradation threshold, but repeated disturbances push it progressively closer to vulnerability, reducing resilience. b. A major acute disturbance causes a collapse below the threshold; restoration efforts alone are insufficient to recover the system if smaller, persistent stressors remain. c. A transient, low-magnitude disturbance temporarily affects the population, but resilience allows full recovery as the system remains above the threshold. Triangles on the right illustrate the balance between resilience and vulnerability under each condition.

These results support the idea that successful seagrass restoration depends not only on the removal of primary stressors but also on addressing chronic, low-level disturbances. Without eliminating these background pressures, even intensive restoration efforts may fail to reach ecological thresholds necessary for full recovery. This underscores the importance of incorporating landscape-scale disturbance assessments into restoration planning and highlights the need for long-term, adaptive management strategies capable of responding to multiple, interacting stressors (Duarte et al., 2020; van Katwijk et al., 2016).

Despite the insights gained, this study was limited by its relatively short duration and the constraints imposed by ongoing anthropogenic activity within the study area. Future research should extend monitoring over multiple seasonal cycles and explore restoration success under different management scenarios, including temporary exclusion of stressors such as boating and fishing. Strengthening the link between ecological thresholds and management thresholds could help ensure that restoration efforts move beyond initial establishment and towards long-term ecosystem recovery. By addressing these knowledge gaps, future research can provide a stronger foundation for restoring and monitoring seagrass in the Algarve region.

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