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Ecological impacts, efficacy and economic feasibility of algal mat removal from temperate intertidal mudflats under blue nitrogen trading schemes

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ABSTRACT

Driven by eutrophic conditions, AM (algal mat) proliferation is now ubiquitous in coastal areas generating significant ecological and economic impacts. The need to mitigate negative effects has prompted the exploration of removal methods, but neither the success nor the impacts on intertidal mudflats have been assessed. Limited success using a specially-adapted vessel, prompted a shift to manual removal by hand-rake at two UK (Portsmouth and Poole) and two French study sites (Brittany and Normandy). Significant reductions in AM biomass and percentage cover were only observed at one site (Portsmouth), in contrast to significant temporal effects throughout the 180 days at each site. Significant effects of removal on the benthos and birds were also limited to an increase in organic content at Brittany and a reduction in macrofaunal abundance at Poole but with all sites dominated by temporal effects. To assess if AM removal can be used to ameliorate excess nitrogen (N) we calculated the amount of N that could be removed from a site and its potential cost-effectiveness (price of N credit after subtraction of removal costs) within an NTS (Nutrient Trading Scheme). N export by AM removal is influenced by site and season, for example, 66 kg N ha^{-1} yr^{-1} (winter) to 95 kg N ha^{-1} yr^{-1} (summer) at Poole. N removal rates from some sites (Poole, all seasons; Brittany, autumn) are comparable to other Nature-Based Solutions (NBSs) such as clam aquaculture. However, a single annual AM harvest at these sites yields lower N removal rates compared to seaweed, mussel, and oyster aquaculture. Using a global mean N credit price, the removals at Poole and Portsmouth have medium/high cost-effectiveness across all seasons, potentially generating up to half a million pounds of N credits, which could be increased if post-harvesting value-chains were maximised e.g. biofuel production. Although, implementation at scale could rapidly reduce the many impacts of AMs and contribute to the blue-green bioeconomy revolution, to improve water quality, AM removal must be framed within a multifaceted management process.

1. Introduction

Macroalgal blooms have become increasingly frequent in coastal regions, particularly in areas with increased human activity (Rabalais et al., 2009; Ye et al., 2011; Smetacek and Zingone, 2013; Bermejo et al., 2023). These dense mats of algae can float in the water column or grow on or entrained within sediment, often accumulating substantial biomass (e.g. >10 kg m^{-2}) in a single estuary (Lyons et al., 2014; Lenzi, 2017), but also extending thousands of kilometres (Liu et al., 2009). Species are predominately green, fast-growing and stress resistant (e.g. *Ulva*, formally *Enteromorpha* spp.) (Littler and Littler, 1980; Hayden et al., 2003), although other taxonomic groups (e.g. *Sargassum* spp.) can proliferate rapidly (Gower et al., 2013; Smetacek and Zingone, 2013; Wang et al., 2019 and Lamb, in press.).

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Eutrophic conditions are the main driver for algal mat (AM) proliferation (Michael Beman et al., 2005; Maier et al., 2009), linked to both point and non-point loadings of limiting nutrients such as N (nitrogen) and P (phosphorus). Despite many legislative drivers (see Hughes et al., 2022), their limited success in reducing inputs (Boesch, 2019), combined with future global predictions that ~1.15 million km^2 of coastal waters have the potential to become eutrophic (de Raus Maure et al., 2021) indicate that the significant impacts of AMs (see Lyons et al., 2014) are still globally extensive and increasing. Improving sewage treatment systems, the reduction of fertilizer use and catchment-sensitive farming are common approaches to tackle eutrophication (Duarte and Krause-Jensen, 2018). Nevertheless, the effects of 'upstream' interventions can take decades to reach the coast because ground water must percolate through the substrate (Luijendijk et al., 2020). This, combined with food security and productivity concerns for more 'radical' changes in farming practices (Dessart et al., 2019) and the expense of upgrading sewage treatment systems (Pajares et al., 2019) has led to the search for more cost-effective and rapid nutrient bioremediation methods.

Coastal estuaries and harbours contain key intertidal habitats (e.g. mudflats, seagrass, saltmarsh, oyster reefs) that provide significant ecosystem services (Watson et al., 2020) and are often protected under numerous conservation designations. These habitats are targets for global restoration efforts, which are being increasingly linked to carbon, nutrient and more recently biodiversity trading schemes i.e. MBIs (Market Based Instruments). MBIs use market mechanisms to provide incentives for private operators to take measures towards targeted environmental improvements, or the avoidance of environmental deterioration. Lockie (2013) presents an overview of MBIs, and includes PES (payments for ecosystem services), pollution taxes, cap-and-trade systems, and eco-certification into this term. PES, in turn, can be defined as being "voluntary transactions between service users and providers that are conditional on agreed rules of natural resource management for generating offsite services" (Wunder, 2015). NTSs (Nutrient Trading Schemes) sit within PESs and allow an organisation to offset nutrient (usually N) pollution generated by its activities (e.g. building houses or direct inputs) (Ferreira and Bricker, 2016; Gren and Elofsson, 2017; Filippelli et al., 2022). Installing on-site removal and attenuation measures (e.g. sustainable urban drainage systems) is not always possible, therefore, the organisation purchases credits that invest in offsite actions (e.g. wetland creation, agricultural land withdrawal etc.) (Hasselström and Gröndahl, 2021; Filippelli et al., 2022). A much clearer understanding of permeance of export and the influence of habitat quality on bioremediation is needed before NTSs could be used to 'invest' directly in restoration of intertidal habitats with significant bioremediation capabilities (Watson et al., 2020; DePiper et al., 2017). In contrast, credits generated by aquaculture (Ferreira and Bricker, 2016) where the nutrient accumulated in the biomass is removed (exported from the system) at harvesting shows considerable promise (e.g. Kotta et al., 2020; van den Burg et al., 2022; Barrett et al., 2022; Filippini et al., 2023), but is hampered in many coastal locations by co-location interactions with the protected coastal habitats and other users (Gouvello et al., 2017) and poor water quality impacting on the cultured species (Webber et al., 2021).

To date, physical removal of AM has generally been used to mitigate health, social and economic impacts in coastal areas directly used by people (e.g. preventing recreational activities, reducing tourism; limiting risks to visitors' health (Charlier et al., 2008; Wang et al., 2009). For example, prior to the 2008 Olympics AM removal costing US$87 million was implemented in China (Wang et al., 2009), and along the French coast AMs are removed routinely from tourist beaches (Charlier et al., 2008). More recently, harvesting of beach-cast AM from sandy beaches for agricultural purposes (e.g. as a fertilizer and soil improver) has now re-emerged as a measure for curbing eutrophication in the Baltic Sea (e.g. Söderqvist et al., 2021). For the first time, we posit that the physical removal of AM biomass from intertidal mudflats could: a) reduce the many direct habitat impacts and b) contribute to the amelioration of eutrophic coastal systems by exporting N from the system. Most importantly, NTSs could be used as the mechanism to make the process financially viable and cost-effective.

Unlike sandy beaches (see Söderqvist et al., 2021), physical removal from intertidal mudflats cannot utilise heavy machinery, requiring the development of a novel boat platform that can remove AMs when the tide is in. To assess the long-term success of removal (i.e. amount of biomass removed and if the AM returns to cleared areas) we removed the AM manually due to the limited success of the mechanical method. The AM biomass as well as sediment characteristics, benthic and wading bird communities were monitored regularly to assess effects and performing the method simultaneously at four sites enabled us to integrate the influence of site-specific conditions (e.g. the seasonal evolution of the AM and sediment characteristics). To realise our final aim of assessing the potential (i.e. cost-effectiveness) for nutrient removal at scale within an NTS we firstly measured the tissue concentration of N in AMs from the four sites and seasonally. These are combined with AM coverage data at the site level, enabling the tonnes of N that could be exported if AM was removed from each site to be calculated. Finally, potential costs for combinations of site/season were generated using standardised costs for mechanical removal methods. The valuation approach multiplies expected N removal (t) by the value of the N credit within existing, relevant trading programs (Dvarskas et al., 2020). We, therefore, used a PES scheme with credit values based on existing global markets (Barrett et al., 2022), but also a terrestrially-derived one developed for the Solent region of the UK linked to housing development (Hughes et al., 2022).

The global blue-green bioeconomy revolution, combined with Payment for Ecosystem Services (PESs) and Nutrient Trading Schemes (NTSs) (see Thomas et al., 2021), has the potential to support environmental managers and policymakers in making decisions to more rapidly tackle the global eutrophication and algal mats (AM) problem with a holistic approach.

2. Materials and methods

2.1. Hand and mechanical AM removal

Four sites located in Portsmouth, Poole, Normandy and Brittany (Fig. 1) were selected for the removal experiments, as they are geographically distinct despite sharing the same water body (The Channel).

Relevant permissions (UK: Natural England, Marine Management Organisation; France: Maritime Affairs Department) were obtained. Within each site, eight 100 m^2 areas at the same mid shore height and at 10 m intervals were delineated; four to be cleared of AM and four control areas, in an alternating pattern. A shallow draft (0.80 m) aluminium vessel (Waste Cleaner 83 Algues) was modified for AM removal (Fig. S1). A hydraulically-operated fork (4 m wide) attached to the front of the vessel was designed to collect the AM and store it on the vessel until reaching the shore. The vessel underwent trials at the French sites in August 2022. However, the collected amount of AM was considerably less than expected and not sufficient to see a visible reduction in AM cover from the cleared area. Given COVID-19 prevented future trials/development, a shift to manual hand removal was deemed necessary to assess the success and consequences of removing AMs from intertidal mudflats with all sites starting in August 2022 (Portsmouth: 5th August, Poole: 10th August, Brittany: 10th August, Normandy: 23rd August).

To assess the impacts of AM removal, hand-held rakes (910 mm wide, 8 mm long teeth, 15 mm gaps) were used, but to prevent trampling the removal areas were reduced to 25 m^2. The surface of the sediment was raked from the boundary, clearing the AM to the edge before placing it on a sledge for return to the laboratory for processing. Site surveys were undertaken one day prior to the removal (T0) and then at T7, T30, T90 and T180 post-removal (except Normandy which was not surveyed at T180 due to logistical constraints). AMB (AM dry biomass) and %AM

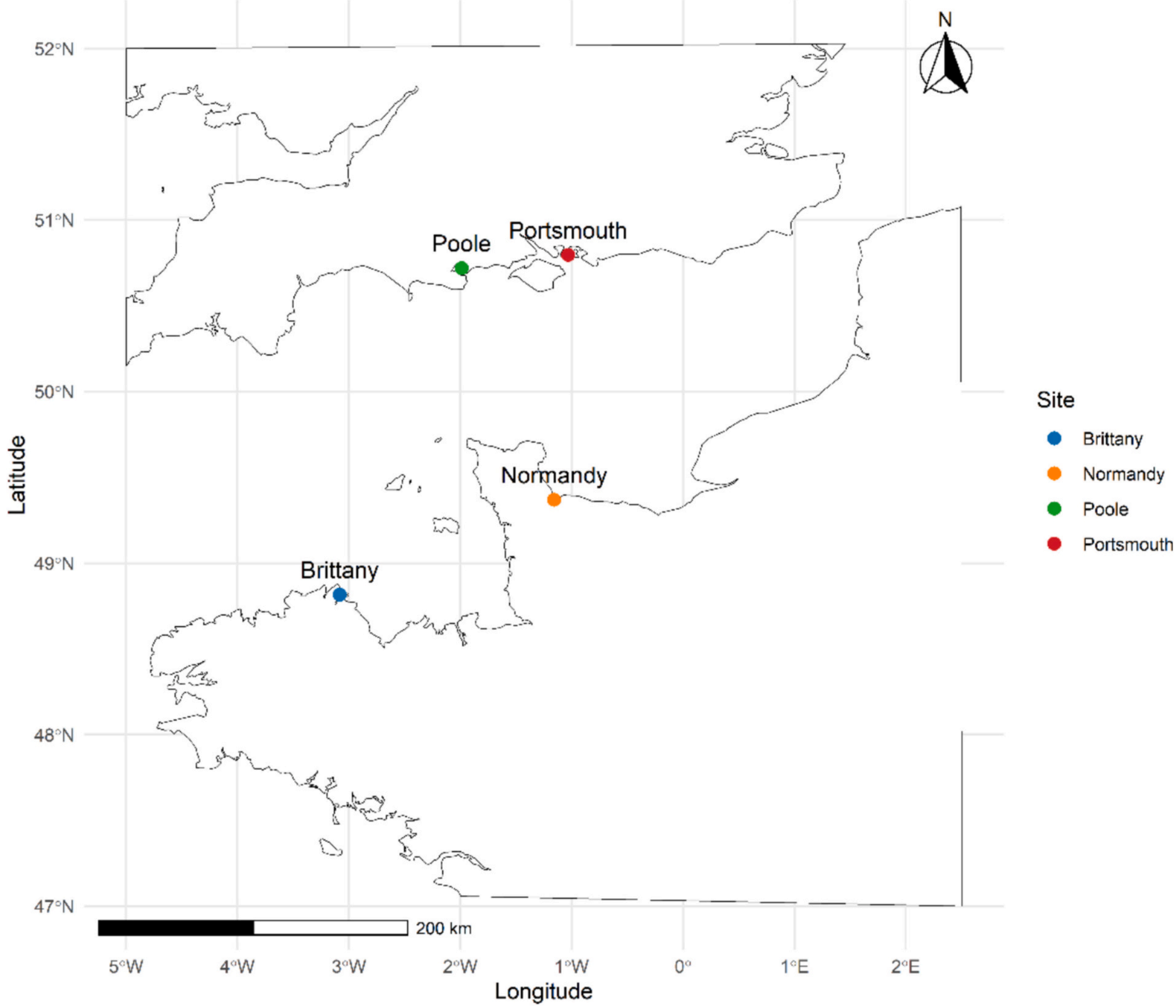

Fig. 1. Map showing location of the four trial sites in the UK and France.

(AM percentage coverage of the sediment) were measured by randomly placing five 0.25 m^2quadrats in each area. Within the quadrat the %AM was recorded and then a knife was used to cut around the quadrat, and all AM on the surface was removed. The area was then checked for entrained AM using Environment Agency (2016) methods with %AM recorded and the entrained AM removed again. All AM samples were placed in small mesh laundry bags (0.5 cm × 0.5 cm mesh, 30 cm × 40 cm in size) and then washed for 5 min in tap water to remove sediment before transferring the samples into buckets for additional debris to settle out. Once rinsed, the sample was spun in a salad spinner (20 cm × 18 cm) for 1 min to remove moisture. Each sample was then weighed before being dried at 60 °C for 72 h to calculate AMB. All AMB and %AM values are of the surface algal mat only, due to little or no entrained AM at all sites.

2.2. Impacts of removal

A 15 cm diameter PVC plastic pipe was used to extract sediment cores of a depth of 30 cm. Sediment was frozen (−18 °C) before analysis for particle size (See S2) (Blott et al., 2004) (wet and dry sieving followed by laser diffraction analysis (Mason, 2016) and organic matter (carbonate removal and sulphurous acid/combustion at 1600 °C/NDIR) (Heron et al., 1997). Macrofauna were collected using a second core, with sediment fixed in a 4 % seawater formaldehyde solution. Sediment was washed through a 0.5 mm sieve and the retained macrofauna identified, counted and then weighed (See S3). To ensure accuracy, a minimum of 10 % of the samples were repicked for additional quality control under the NE Atlantic Marine Biological Analytical Quality Control (NMBAQC) Scheme. Bird surveys were performed starting 1 h before low tide, with cleared and control areas observed (binoculars MARINE™ 7 × 50) from a distance of ~70 m. Each area was observed for 5 min with the observer recording the number of individuals of each species (See S4). This process was repeated three times per area and at each time point, although no observations were performed at T180 at Normandy.

2.3. Scale up

AM nutrient assimilation capacity (N tissue content) was investigated seasonally (March, June, September, and December 2021) by deploying nine quadrats (0.25m^2) to represent three %AM levels: low (0–33 %), medium (34–66 %), and high (67–100 %). In each quadrat the %AM cover was recorded, and corresponding AM collected for processing as above, prior to N-analysis by an external supplier using a Thermo Fisher Scientific Flash*Smart* Elemental Analyser (Thermo Scientific, 2018). Briefly, samples were oven dried to a constant weight and finely ground in a pestle and mortar with 2–3 mg introduced into the combustion reactor via the Thermo Scientific™ MAS Plus Autosampler with oxygen. A standard was run every fifth sample to ensure that the instrument is performing to within its calibrated tolerances (for element concentrations below 10 %, the tolerance is 0.1 %, above that is 0.3 %).

Previous surveys of the four sites (Watson et al., 2020; Witt, 2022; Ballu et al., 2024; Orvain et al., 2012) were used to generate the number of ha of intertidal sediment covered by AM at each of the sites. This was combined with a mean %AM generated from 24 months' worth of monthly surveying from January 2021 until December 2022. Ten quadrats (0.25 m^2) were placed randomly over each site with %AM recorded and removed for cleaning and weighing for AMB. To calculate wet weight AMB per ha and tonnes per site we used the WW of collected AM at different densities during seasonal sampling in 2021 in spring, summer, autumn and winter at each site. Nine quadrats were selected to represent varying algal mat coverage percentages: low (0–33 %), medium (34–66 %), and high (67–100 %). At each quadrat, the AM% cover was recorded, and corresponding algal mat samples were collected for CHN analysis (Table S5).

Due to a lack of success with the mechanical removal method we used harvest rate of Crane and Ramsay (2012). In their US trial (the AM was composed of *Ulva* spp.) they used a boat-mounted conveyor belt system with efficiency dependent on the type of AM encountered. When the AM was fragmented and sitting on the substrate surface a mean removal rate of 3.6 t h^{-1} was achieved. This increased to 5.4 t h^{-1} if AM was healthy and lifting off the substrate slightly and a maximum of 8.1 t h^{-1} if the AM was floating in the water column. These removal rates were then divided by the WW AMB per site to determine the number of hours it would take to remove the AM from the whole site. The financial cost of removal was determined by using the $250 (in 2012) per hour of Crane and Ramsay (2012), which in November 2023 is £280 h^{-1} after conversion to GBP from USD and adjusted to account for inflation (using consumer price index inflation data, OECD). This figure was then multiplied by the number of hours required to remove all the AM from a site to get an overall cost per site. To generate cost-effectiveness the cost price was subtracted from the potential value of the AM under using two NTS price points (£25 N kg^{-1}, mean of 75 studies from Barrett et al., 2022 and £3000 N kg^{-1} from the terrestrially-derived Solent region). The DW AMB (g m^2) obtained through field surveys was multiplied by the N tissue content values for each season and scaled to the hectare and site, generating the total N available in tonnes. Cost-effectiveness is defined into three levels dependant on profitability per hectare: low (<£100), medium (£100–£1000) and high (£1000 +).

2.4. Data analysis

A repeated measures ANOVA (IBM SPSS Statistics 27) was used to assess the differences between control and removal treatments at each site for AM% and AMB. The Bonferroni confidence interval adjustment was applied to account for estimated marginal means. If Mauchly's test revealed a violation of the sphericity assumption, degrees of freedom were corrected using Greenhouse-Geisser estimates of sphericity (O'Brien and Kaiser, 1985). Particle size analysis was performed using GRADISTAT Ver. 9.0 (Blott and Pye, 2001) with a geometric method applied (Folk and Ward, 1957) to generate a mean particle size and % mud. Macrofaunal species data were synonymised with the WoRMS database before removing planktonic species and juveniles. Both univariate and multivariate methods (square root transformed Bray Curtis similarity matrix of species abundance) followed by a PERMANOVA analysis were performed using Primer v 7.0 (Anderson and Willis, 2003). Univariate indicators (Species number [S], number of individuals [n] and Hill's N1 diversity index) were further analysed using a repeated measure GLM using SPSS Version 29.0.1.0 with Bonferroni adjustment to assess effect of treatment, time and any interaction. Low numbers of birds precluded site-specific analysis, therefore, a PERMANOVA analysis was performed with site, time and treatment as factors.

3. Results

3.1. %AM cover and biomass

At both UK sites prior to removal (T0), mean %AM was very high with 96 ± 2 % (control) and 95 ± 2.2 % (removal) coverage at Portsmouth and 100 % for both treatments at Poole (Fig. 2). For Poole there was no reduction over time in coverage except at T90 when mean %AM dropped to 74 ± 42 % at the control site and at T180 when all the AM disappeared. At Portsmouth a steady reduction in mean %AM was seen for the control site from 88 ± 5.3 % (T7) to 71 ± 8.7 % (T90). Although there was a reduction to 35 ± 7 % at T7 for the removal site, at T30 the mean %AM then increased before falling again to 42 ± 8 % at T90. No AM was recorded at Portsmouth at T180. ANOVA results confirm that there was a significant effect of time at Portsmouth (F = 93, $p = 0.001$) and Poole (F = 107, $p \leq 0.001$) with removal only having a significant effect at Portsmouth (F = 19, p = 0.001). At the French sites, mean %AM was lower and more variable. For example, at Normandy the mean %AM for the control at T0 was 28 % compared to 18 % for the removal treatment. Post T0, mean %AM coverage never exceeded 22 % (T30, control). At Brittany, mean %AM coverage peaked at 69 % (T30 removal) and 65 % (T7 control), remaining above 30 % until the AM disappeared from the site (T180). For the French sites, only time had a significant influence on %AM (Brittany (F = 13, p ≤ 0.001) and at Normandy (F = 13, p ≤ 0.001), with details of significance detailed in S6 between time points.

Unsurprisingly, the AMB (Fig. 3) tracks the site and time differences reported for %AM. Poole had the highest mean AMB (e.g. T0 control: 205 g m^{-2}), but despite a similar %AM to Poole the AM at Portsmouth was a lot thinner, e.g. AMB at T0 in the control was 69 g m^{-2}. The removal process only had a significant effect at Portsmouth (F = 3, $p = 0.012$) reducing mean AMB to 15 g m^{-2} compared to 62 g m^{-2} for the control at T7, before mean AMB in both treatments subsequently increased. The repeated measures ANOVA revealed a significant effect of time for Portsmouth (F = 19, p = 0.001), Brittany (F = 5, $p = 0.014$) and Poole (F = 59, p = 0.001), but not Normandy (F = 3, $p = 0.08$), with details of significance detailed in S6 between time points.

3.2. Effects of removal on sediment characteristics, macrofauna and birds

The sediment at Portsmouth was poorly sorted, with symmetrical skewness and mesokurtic kurtosis (see Table S2. The site predominantly featured medium silt at both control (mean particle size: 10.4 ± 0.8 μm) and removal (8.9 ± 0.6 μm) areas. Mud content was high, measuring 93 ± 2.2 % for the control and 96 ± 1.2 % for the removal sites. Organic content at T0 was 3.4 ± 0.3 % for the control and 3.7 ± 0.5 % for the removal. At T180, both control (7.3 ± 1 %) and removal (7.6 ± 0.8 %) sites had increased levels of organic content, but GLM repeated measures ANOVA revealed no significant effect of time or treatment on organic content or %mud.

Generally, the sediment type at Poole was very coarse silt, characterised by very poor sorting with symmetrical skewness, and an associated mean particle size of 53 ± 6 μm in the control areas at T0 and 50 ± 4.3 μm in the removal areas. The mean %mud at T0 was 52.3 ± 2.9 % for the control and 54 ± 2.4 % for the removal area. This percentage remained consistent throughout the sampling period with mud levels of 56 ± 1.5 % (control) and 54.3 ± 1.6 % (removal) at T180. The mean percentage organic content in the control and removal areas at T0 were 3.6 ± 0.5 % and 3.7 ± 0.7 % respectively, which increased slightly to 4.6 ± 0.6 % and 4.4 ± 0.4 % by T180. GLM repeated measures ANOVA revealed no significant effect of time or treatment on the %mud, however the effect of time (F = 32, $p = 0.004$), but not treatment was significant for organic content.

Fine sand dominated the Normandy site, with a mean particle size of 167 ± 9.7 μm at T0 for the control areas and 166 ± 6.3 μm at the removal areas. The mud content was 8 ± 1.9 % and 8 ± 1.5 % at T0 for

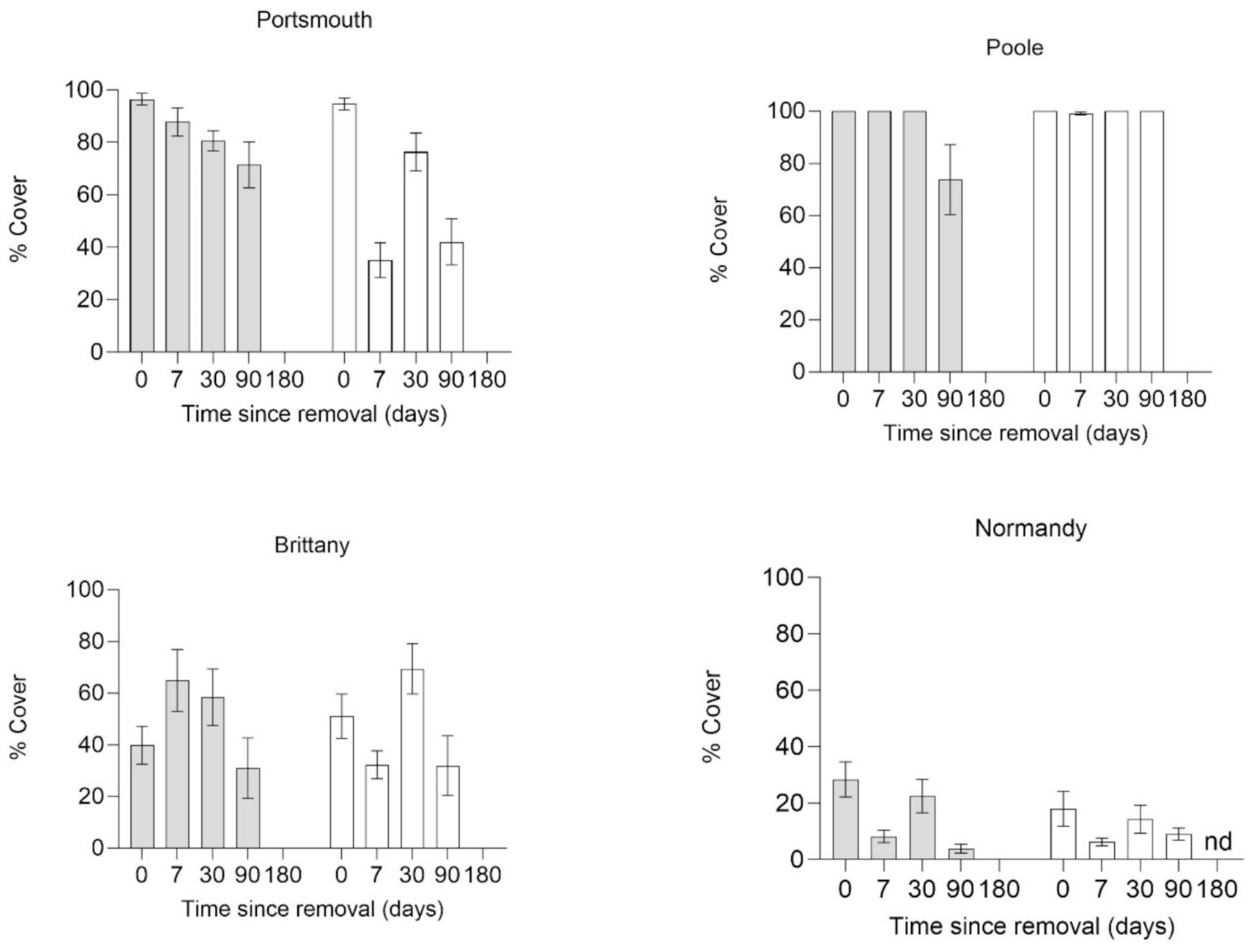

Fig. 2. Mean (±SEM) AM% cover at day T0, T7, T30, T90 and T180 for the four sites. Grey bars represent the control sites and the white bars represent the areas where AM was removed. nd: No samples were collected at Normandy at T180.

the control and removal, respectively, increasing to 12 ± 1 % and 11 ± 3.2 % at T90. Organic content was 1.7 ± 0.1 % at T0 for the control and 1.9 ± 0.1 % in the removal areas, compared to 6.1 ± 0.6 % and 8 ± 0.5 % at T90. Overall, the sediment displayed poor sorting, was strongly skewed with high kurtosis. GLM repeated measures ANOVA revealed a significant effect of time on organic content only ($F = 32$, $p \leq 0.001$).

In Brittany, sediment descriptions reflect coarse and very coarse silt. At T0, mean particle sizes were 41 ± 3.8 μm for the control and 68 ± 23.5 μm for the removal areas, with reductions to 28 ± 2.6 μm and 30 ± 1.8 μm, respectively, by T30. At T0 the mean percentage mud was relatively high (control areas: 64 ± 2.6 %; removal areas: 55 ± 2.5 %). The mean organic content at T0 was 3.2 ± 0.2 % for the control and 2.8 ± 0.1 % for the removal areas. Furthermore, while the mean organic content remained stable in the control areas at T180 (3 ± 0.5 %), it increased to 5 ± 0.5 % in the removal areas. Raw data were not provided so % mud ANOVA could not be carried out, however there was a significant effect of time ($F = 3$, $p = 0.028$) and treatment ($F = 4$, $p = 0.007$) on organic content in Brittany.

A total of 35 species across treatments and time points were identified from Portsmouth and at Poole, while in Brittany and Normandy there were 27 and 9 species, respectively (Table S3). In Portsmouth, four dominant taxa (nematodes, *Peringia ulvae*, *Tubificoides benedii*, and *Capitella* spp.) constituted 98.5 % of the total species recorded at the site across all time points and treatments. Poole had a similar pattern with dominant taxa (*P. ulvae*, *Tharyx killariensis*, *T. pseudogaster*, *T. benedii*, and nematodes). The species composition at Normandy was characterised by three major taxa (*C. capitata*, *Malacoceros fuliginosus*, and *Oligochaeta* sp.), accounting for 98.4 % of the total recorded species while the site at Brittany was dominated mainly by *P. ulvae* which made up 96.5 % of recoded species. An n-MDS ordination for the macrofauna community of each site with samples labelled by time and treatment is presented in Fig. 4. None of the sites' communities formed distinct groups around control and removal areas, however, clear groupings are visible by time point. The PERMANOVA analyses support the visual interpretation of Fig. 4 with highly significant temporal effects for all four sites (Portsmouth, $F = 7.10$, $p = 0.001$; Poole, $F = 2.6$, $p = 0.001$; Normandy, $F = 6.29$, $p = 0.001$; Brittany, $F = 3.13$, $p = 0.006$), indicating substantial changes over time, but no significant effect of treatment nor any interaction between treatment and time. The vast majority of the GLM analyses of the univariate measures (S, n and Hill's N1) confirm a significant effect of time only. Only at Poole, did the removal areas have significantly lower number of species than the control ($F = 3.5$, $p = 0.021$) but not Hill's N1 or S.

One hundred and sixty-five birds were recorded inside the removal/control areas during the study period across the four sites, with the majority recorded at Normandy (111 birds from five species (dunlin (*Calidris alpina*), black-headed gull (*Chroicocephalus ridibundus*), great plover (*Charadrius leschenaultia*), grey plover (*Pluvialis squatarola*) and common pigeon (*Columba livia*). *C. alpina* was the most abundant species, with 32 individuals in control plots and 36 in removal plots, while only one *C. livia* and *P. squatarola* were sighted. Poole recorded 19 birds from two species: *C. ridibundus* and herring gulls (*Larus arentatus*). There were 35 birds recorded at Portsmouth, predominantly European starlings (*Sturnus vulgaris*) (31), along with one *L. arentatus*, *C. ridibundus*, crow (*Corvus corona*) and Common Gull (*Larus canus*). No birds were recorded at any time point from Brittany. An n-MDS ordination

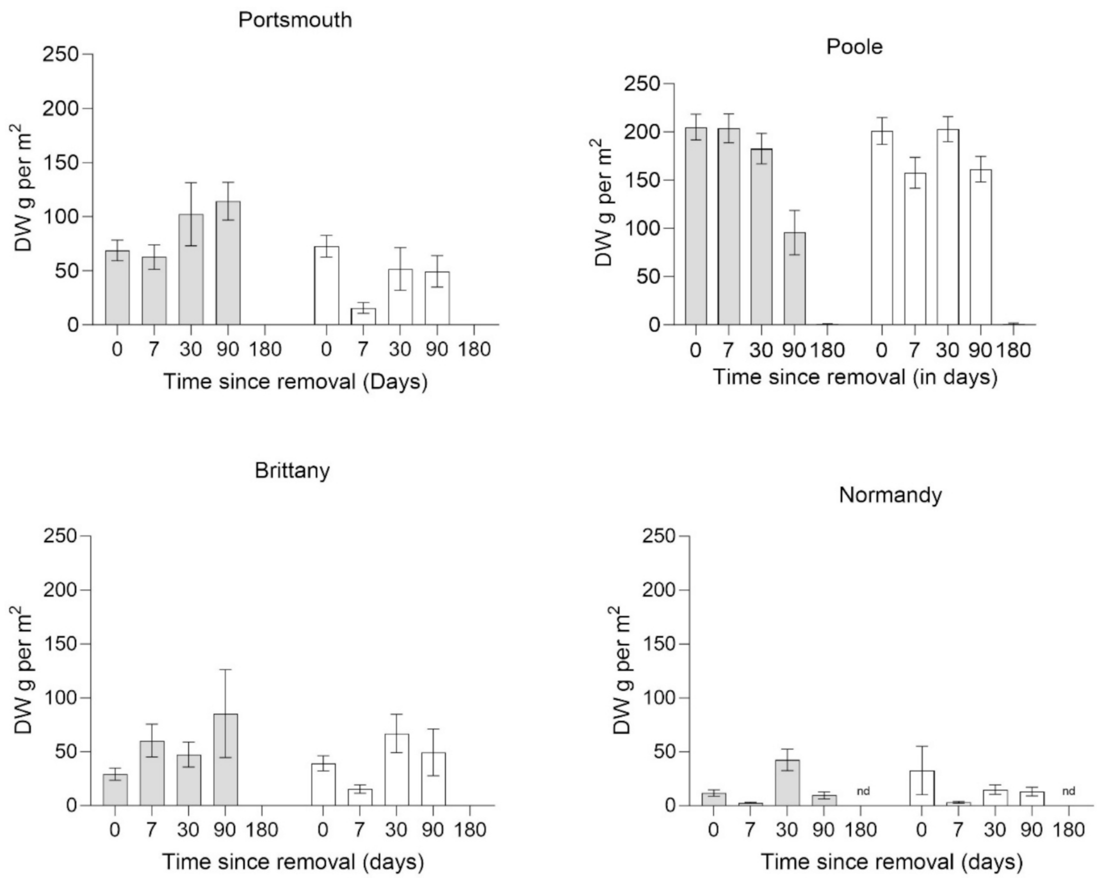

Fig. 3. Mean (±SEM) DW biomass (g m^{-2}), at T0, T7, T30, T90 and T180 for the four sites. Grey bars represent the control sites and the white bars represent the areas where AM was removed. nd: No follow up samples were collected at Normandy at T180.

combining the four sites with samples labelled by site, time and treatment is presented in Fig. 5. Although site and time-specific patterns are obvious, clear groupings for the effect of removal are not. PERMANOVA analyses support this with highly significant effects of time and clear differences between sites, but no significant effect of treatment ($F = 2$, $p = 0.1$).

3.3. Scale up

Prior to any scale up calculations of the potential amount of N that can be removed from a system (e.g. harbour/estuary), the amount of N in AM tissue needs to be assessed. Mean (±SEM) N assimilation capacities (presented as %N of AMB) at different AM densities for each site and across the four seasons are presented in Table S5. Analysis confirms that there are significant differences between sites ($F = 18.91$, $p \leq 0.000$) and season ($F = 22.27$, $p \leq 0.001$), but no effect of percentage coverage ($F = 0.82$, $p = 0.444$). For example, the mean %N of the AM from Portsmouth in spring (March) was 4.35 ± 1.9 %, which fell to 1.18 ± 0.9 % by June, before returning to 4.01 ± 0.5 % in winter. In contrast, mean %N remained relatively constant over the year at Poole, ranging from 4.08 ± 0.7 % (summer) to 4.86 ± 0.05 % (winter).

Table 1 summarises the potential N removal and economic value associated with AM removal at the four sites and across the seasons. Given the variability in the amount of N present in the AM (temporally and spatially) the mean %AM per site and season is paired with AM extent data (ha) for the intertidal sediment at each site (Watson, 2020; Witt, 2022; CEVA, 2023; Orvain et al., 2012). These generate AMB (wet weight) per hectare and per site, which using the season and site-specific %N and AMB (dry and wet weights), generates the tonnes of N that could be removed if either a hectare of AM or the whole site's AM was mechanically removed. Witt (2022) estimated that approximately 242 ha of the site at Poole is covered by AM. Assuming a consistent average percentage of AM across the harbour, this corresponds to approximately 198–235 ha covered in AM across different seasons, equating to 8–15 t/ha or 1715–3597 t $site^{-1}$. We estimate that there is between 13 and 22 t of N that could be removed from this site. Although Portsmouth has more intertidal sediment covered by AM (364 ha), the lower %AM coverage, AMB and N tissue content results in a lower and more variable amount of N that can be removed; ranging from 0.73 t in spring to just over 6 t in autumn. Even though Normandy has 1233 ha of AM-covered intertidal sediment, consistently low %AM and AMB across all seasons mean that 0.3–1.3 t of N could be removed from the whole site, equating to approximately a factor of 10 less per ha than either Poole or Portsmouth. With just a 100 ha of intertidal sediment covered in AM at the Brittany site (, combined with the highly variable %AM, generates N removal estimates in autumn of 3.4 t, compared with summer values that were an order of magnitude smaller (0.25 t).

Table 1 also includes the cost-effectiveness (i.e. potential profitability) of AM removal. Cost-effectiveness uses a £25 kg^{-1} mean NTS trading price from Barrett et al. (2022), combined with the estimated cost of mechanical removal. Mechanical removal is based on the hourly vessel costs, adjusted for inflation, from Crane and Ramsay (2012) and also incorporating different removal rates, which we have linked to the %AM. Depending on the season, the 13–22 t of N contained within the AM in Poole is worth between £325,500 (winter) and £555,000 (summer) within an NTS. The values per hectare (£1643–2361) always exceed the mechanical removal cost so cost-effectiveness for each season is categorized as high. For example, if all the AM was removed in summer, a 'profit' of £1984 ha^{-1} would be generated equating to £466,230 for the whole harbour. Although Portsmouth is not as cost

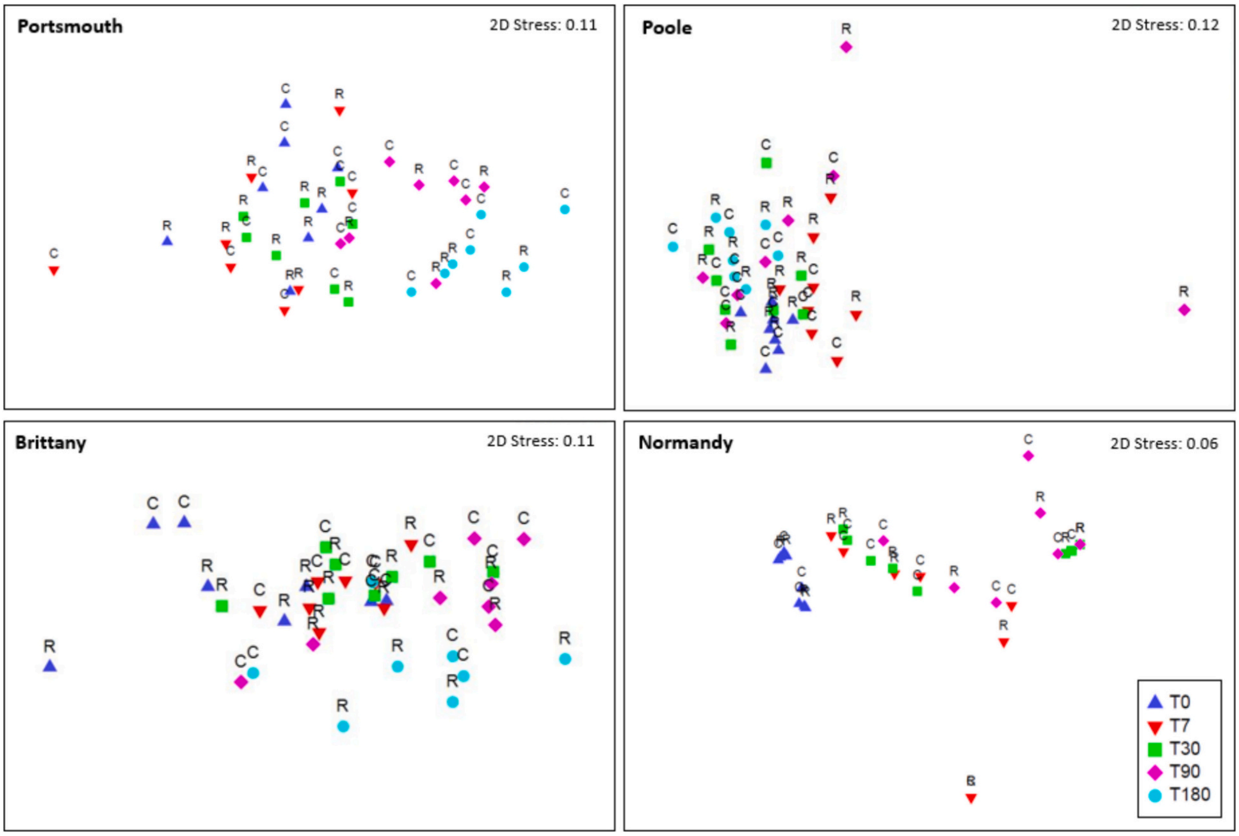

Fig. 4. Non-metric MDS plots of benthic community at the removal (R) and control (C) areas for Portsmouth, Poole, Brittany and Normandy at T0, T7, T30, T90 and T180. No data were collected for Normandy at T180.

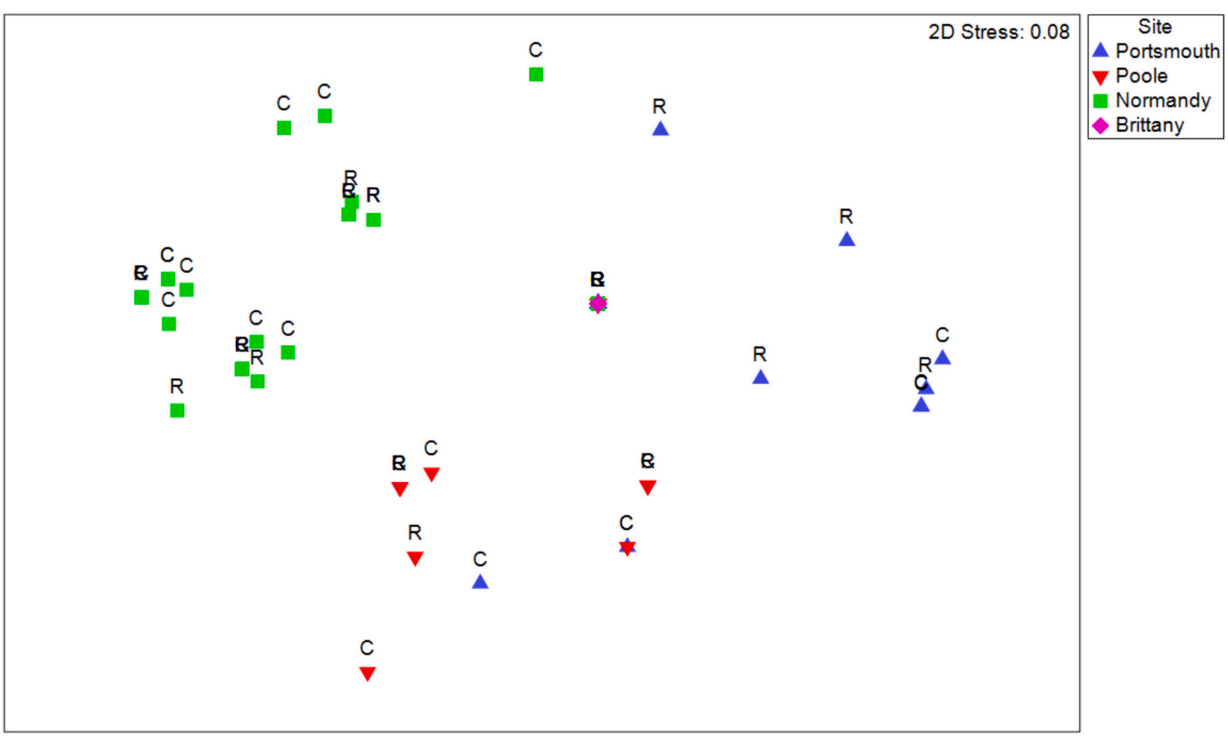

Fig. 5. Non-metric MDS plots of the bird community at the removal (R) and control (C) areas for Portsmouth, Poole, Brittany and Normandy at T0, T7, T30, T90 and T180. No data were collected for Normandy at T180. Birds were not observed at Brittany or Poole during T0 and T180. Similarly, no birds were recorded at Portsmouth during T30 or T90. This resulted in points being overlaid and obscuring those beneath.

Table 1
Table showing the cost effectiveness of each site and season for algal mat removal when compared to values of nitrogen trading schemes. Season: The time of year during which the data was collected; ha: The area in hectares (ha) of the site location; Average % cover: average percentage cover of present algal mat in each season; ha with AM: hectares of site covered in algal mat as determined by references; WW (t/ha): wet weight in tonnes per hectare; WW (t/site): the tonnes of weight wet per site. Efficacy Rate: The effectiveness of the removal process as determined by Crane and Ramsay based on tonnes per hour removed (H, 8.1 t (maximum harvest efficiencies)), (M, 5.4 t ("achieved when the sea lettuce was relatively healthy and was lifting off of the bottom slightly")), (L, 3.6 t, ("submerged lettuce")). Cost to remove (site): cost incurred to remove algae per site based on $250 = £280 adjusted for inflation from November 2011 to November 2023. Cost to remove (ha): scaled cost to remove in ha. Value per site: monetary value generated by site by multiplying kg available by £25 (value given to 1 kg N). This figure is then divided by ha at each site to generate value per site to infer whether or not the cost of removal is worth the N credit potentially available, as determined by low (<£100), medium (£100–£1000) and high (£1000 +).

Season	ha	Av % cover	ha with AM	WW (T/ha)	WW (T/site)	Efficacy Rate	Hours to remove	Cost to remove (site)	Cost to remove (ha)	Algal DW g/m^2	N g/m^2	Kg available	Value per site	Potential value per hectare	Difference (value - cost ha^{-1})	Cost Effective	Housing development scheme value
Portsmouth																	
Spring	364	21	76	2	24	L	6.7	£1867	£25	20.2	0.9	731	£ 18,275	£240	£216	M	£2,558,500
Summer		75	273	15	1354	H	167.2	£46,805	£171	130.2	1.6	4380	£109,500	£401	£230	M	£15,330,000
Autumn		72	262	23	2044	H	252.3	£70,657	£270	167.8	2.3	6110	£152,750	£583	£313	M	£21,385,000
Winter		22	80	4	108	L	30.0	£8400	£105	33.5	1.3	1070	£ 26,750	£334	£229	M	£3,745,000
Poole																	
Spring	242	86	209	8	1715	H	211.7	£59,284	£284	191.3	8.3	17,290	£432,250	£2068	£1785	H	£60,515,000
Summer		97	235	11	2568	H	317.0	£88,771	£ 378	177.1	9.5	22,200	£555,000	£2362	£1984	H	£77,700,000
Autumn		96	234	15	3597	H	444.1	£124,341	£531	269.4	8.9	20,850	£521,250	£2228	£1696	H	£72,975,000
Winter		82	198	11	2183	H	269.5	£75,462	£ 381	138.3	6.6	13,020	£325,500	£1644	£1263	H	£45,570,000
Normandy																	
Spring	1233	13	162	0.47	77	L	21.4	£5989	£37	23.9	0.82	1300	£32,500	£201	£164	M	£4,550,000
Summer		17	211	0.78	165	L	45.8	£12,833	£61	22.8	0.49	1040	£26,000	£ 123	£62	L	£3,640,000
Autumn		20	246	0.83	205	L	56.9	£15,944	£65	19.7	0.62	1530	£38,250	£155	£91	L	£5,355,000
Winter		10	127	0.16	21	L	5.8	£1633	£13	10.9	0.27	345	£8625	£68	£55	L	£1,207,500
Brittany																	
Spring	100	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Summer		19	19	3	59	L	16.4	£4589	£242	36.4	1.3	250	£6250	£329	£87	M	£875,000
Autumn		78	78	4	303	H	37.4	£10,474	£ 134	133.6	4.4	3420	£85,500	£1096	£92	M	£11,970,000
Winter		5	5	1.5	7.4	L	2.1	£576	£115	0.1	0.1	0.35	£8.75	£2	-£113	N/A	£1225

effective as Poole, removal of all AM during any season would generate substantial value (£18,275–152,750) and would also be cost-effective, although we have assigned this as medium. Driven by high %AM and AMB, removal of AM from Brittany is only cost-effective in autumn; removing 3.4 t of N from 100 ha valued at £1098 ha^{-1}. Despite the extensive area of intertidal sediment covered by AM at the Normandy site, the low %AM combined with the extremely low AMB means that a medium cost-effectiveness is only achieved for spring (e.g. maximum 'profit' per ha is: £163) with all others having a 'profit' of £55–91 ha^{-1} after subtracting removal costs. The final valuation replaces the global N-credit price with the (£3000) from the terrestrially-derived Solent region NTS linked to housing development (PUSH, 2021). Under this scheme the 'profitability' per site could be as high as £66,511,230 for Poole in summer and £18,259,343 for Portsmouth if the AM was removed in autumn.

4. Discussion

4.1. Success of AM removal from intertidal mudflats

The removal of AM using rakes did significantly reduce the mean %AM at Portsmouth seven days after removal. However, the success was short-lived, as the mean %AM subsequently increased in the removal area at T30. Apart from this time-limited success, none of the other sites had significant removal success. The most likely explanations for this rapid re-accumulation are the relatively small areas targeted combined with the dynamic nature of AMs (Louis et al., 2023). Significant temporal variability was present at all sites with considerable changes in %AM/AMB between adjacent sampling times (Figs. 1 and 2). Favourable conditions for growth at the time of removal (Singh and Singh, 2015; Nazari-Sharabian, 2018) combined with relatively rapid growth rates of dominant AM species such as *Ulva* spp. (Scanlan et al., 2007) would lead to rapid regrowth of remaining AM entrained within or on the sediment in the removal areas. AM could also have been washed into the removal areas from outside at all sites. Although Capuzzo and Foster (2014) have suggested some benefits to harvesting by hand, for example selecting the best quality macroalgae for human consumption, the low efficiency and physically demanding process combined with high labour costs and potential exposure to H_2S gas (Résière et al., 2020) would not make it appropriate for scaling even if our experiments had shown consistent reductions in %AM and AMB.

The most common AM harvesting method from muddy areas is using a vessel at high tide (e.g. Sfriso and Marcomini, 1996; Crane and Ramsay, 2012; Lenzi et al., 2015). Two approaches have been reported: the removal of floating AM using a conveyor belt system; the second method harvests submerged algae by means of a mounted cutting apparatus. No conveyor belt system vessel was available in the UK or France, while a cutting blade, usually used for upright submerged vegetation attached to the substrate, was not deemed to be suitable as the AM at the sites was not firmly attached or erect. The hydraulic rake design was, therefore, taken forwards as an alternative that would replicate hand removal methods. The inability to optimise the method within the timeframe of the project, due to Covid-19 pandemic restrictions, compounded by the vessel's draft reducing access to mudflat with high AMB, resulted in deployments at both French sites only removing small amounts of AM (~20 kg). The mechanised rake approach may offer some benefits, such as lower levels of sediment resuspension compared to conveyor belt systems (Crane and Ramsay, 2012; Lenzi et al., 2015), however optimisation would need to increase AM removal substantially to compare favourably with the 1–15 t (wet weight) h^{-1} reported (Lenzi et al., 2005; Crane and Ramsay, 2012), therefore, scaleup should focus on the conveyor belt system.

4.2. Impacts of removal process and the loss of AM

Despite all four sites sharing the same water body, the analyses confirm that they already differed in their sediment characteristics at the start of the experiment (T0). Generally, Portsmouth had the lowest mean particle size and highest organic content, followed by Brittany and then Poole. Normandy had sediment described as fine sand with a much lower organic content, % mud and higher mean particle size. The macrofauna communities reflect these sediment conditions, but are also typical of systems that are anthropogenically impacted. All four sites had relatively low diversity and were dominated by species such as *P. ulvae*, *C. capitata* and *Tubificoides* spp. (Guerra-García and Gómez, 2004) which are known to be tolerant to pollution, with three of the four sites having high numbers of *C. capitata* which are known to inhabit sediment below AMs (Bolam et al., 2000).

As hypothesised, the four sites differed in their bird communities and these also changed over time, but the birds did not reflect the diversity of wading birds routinely seen on intertidal mudflats. In fact, a number are of terrestrial origin (*C. livia*, *S. vulgaris*, *C. corone*), although separate author observations (Watson pers. Obser) have noted large flocks of *S. vulgaris* exploiting inter-tidal mudflats and moving across the AM at Portsmouth. Presumably they are feeding on *P. ulvae*, which would be the only species easily accessible to this bird. The very low number of 'true wading' birds recorded from all sites is not driven by season as T90 (November) and T180 (January) occurred when wading birds are still present in large numbers on European coastal habitats (Marra et al., 2023). Instead, we posit that the proximity of all four sites to human activity (e.g. people on a footpath at Poole) or buildings close to the shore (e.g. Portsmouth) may have reduced the utilisation levels by wading species as a result of human disturbance (Cardoni et al., 2008).

Given the only effect of the removal process was a reduction in a single univariate measurement (Hill's diversity index) at Poole Harbour for macrofauna we can be confident that the small-scale removal by raking has minimal impact on the sediment and associated macrofaunal/bird communities up to 180 days post-removal. Despite the global problem that AMs pose and the substantial number of studies that have investigated the impacts (see Lyons et al., 2014) very few have evaluated the effects of harvesting AMs. De Leo et al. (2002) used a mathematical model to evaluate costs/benefits in relation to bivalve production. Lavery et al. (1999) estimated the recovery time for macrofauna on sandier beaches after AM removal with a tractor and Lenzi et al. (2013) focussed on physical boat-driven resuspension of sediment, but not AM removal. None of these studies and even the more relevant study by Crane and Ramsay (2012) investigated macrofauna communities in muddy sediments or the birds that utilise them. While it is, therefore, impossible to compare our results directly, we should be cautious about extrapolating our 'minimal effect' conclusions. The significant changes in sediment characteristics and associated macrofauna recorded at all four sites are consistent with temporally heterogeneous soft sediment systems and compounded by dynamic AM processes in time and space (e.g. Rueda and Salas, 2003). The bird communities observed at all sites were of low diversity and number and not representative of the full wading and wildfowl species that utilise temperate intertidal mudflats (Pollitt et al., 2003). Most importantly, the lack of differences in %AM and AMB between control and removal areas except at one site (Portsmouth) means that it is unsurprising that differences in sediment and associated macrofaunal/bird communities would be evident. Therefore, any scale up of the removal process would still require an assessment of the impacts on the integrity and function (e.g. denitrification) of the intertidal mudflats and associated invertebrate and bird communities. In addition, the resuspension of sediment from disturbance which might elevate local nutrient concentrations (Lenzi et al., 2005, 2013) and increase the bioavailability of sediment-bound contaminants such as metals (Kalnejais et al., 2010) should be included in the assessment.

Of critical importance for sites impacted by AMs but protected under many levels of conservation legislation is whether the 'integrity' of the whole designated site is transformed. Site integrity has been defined as 'the coherence of its ecological structure and function, across the whole area, that sustains the habitat, complex of habitats and/or the levels of populations of the species for which it was classified' (European Commission, 2000). Localised changes in species, communities or biotopes from mechanical AM removal, which are also likely to be short term (e.g. Lavery et al., 1999) might be interpreted as not compromising site integrity, but some conservation agencies have concluded that even the loss of <1 % could adversely affect site integrity (Hoskin and Tyldesley, 2006). Just like other activities that have impact, but also benefit society (e.g. intertidal fisheries [Watson et al., 2017]), managers will need to balance the impacts with the potential wider benefits of the intervention, i.e. the removal of AM and associated removal of N.

4.3. AM removal and its potential within NTSs

For the first time, we highlight that mechanical removal of AM from intertidal mudflats has the potential to remove significant amounts of N from coastal systems. However, most importantly we confirm that the location and time of removal are critical with a number of ecological factors contributing. Firstly, the ~6-fold difference in AM nitrogen content across season and site (e.g. a mean 1.03 % from Portsmouth in summer compared to the mean 5.7 % of Brittany's AM collected in spring) is driven by the diverse tissue composition of AM-forming species (Kostamo et al., 2020) combined with site-specific AM assemblages (Lamb, in prep; Wan et al., 2017) and seasonal changes in the concentrations of the same species (Stedt et al., 2022). Although species composition does have important consequences for the amount of N in the AM tissue, a rapid assessment of diversity at a site combined with our published values of N concentrations is likely to be sufficient to generate robust data that can be used in future scale-up assessments. The second ecological factor is the dramatic difference in AM extent across the sites and over the year. Many studies show this spatial and temporal variability to be common (Pihl et al., 1999; Aldridge and Trimmer, 2009; Wan et al., 2017; Schreyers et al., 2021) with additional within-site variability (Thornton, 2016). Several environmental factors will be important drivers, but any future scaleup assessment for a site must include extent and %AM as these have considerable influence on the N removal estimates as shown by the much higher biomass per site at Poole compared to Portsmouth. AMs are routinely monitored as part of the EU's WFD (Water Framework Directive) and other assessments (Environment Agency, 2016) so these data should be readily available. In addition, modelling and EO (earth observation) have been used (e.g. Aldridge and Trimmer, 2009; Schreyers et al., 2021); although recognising vegetation type (e.g. seagrass and AMs) in intertidal mudflats is still a significant EO challenge. The final ecological factor is AMB per area of sediment. Even though extent and %AM values in Portsmouth were similar for summer (364 Ha, mean 75 % AM) and autumn (364 Ha, mean 72 % AM) there was ~40 % more wet weight biomass present in autumn (2044 t) than summer (1354 t). These findings highlight the importance of routine monitoring of AMB that is required for WFD condition assessments.

N credit market prices in coastal systems vary considerably, influenced by national economic landscapes and regulatory requirements. They will also be dependent on the specific market scheme employed with Pigouvian PES (tax-payer financed); Coasean PES (voluntary agreements between two or more market actors) or replacement cost methods (the difference in costs between the capacity of natural systems as opposed to utilising a manufactured alternative) all changing the price (Hasselström and Gröndahl, 2021). The selection of the £25 kg^{-1} median from 75 PES schemes generates a robust value that accounts for the diverse global prices (0–2384 USD kg^{-1}) of PES and replacement cost schemes (Barrett et al., 2022), but also aligns with other coastal schemes such as Connecticut, USA (£14.1) and North Carolina (£26.9) (see references within Dvarskas et al., 2020). Our analysis confirms that using this value makes AM removal cost-effective at all sites and across all seasons except Brittany in winter. Poole and Portsmouth have medium or high cost-effectiveness across all seasons. In contrast, AM removal had low cost-effectiveness at Normandy for all seasons except spring (medium), and only removal in autumn would be highly cost-effective at Brittany. Deciding on whether AM removal should be implemented at the four sites and which seasons are best can, therefore, be based on these data. However, we would caution that those site/season combinations with only high or medium cost-effectiveness are targeted. Not only is this because of the known site-specific inter-annual variation in AMB (Capuzzo and Foster, 2014), but additional costs of removal may need to be included. For example, licensing costs are country and activity-specific, and even though low, medium and high harvest efficiencies are included in the analysis based on the site-specific %AM, different site conditions (vessel entry points and tides etc.) in relation to AM collection will impact costs and harvest efficiencies. Focussing on those sites/seasons with medium or high cost-effectiveness should give a sufficient buffer for uncertainties in credit price and harvesting costs.

Predictably, a considerable increase in profitability of AM removal can be achieved with a different N credit price, for example the £3000 of the Solent Nutrient Trading Pilot land-use NBS (Nature-based Solution). Using this price makes all site/season combinations highly cost-effective; achieving several million pounds per site. The Solent Nutrient Trading Pilot land-use NBS was developed by councils, wildlife NGOs and private companies to gain planning permission for housing development (Natural England, 2021). As increased urbanisation (i.e. new housing), and subsequent growth of inhabitants has led to an increase in the nutrient loading (Ullah et al., 2018), the statutory advisor to the English Government directed local councils to assess developments on a case by case basis to mitigate the impacts of developments on nitrate levels (Natural England, 2020). These actions led to a 60 % reduction in housing development in the designated areas (Williams and Eve, 2023), and affected approximately 10 % of English planning authorities. While all the current NBSs available within the scheme are terrestrially-based, the considerable cost-effectiveness shown here of AM removal should stimulate urgent discussion about inclusion of AM removal. Nevertheless, as the £3000 N-credit price is set to be equivalent to 1 kg N yr^{-1} for the lifetime of the development, how AM removal cost-effectiveness is annualised over 80–125 years (Natural England, 2020) is an important consideration.

While terrestrial nutrient trading schemes have a well-established regulatory framework and infrastructure, marine NTSs are relatively new (Hughes, 2021). Potential challenges of marine NTSs include individual water body complexity within a larger catchment, ownership and allocation of credits, water bodies crossing multiple jurisdictions and a need for political support rather than relying on spontaneous market-based initiatives (Hasselström and Gröndahl, 2021; Thomas et al., 2021; Filippelli et al., 2022). Despite these challenges, blue nitrogen is now regularly being proposed in the context of ecosystem services and natural capital valuations underpinning investment in multiple NBSs (Gregg, 2015; Cornwell et al., 2016; Thomas et al., 2021; Kotta et al., 2020; Barrett et al., 2022). The 66–95 kg N ha^{-1} yr^{-1} removed from Poole (all seasons) and the 44 kg N ha^{-1} yr^{-1} Brittany (autumn) align with other NBSs e.g. clams (a mean including 95 % CI of 107-3-477 for t N ha^{-1} yr^{-1}) and scallops (52 t N ha^{-1} yr^{-1}). Although AM removal values are lower than seaweed (275 ± 96–678 t N ha^{-1} yr^{-1}); mussel (581 ± 275–1172 t N ha^{-1} yr^{-1}) and oyster (314 ± 150–612 t N ha^{-1} yr^{-1}) aquaculture, the potential for multiple AM harvests in one year would change these comparisons (Barrett et al., 2022). The large CIs surrounding these means also indicate site/farm/species specific factors are important so a comparison of AM removal with NBSs (e.g. aquaculture [seaweed, bivalve]; habitat restoration (biogenic reef, saltmarsh, seagrass) from the same location (catchment) would be a critical step to assess the best blue nitrogen solution. These should also be compared to utilising replacement costs for a manufactured alternative (e.g. sewage treatment work improvements).

4.4. Post harvesting value-added

The price of an NTSs credit is limited to the service that it is assigned to. However, there may also be significant additional value if a usable product contributes to the blue economy. With aquaculture, generally biomass provisioning is the main motivation for harvesting, with regulating (e.g. N bioremediation) services as side effects. While our valuations of AM removal are predicated on N bioremediation within an NBS framework, reviews confirm that the AM biomass can have substantial value post-harvest (Sarwer et al., 2022). The downstream use of AM will firstly determine any pre-treatment, which could generate additional costs and logistical challenges (Bruton et al., 2009). Screening or washing to remove foreign objects/sand may be necessary if the AM is to be chopped/milled. Squeezing to reduce the water content allows for stabilization of the biomass as well facilitating transport and reducing drying costs if required. The AM may also need to be desalinated prior to anaerobic digestion, alternatively it would have to be dried for human or animal feed. Despite these additional steps, downstream processing can add considerable value to AM removal. Numerous studies (see review by Sharmila et al., 2021) have explored the viability of macroalgae for biofuel production, with *Ulva*, (Abomohra et al., 2018) and *Chaetomorpha* (Kraan, 2013) proving promising genera. Relying on 'problem' AMs rather than competing with seaweed aquaculture also generates a sizable food security argument. Other products and uses of green macroalgae include conversion to animal feed, paper, compost, human food and even high value molecules such as ulvans; with the latter increasing the value per kilogram by several orders of magnitude (Capuzzo and Foster, 2014; Lakshmi et al., 2020). The post-harvest use of AM, therefore, provides significant potential to diversify value beyond an NTS thus generating more robust value-chains and markets.

5. Conclusion

The planetary boundaries of nutrient biogeochemical flows have been exceeded generating significant eutrophication in numerous coastal systems. We show that AM removal from intertidal mudflats has the potential to remove considerable quantities of N with standardised amounts (e.g. by hectare) similar to biomass removal of some non-fed aquaculture species. While the impacts of the AM removal process at scale on the benthos and bird communities are yet to be resolved, our data confirm the requirement for site and season-specific baseline information (spatial extent, %AM and AMB) to underpin decision-making. Within the Channel region, Poole and Portsmouth are the most appropriate of the four sites for trials using a mechanical conveyor belt system. For AM removal at scale to be cost-effective requires the development of accredited blue nitrogen NTSs or inclusion of AM removal and other NBSs in terrestrial-based solutions. These must have a robust trading infrastructure, broad stakeholder support and implemented at the catchment-level or higher. Nevertheless, with substantial post-harvest value for AM as a marine biomass resource we propose that AM removal can contribute to the blue-green bioeconomy revolution outside of an NTS framework. The blue nitrogen credit value, ease of implementation and link to a direct impact from the AM are also likely to make it much more attractive than blue carbon or biodiversity credits.

AM removal has potential to improve water quality and rapidly reduce the many impacts of AMs at the local scale, thus contributing to a waterbody achieving good environmental status. AM removal has the potential to improve water quality and rapidly reduce many of the local-scale impacts of AMs, contributing to a waterbody achieving good environmental status. It can also help mitigate some of the ecological impacts of predicted human population growth within a catchment. For example, an additional 15 t of N yr^{-1} will enter the Poole catchment from a forecast population growth of 17,490 people compared to the 6.6–9.5 t of N removed with the AM. However, to address the wider catchment nutrient loads in coastal areas, e.g. ~2300 t N yr^{-1} in Poole catchment (Bryan et al., 2021), it must be considered as part of a multifaceted management process with a focus on reducing point and diffuse sources. While the necessary actions are well known, many require significant investment with timescales to deliver results spanning decades, by contrast rapidly implemented AM removal can generate immediate results and within a blue nitrogen NTS could be highly cost-effective.

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CRediT authorship contribution statement

Zoe Morrall: Writing – original draft, Project administration, Methodology, Investigation, Formal analysis, Data curation. **Joanne Preston:** Writing – review & editing. **Sophie Richier:** Writing – review & editing, Methodology. **Daniel J. Franklin:** Writing – review & editing, Methodology. **Annesia Lamb:** Writing – review & editing, Methodology. **Andrew Van Der Schatte Olivier:** Writing – review & editing, Investigation. **Eric Harris-Scott:** Writing – review & editing, Methodology. **Dominic Parry:** Writing – review & editing, Methodology. **Graham Horton:** Writing – review & editing. **Stephanie Lemesle:** Writing – review & editing, Methodology. **Claire Hellio:** Writing – review & editing. **Marilyn Fauchon:** Writing – review & editing. **Gordon Watson:** Writing – original draft, Funding acquisition, Conceptualization.

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Declaration of competing interest

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Data availability

Data will be made available on request.

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