Small-scale field trials identify optimal habitat restoration options on exclosed nuclear sites

Braidwood, David W.; Taggart, Mark A.; Smith, Melanie; Morgan, Graeme; Andersen, Roxane

Published in:
Ecological Engineering

Publication date:
2020

Citation for published version (APA):
Braidwood, D. W., Taggart, M. A., Smith, M., Morgan, G., & Andersen, R. (2020). Small-scale field trials identify optimal habitat restoration options on exclosed nuclear sites. *Ecological Engineering*, *156*, 1-14. Article 105976. https://doi.org/10.1016/j.ecoleng.2020.105976

Title

Small-scale field trials identify optimal habitat restoration options on exclosed nuclear sites

Author names and affiliations

David W.; Braidwood[a]; Mark A. Taggart[a]; Melanie Smith[b]; Graeme Morgan[c]; Roxane Andersen[a]

[a] Environmental Research Institute, University of the Highlands and Islands, Castle Street, Thurso, Scotland, KW14 7JD

[b] Inverness College, University of the Highlands and Islands, 1 Inverness Campus, Inverness, Scotland, IV2 5NA

[c] Dounreay Site Restoration Limited, Dounreay, Thurso, Caithness, Scotland, KW14 7TZ

david.braidwood@uhi.ac.uk *Corresponding author; mark.taggart@uhi.ac.uk; melanie.smith.ic@uhi.ac.uk; graeme.morgan@dounreay.com; roxane.andersen@uhi.ac.uk

CRediT roles

David Braidwood: methodology, formal analysis, investigation, resources, data curation, writing – original draft, writing – review and editing, visualisation, validation **Roxane Andersen**: funding acquisition, project administration, supervision, validation, conceptualization, methodology, writing – review and editing **Mark Taggart**: supervision, conceptualization, methodology, writing – review and editing **Melanie Smith**: supervision, writing – review and editing **Graeme Morgan**: conceptualization project administration, supervision, resources, writing – review and editing

Abstract

Degradation of cliff-top habitats due to industrial activity is infrequent, leading to unique challenges when activity ceases and a site requires remediation. Dounreay, an ex-nuclear power facility on the north coast of Scotland, is currently being decommissioned prior to complete demolition of all buildings. A small area of the site will then be covered in a 1000 mm capping layer to prevent contamination of bioreceptors at the surface from below ground contaminants. Topsoil replacement is frequently used in such instances, however, limited availability of topsoil in the region means that other materials are sought to mitigate against undesirable ecological and economic costs. We tested combinations of differentially graded rocks and topsoil to assess their suitability in supporting re-vegetation. The vegetation response was measured as richness (numbers of species), ground cover, species diversity, and biomass. Wider ecosystem function was assessed by measuring invertebrate numbers and diversity, and comparing these to locally relevant reference site data. Fine crushed rock supported native vegetation establishment and growth, though cover, diversity, and biomass (after three years) were significantly below levels found when topsoil was used instead. Initial establishment on the fine crushed rock may be limited by the low water retention and nutrient availability of the material. We suggest that a rich heterogeneous habitat able to support a wide range of native vegetation and invertebrates may be provided by mixing topsoil with the fine crushed rock in the future. This mosaic combination would also allow for significant ecological and financial cost savings (against topsoil only remediation) and could be tested on larger scales.

Highlights

Vegetation returns on crushed rock surface layer without topsoil addition

Invertebrate numbers recover well on crushed rock surface vegetation

Opportunist colonisers generally outcompeted by targeted sown vegetation

Phosphate addition could speed up vegetation recovery on crushed rock surface layer

Remediation using fine crushed rock could reduce financial and carbon costs

Keywords

Remediation; Capping design; Dounreay; Vegetation; Invertebrates; Fine crushed rock; Topsoil

1. Introduction

Cliff-tops exposed to ocean spray provide large areas of high conservation value habitat around the coastlines of northern and western Europe (and beyond). Much of the coastline of northern Scotland is formed of high sea cliffs, where habitat is often protected and designated (i.e., as 'Sites of Special Scientific Interest' (SSSI)), representing often rare natural heritage (Scottish Natural Heritage, 2017). In Caithness (the regional focus for this study), 76% of the coastline is made up of sea cliffs, ranging from 5m to 90m high (European Commission, 2004). Levels of exposure (including wind strength and direction, and cliff height) partly dictate the vegetation found here. This vegetation is often either grassland or heathland (Malloch, 1972), or is composed largely of species resilient to these habitat types (Sawtschuk et al., 2012).

Anthropogenic activity has led to many changes along these cliff-tops, through the intensification of agriculture, pressures from habitation and tourism, and via the siting of industry (e.g. Sawtschuk et al., 2012; Shiflett et al., 2013). More recently, restoration and remediation to cliff-tops has occurred to prevent further loss of habitat, species and ecosystem function. Both passive restoration (i.e. the exclusion of environmental stressors) and active restoration (i.e. land management intervention) have been used. Both have their benefits, though it generally takes longer to achieve desired goals with passive restoration (Zahawi et al., 2014). Passive restoration generally works better on small sites surrounded by desired natural vegetation (Prach and Pyšek, 2001), and where environmental conditions are not too extreme (Prach and Hobbs, 2008). Where there are significant constraints to passive restoration succeeding, a sliding scale of restorative activities will be required.

The Dounreay Nuclear Power Development Establishment in Caithness (henceforth called Dounreay) is sited on the north coast of Scotland and is one example of a degraded cliff-top industrial site where intensive restorative activities are necessary and underway. As well as encompassing large areas of industrial infrastructure, Dounreay is bordered by the sea, the Vulcan Naval Research Test Establishment, and agricultural land. Thus, there is limited scope for natural dispersal of desired vegetation onto the site. Commissioned in 1955 (on the site of a WWII airfield), the site hosted three test reactors and other support facilities. Reactor operations ceased in 1994 and the focus has been on site decommissioning since 2000. Following infrastructure demolition and site remediation (including capping where required), an area of the site (<1 ha wherein residual low-level radioactivity may be present) will be exclosed to the public and receive no on-going maintenance – it is this small area that is discussed below. The current aim of the decommissioning process is to reach this exclosed ‘interim end state’ (IES) by 2033, and a final end state (FES) by 2333, at which point the site will be released back to public access/use. To achieve the IES, native vegetation must be returned, whilst having minimal environmental impact (i.e., on donor sites where materials may be extracted from), minimise carbon cost (with respect to both transportation of materials and processing of those materials for remediation), and the process should also maintain or enhance levels of biodiversity in the area. The intention is to also achieve this with minimal financial expenditure which may be achievable using different remedial methods.

Methods such as "soil-mixing" (i.e., combination of topsoil and crushed rock) may be used on site, which have been shown to reduce the cost of remediating contaminated soils, successfully enhancing recovery of soil biological activity and available nutrients (O'Brien et al., 2017), and promoting increased vegetation cover (Merino-Martín et al., 2017). Nevertheless, the effectiveness of this approach in restoring vegetation communities and providing a functional ecosystem, remains largely untested. In other scenarios (e.g., at mining sites (Borůvka et al., 2012), or for oil contamination remediation (O'Brien et al., 2017)) topsoil replacement is commonly used to restore degraded land, as this facilitates the return of vegetation through the provision of a readymade soil seed bank, accessible nutrients and organic matter (Borůvka et al., 2012). Topsoil depths of up to 1000 mm have been added to contaminated sites to create an instant restoration layer (Maiti and Maiti, 2015), though successful restoration can occur with a much shallower topsoil depth (i.e., of 150 mm (Paschke, 2008)).

A major challenge near Dounreay is limited availability of topsoil for remediation (with local soils often being <300 mm deep). To establish a capping layer involving minimal topsoil importation, we investigated if locally available materials/combinations could promote the return of local cliff-top vegetation. Further remedial constraints were also specified by Dounreay: (i) the site should maintain or enhance local biodiversity and provide suitable habitat for micro and meso-fauna in the area, delivering multiple ecosystem functions/benefits; (ii) there should be no need for on-going maintenance of the capping vegetation over the coming decades to centuries, while the site is in its IES; (iii) this would be possible under current remediation and capping design plans, which assumes a capping layer of 1000 mm (in depth).

Understanding how substrate/soil geochemistry influences initial establishment and growth of vegetation over the short-term (< 3 years), will help to inform longer term strategies. Similarly, an improved knowledge of local ecological processes (that may affect succession trajectories) will increase the likelihood of reaching desired results. Controlled assessment and evaluation of method suitability to initiate recovery, and facilitate the desired remediation trajectory can be achieved through small-scale trials in the first instance (Sawtschuk et al., 2012), and was the focus of this study.

More precisely, the aims of the study were to test if topsoil is required to re-establish vegetation growth, and whether different grading of material – on both the surface and lower down - affected vegetation communities (species numbers, diversity, cover, and biomass). Furthermore, we aimed to determine what combination of treatments favoured the return of functionally diverse invertebrate communities.

2. Materials and Methods

2.1 Vegetation trial plot set up

An experimental area for *in situ* trial plots was set up 200 m SE of the eastern most corner of the Dounreay site licence area, which sits 58.58°N, 3.75°W (Figure 1). The range of climate variables are: mean summer/winter daytime temperatures 13°C/4°C (respectively); mean monthly rainfall in summer/winter 38 mm/84 mm (respectively); and daylight in midsummer/midwinter, 18.3 hours/6.3 hours (respectively) (Met Office, 2013).

Using a full factorial design, eight treatments were tested (using plots), trialling combinations of two substrates (coarse or fine crushed rock), two top layers (topsoil or fine crushed rock), and two seed mixes (grassland and coastal). Six replicate blocks, measuring 12 m x 6 m x 1 m deep, were constructed and each block contained the eight treatments, which were randomly assigned to 3m x 3m plots within the block (see Figure 9 & Figure 10 in appendix). The top surface of the trial plots was 0.7 m above the surrounding ground level, following excavation down to bedrock (i.e., also 0.3 m below ground level), thus replicating a 1000mm deep total capping layer. An unseeded crushed rock surface layer of a recommended 2 m distance was left between adjacent replicate blocks (Smith et al. 2014).

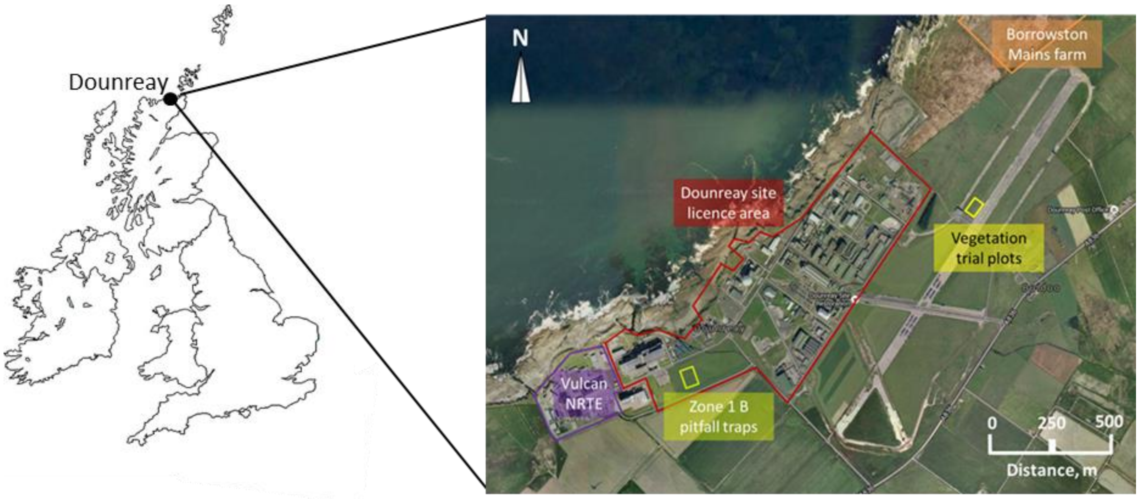

Figure 1. Location of Dounreay on a map of the UK and Ireland (left) and a Google Earth™ image showing how the site sits by the coast (right). Areas of note are highlighted on the map with colour linking to location.

As part of plot construction, vegetation was cleared and topsoil salvaged and stockpiled next to the site (for two weeks, prior to re-use on the trials). Across the whole trial area, coarse rock was placed to a depth of 300 mm. Gabion baskets (3000 mm long x 1000 mm wide x 700 mm high/deep, three for each 3 m x 3 m replicate) were then filled first with coarse

(90mm mesh) or fine (30 mm mesh) rock to ca. 100 mm from the top (when coarse used) or ca. 50 mm from the top (when fine used). On top of the coarse rock a layer of fine rock ca. 50 mm deep was added. The treatments were then topped with a further 50 mm of fine crushed rock, or, with 50 mm of topsoil. Once completed, treatments were seeded (in July 2014) with one of two amended seed mixes from Scotia seeds (http://www.scotiaseeds.co.uk/), the MG5 (UK national vegetation classification (NVC) mesotrophic grassland) seed mix or the coastal mix (see Table 2 in appendix for species sown) at a density of 3 g m^{-2}. All changes made to the standard seed mixes were to remove species not present in Caithness (Vice County 109), and any additions made were based upon recommendations by staff at Scottish Natural Heritage (SNH), Plantlife, and the Royal Botanic Garden Edinburgh (RBGE). Batters, using remaining topsoil, were also constructed around the outer edges of all trial plots to aid stability and help prevent desiccation at plot edges (these were not seeded).

2.2. Vegetation data collection from trial plots

Vegetation data were collected from the central 1 m x 1 m of each plot, using the Braun-Blanquet method (Poore, 1955) to record cover, with individuals identified to species level where possible. Data were recorded weekly for the first seven weeks (from September 2014), and subsequently every four weeks (between October and March), or fortnightly (between April and September), until the end of October 2017, amounting to 58 data sets.

During the second growing season the flowering heights for *Rhinanthus minor* L. (Yellow Rattle) were measured, and at the end of third growing season one half of each plot had the vegetation cut to ground level. The fresh material was weighed, and then dried, and the dry mass weighed, allowing for any differences in above-ground biomass between treatments to be recorded. These acted as simple measurements to test as a proxy for plant fitness (Younginger et al., 2017).

2.3. Vegetation data collection from reference sites

Nine reference sites were surveyed across the north coast of Scotland (see Figure 2), seven during July and August 2014, and two during July 2015. At each site, up to ten quadrats (2 m x 2 m) were surveyed along each of three transects perpendicular to the coast, 100 m apart. The first quadrat was placed as near to the cliff-top as possible, then the transect progressed inland, with the second quadrat placed when a change in vegetation was observed – or, at a maximum 30 m distance from the first one. These quadrats were randomly placed 2 m either side of the transect. Beyond the second quadrat, the maximum distance between quadrats rose to 50 m, and beyond the fifth up to 100 m. Where there was a farmed area within 500 m inland from the coast, fewer than ten quadrats were recorded. Vegetation data were again collected using the Braun-Blanquet method to record cover, with individuals identified to species where possible. The data from reference sites used here are those that were classified as not being peat soils – i.e. they had under 50% organic material in them.

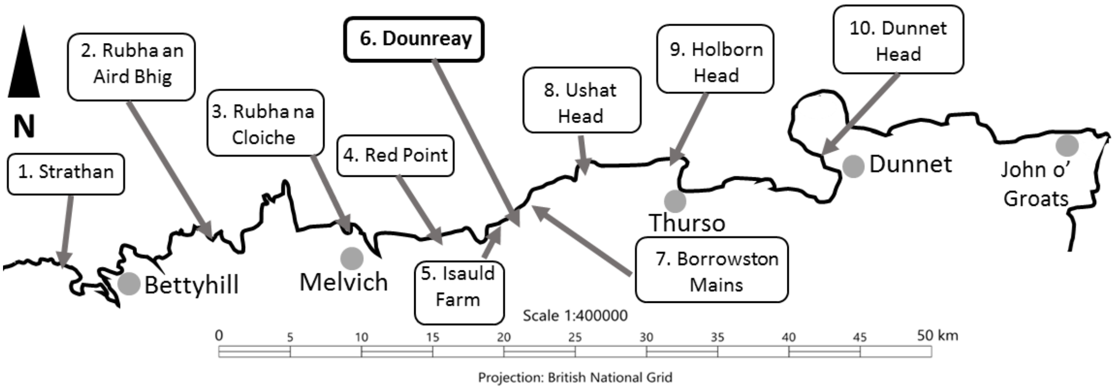

Figure 2. Outline map showing locations of the reference sites across the north coast of Scotland and main settlements.

2.4. Invertebrate data collection from trial plots

Pitfall traps were placed just outside the north-east corner of each 1 m x 1 m square plot from which vegetation was recorded. The trap consisted of a 0.5 l plastic beaker sunk into the ground so that the rim of the beaker was level with the ground (Barber, 1931). Propylene glycol (for preservation) was poured into the beaker to a depth of ~50 mm, chicken wire added to allow any small mammals to escape, and a wooden lid placed over the top of the beaker. The traps were emptied every 2 weeks, with a total of 12 collections made over two years (July – September 2015 and April – July 2016); thus providing a non-continuous time set regarding one year's data for analysis (Yanahan and Taylor, 2014). Once collected, specimens were stored in a freezer until they were identified down to the level of taxonomic order. This allowed for meaningful comparisons between treatments, while requiring less identification time (Biaggini et al., 2007). Using the same method and collection dates, three sets of four traps (set at each corner of a 1 m square), were also set up on an area of restored grassland within the site licence area, zone 1B (Figure 1), and four sets of traps were set up at Borrowston Mains farm reference site (Figure 1; three sets >65 m from the sea and one set <65 m from the sea).

2.5. Soil analysis

A sample of soil or fine crushed rock from the surface of all replicates (n=48) was collected immediately following plot construction and again at the end of September 2016. Soil samples were also collected from each of the reference site quadrats. Soil samples were analysed for volumetric water content, organic matter content (measured as loss on ignition [LOI] 450° C for 8 hours), pH (Jenway 3345 pH meter), conductivity (HANNA® HI991300 meter), total oxidised nitrogen (TON) (sum of nitrate and nitrite; using Seal Analytical AQ2 discrete analyser), and the following macro-, micro-, and potentially toxic elements: P, K, Ca, Mg, S, Cu and Pb (ICP-OES; Varian 720 ES with SP3 autosampler).

Extractions used prior to ICP-OES analysis targeted three "phases": Mehlich 3 extractable, water soluble, and acid digestable phases (using concentrated nitric acid, HNO_3). Mehlich 3 extractions provide an estimate of bioavailable nutrients; and the methods outlined in Mehlich (1984) and Zhang et al. (2014) were used. Water soluble extracts utilised the supernatant solution derived from a 1:3 soil:Milli-Q® water suspension, which was shaken for 30 minutes at 250 rpm, followed by centrifuging for 20 minutes (Thermo Scientific IEC CL30R) at 2500 rpm (this method was also used to prepare samples for measuring TON, pH, and conductivity). Acid digestions utilised 0.3 g of oven dried material (105° for 24 hours) digested using 3 ml of trace metal grade HNO_3, and left at room temperature for 48 hours. These were then heated to 140°C for one hour, 0.5 ml of H_2O_2 was added, and heated to 140°C for a further hour. A 0.2 ml aliquot of H_2O_2 was added, then placed in a UV digester (705 UV Digester Metrohm, Switzwerland) for 10 minutes. This was repeated until a total of 1 ml H_2O_2 had been added. Recovery rates for the CRM was > 95% for Ca, K, P, Pb, and S; Cu recovery was 93% and Mg was 88%.

2.6. Statistical analysis

Statistical analyses of the data were performed using R version 3.3.1 (R Core Team, 2016) in RStudio (Version 0.99.903 (RStudio Team, 2015)).

2.6.1. Vegetation analysis

The non-numeric data points generated by the Braun-Blanquet scale, 'r' (for a single individual) and '+' (for a few individuals), were converted to numeric values of '0.1' and '0.5' respectively prior to analysis (Lepš and Smilauer, 2003), while other values remained at '1', '2'. '3', '4', or '5'. Richness (total species numbers), diversity (Shannon's *H*), and cover (sum of Braun-Blanquet relevés) were calculated using data from the 2016 growing season as there were more data sets (13) than for 2017 (9). Means were calculated for each replicate of each treatment, using ‘specnumber’ (richness), ‘diversity’ (Shannon’s *H*), and ‘rowsums’ (cover), using the *vegan* package (Oksanen et al., 2012). Differences between treatments were then tested using ANOVAs (‘lsmeans’ function ‘aov’ (Lenth, 2016)). Community assemblages were compared across treatments using permutational multivariate ANOVA and the ‘adonis’ function in ‘*vegan*’ after the data had been transformed using the ‘hellinger’ method and the ‘decostand’ function (also in ‘*vegan*’), thus giving less weight to matching ‘0s’.

Vegetation composition changes over time were compared using principal response curves (PRC). This multivariate method is a form of redundancy analysis, allowing temporal changes within the data to be compared to a selected reference (Van den Brink and Ter Braak, 1998; Van den Brink and Ter Braak, 1999), and providing a way to specifically test the interaction between time and treatment (displayed as a deviation from a reference set *a priori*). The analysis also highlights which of the response variables (i.e., plant species) are more strongly related to the temporal trend, with the difference in trajectories tested statistically using Monte Carlo tests. Data from all 58 collection dates were used, though only species present in ≥ 10% of all records were used (Poulin et al., 2013). Two PRCs were conducted, with treatments separated by seed mix, with the no topsoil + coarse subsoil treatments (i.e., least intervention) used as the reference to test other treatments against.

Differences between treatments in above ground biomass (fresh and dry weight) were tested using a three-way ANOVA.

2.6.2. Invertebrate analysis

Numbers and diversity of invertebrates were compared using ANOVAs, more specifically using the ‘lsmeans’ function ‘aov’ (Lenth, 2016). One order, the Amphipoda (sandhoppers), were omitted from the analysis as they appeared in very large numbers (often >100 per trap) at just one set of replicate traps near the cliff top at Borrowston Mains farm. Taxonomic orders were subdivided into four functional groups (carnivores, herbivores, detritivores, and generalists) according to Tilling (2014) to assess whether a functionally diverse range of invertebrates was present within the different treatments.

2.6.3. Soil analysis

Principal component analysis (PCA) was carried out to consider how vegetation trial plot treatments compared to each other, and to the soils from the reference sites, according to the attributes measured. ANOVAs were used to test between the sites/treatments for the principal components that explained >10% of the variance. ANOVAs were used to compare changes in soil chemistry from the start to the end of the trials.

3. Results

3.1. Vegetation

3.1.1. Cover, richness and diversity

Overall, vegetation cover, species' richness, and species' diversity were consistently and significantly higher in treatments with topsoil when compared to those with fine crushed rock during the growing seasons (Table 1, Figure 3). The MG5/fine crushed rock/coarse subsoil treatment supported fewer species and had lower cover than all the others, with the MG5/topsoil/coarse subsoil treatment having the most species and greatest cover (Table 1).

Table 1. Summary of richness, cover, and diversity of vegetation from the trial plots.

Treatment	Seed mix	C	MG5	C	MG5	C	MG5	C	MG5
	Surface layer	Fine rock	Fine rock	Fine rock	Fine rock	Topsoil	Topsoil	Topsoil	Topsoil
	Subsoil	Coarse	Coarse	Fine	Fine	Coarse	Coarse	Fine	Fine
	Mean number of species - Richness								
Time period	Summer 2014	3.87	4.00	5.03	4.60	12.40	12.10	12.60	12.87
	Winter 2014/2015	4.79	4.25	5.44	5.42	15.13	15.40	14.86	14.21
	Summer 2015	6.22	5.17	7.05	6.55	17.47	16.81	17.07	16.88
	Winter 2015/2016	7.08	5.81	7.83	7.50	13.42	14.04	13.06	13.35
	Summer 2016	11.14	9.86	12.41	14.11	16.35	17.77	16.73	16.98
	Winter 2016/2017	7.23	6.23	8.77	8.43	9.93	12.50	9.60	12.30
	Summer 2017	8.67	7.60	10.50	10.29	14.60	16.63	13.92	16.13
	Throughout study	**7.51**	**6.39**	**8.45**	**8.54**	**14.80**	**15.60**	**14.64**	**15.14**
	% increase in spp. no.	124.14	90.10	108.61	123.73	17.78	37.40	10.45	25.32
	Braun Blanquet amended cover total								
Time period	Summer 2014	1.86	2.22	2.58	2.72	7.82	8.06	8.16	8.67
	Winter 2014/2015	2.65	2.55	3.24	3.63	11.56	11.55	11.59	11.32
	Summer 2015	3.63	3.32	4.36	3.98	17.78	17.43	16.11	17.57
	Winter 2015/2016	5.46	4.63	6.26	5.70	15.83	16.78	15.10	15.31
	Summer 2016	9.43	7.28	10.51	10.78	19.59	22.21	19.30	20.73
	Winter 2016/2017	6.95	6.07	7.87	7.35	12.13	14.69	11.77	14.64
	Summer 2017	8.95	7.55	11.03	9.31	18.95	21.62	18.52	22.01
	Throughout study	**5.95**	**4.93**	**6.79**	**6.46**	**15.74**	**17.06**	**15.27**	**16.55**
	% increase in cover	380.96	240.00	328.04	241.88	142.48	168.27	126.88	153.99
	Diversity								
	Throughout study	**1.74**	**1.55**	**1.88**	**1.82**	**2.50**	**2.55**	**2.49**	**2.52**

Treatments with topsoil had a significantly higher level of richness ($F = 108.87, p < 0.001$), cover ($F = 416.23, p < 0.001$), and diversity ($F = 71.71, p < 0.001$), than those with fine crushed rock. Topsoil and seed mix had a significant impact on the composition of species found on the treatments ($F = 39.51, p < 0.001$; $F = 27.83, p < 0.001$ respectively). The subsoil layer also had a significant difference (though at an increased α level of 0.05) on diversity ($F = 5.77, p = 0.021$) and richness ($F = 4.60, p = 0.038$).

The number of species, level of cover, and diversity, all showed a seasonal trend. Over the course of the trial there was also an upward (improving) trend for those variables within the fine crushed rock treatments and can be seen in (Figure 3). The levels of richness, cover, and diversity of treatments that were cut remained slightly lower than those of the same uncut treatments, during the following 12 months, and this was significantly different for the level of cover ($F = 23.70, p < 0.001$), and at a higher α level for diversity ($F = 6.27, p = 0.014$).

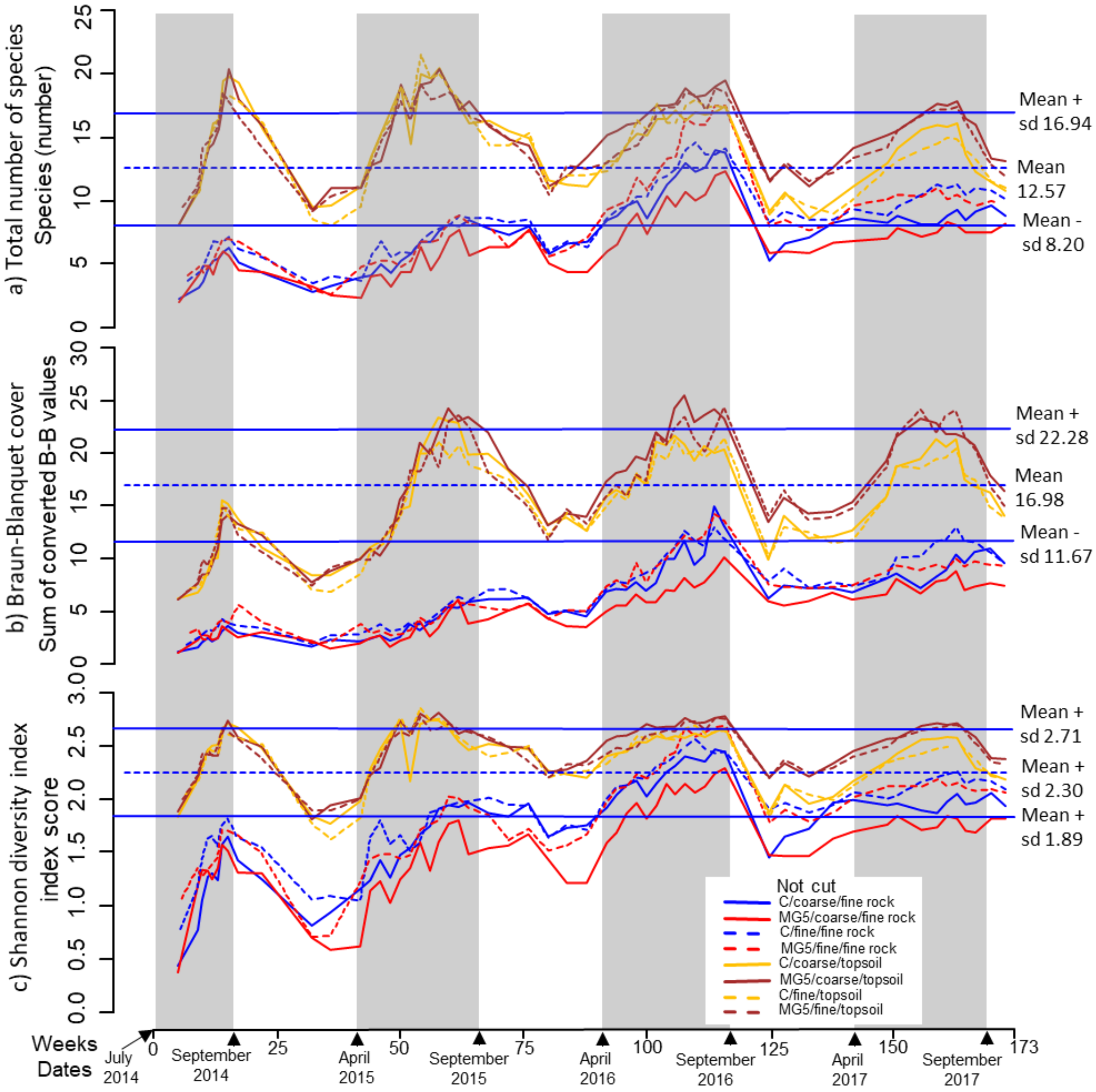

Figure 3. Changes in vegetation during the time-course of the vegetation trials. Panel 'a' shows the change in the total number of species recorded on the plots; panel 'b' shows changes in ground cover; and panel 'c' shows changes in diversity. The sections of the plots highlighted vertically show the 'growing season(s)', 1 April to 30 September. All plots with topsoil had a greater number of species, more ground cover, and a higher diversity index score than those with fine rock cover, until the end of the trials. For each measurement shown the mean and mean ± standard deviation are shown as straight blue dotted and full lines, respectively, for the non-peat reference sites (for comparison). The treatments were based on a factorial design using two seed mixes ("prefix" coastal (C) and grassland (MG5)), two types of subsoil (crushed rock (Coarse) fine crushed rock (Fine)), and two cover options 'topsoil' or 'fine rock'.

3.1.2 Comparison of plant species by treatment

Seed mix and topsoil had a significant effect on vegetation composition (Table 1), with different treatments favouring the growth of the species sown onto that treatment (Figure 4). During the first three years (2014-2016) there was an increase in mean species numbers for all eight treatments, with these numbers reducing in 2017. The numbers of species on treatments that had been cut at the end of the 2016 season (identified by an asterisk [*] above the right hand column in Figure 4) remained similar to those that were uncut.

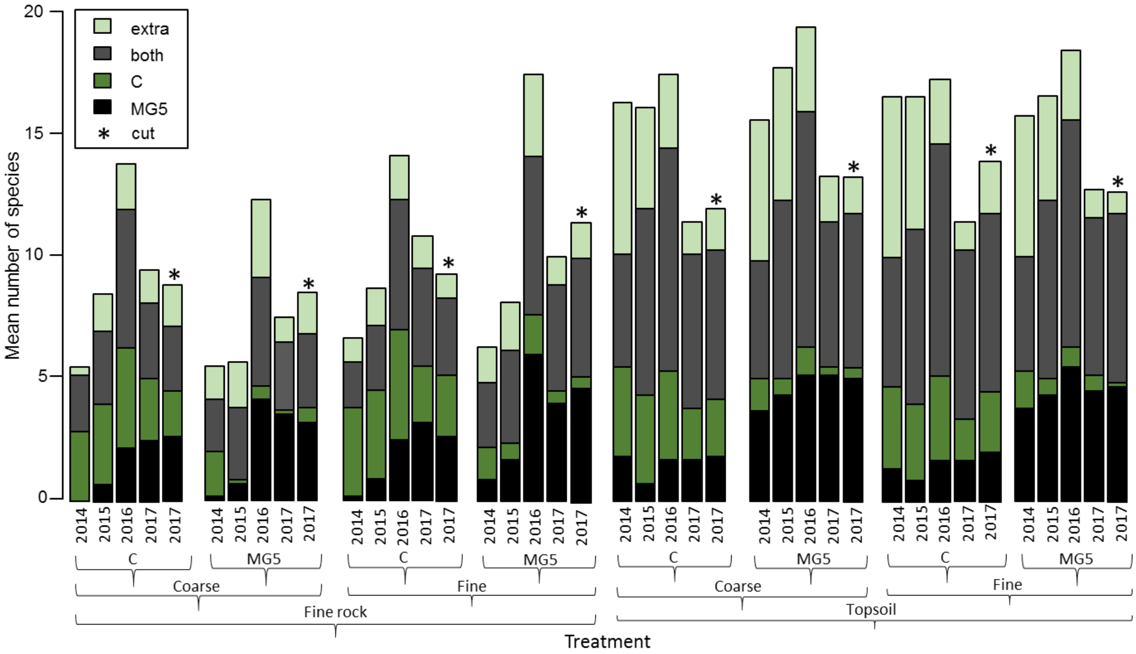

Figure 4. Number of species according to their presence or absence in the two seed mixes used. Each of the 8 treatments has been divided up into the mean number of species recorded on replicate blocks at the end of September for each of four years (2014 – 2017), and includes a final column (from 2017) at the far right of each treatment group (with an '*' above the column) for the treatments cut at the end of 2016. Species are broken down into one of 4 categories: species exclusive to MG5 seed mix; species exclusive to Coastal seed mix; species in both MG5 and Coastal mix; and species which were in neither seed mix (i.e., from soil seed bank or naturally dispersed).

The temporal pattern in vegetation cover (for both the MG5 and coastal seed mixes (Figure 5)) differed significantly to that of the reference treatment (fine rock/coarse subsoil); for

MG5: PRC axis 1 = 22.0%, *F* = 3.89, *p* = 0.001; for coastal: PRC axis 1 = 19.2%, *F* = 5.09, *p* = 0.001. During the first two years, the topsoil treatments provided more opportunity for non-sown species than treatments using fine rock (Figure 4), though by the final year of the trials, there were similar numbers of non-sown species on all treatments. For the MG5 seed mix, the topsoil treatment was favourable for *Trifolium repens* L., *Hypochaeris radicata* L., *Trifolium pratense* L., *Ranunculus repens* L., *Achillea millefolium* L., *Anthoxanthum odoratum* L., *Plantago lanceolata* L., *Holcus lanatus* L. and *Sagina procumbens* L., while the fine rock treatments were favourable for unidentified monocots (likely to have been mostly grasses), and *Cerastium fontanum* Baumg.; all of which were correlated with temporal changes in species compositi on (scores > 0.5 or <-0.5). For the Coastal seed mix, the topsoil treatment was favourable to *T.repens*, *H.lanata*, *P.lanceolata*, *R.repens*, *A.odoratum*, *Bellis perennis* L., *S.procumbens*, *Lotus corniculatus* L., while fine rock was favourable for *Silene dioica* (L.) Clairv., Primula spp., *Cochlearia danica* L. and *Silene uniflora* Roth. During the latter stages of the trials, the treatments and the trial plot reference (fine rock/coarse subsoil treatment) become more similar with both seed mixes, though during the earlier growing seasons (of those monitored), there is a greater decoupling of the topsoil treatments from the reference ones.

The species on the trial plots also differ from those that were recorded from the reference sites, though most of the species occurring in >20% of the reference sites were well represented in the species mix recorded from the trial plots (Figure 6).

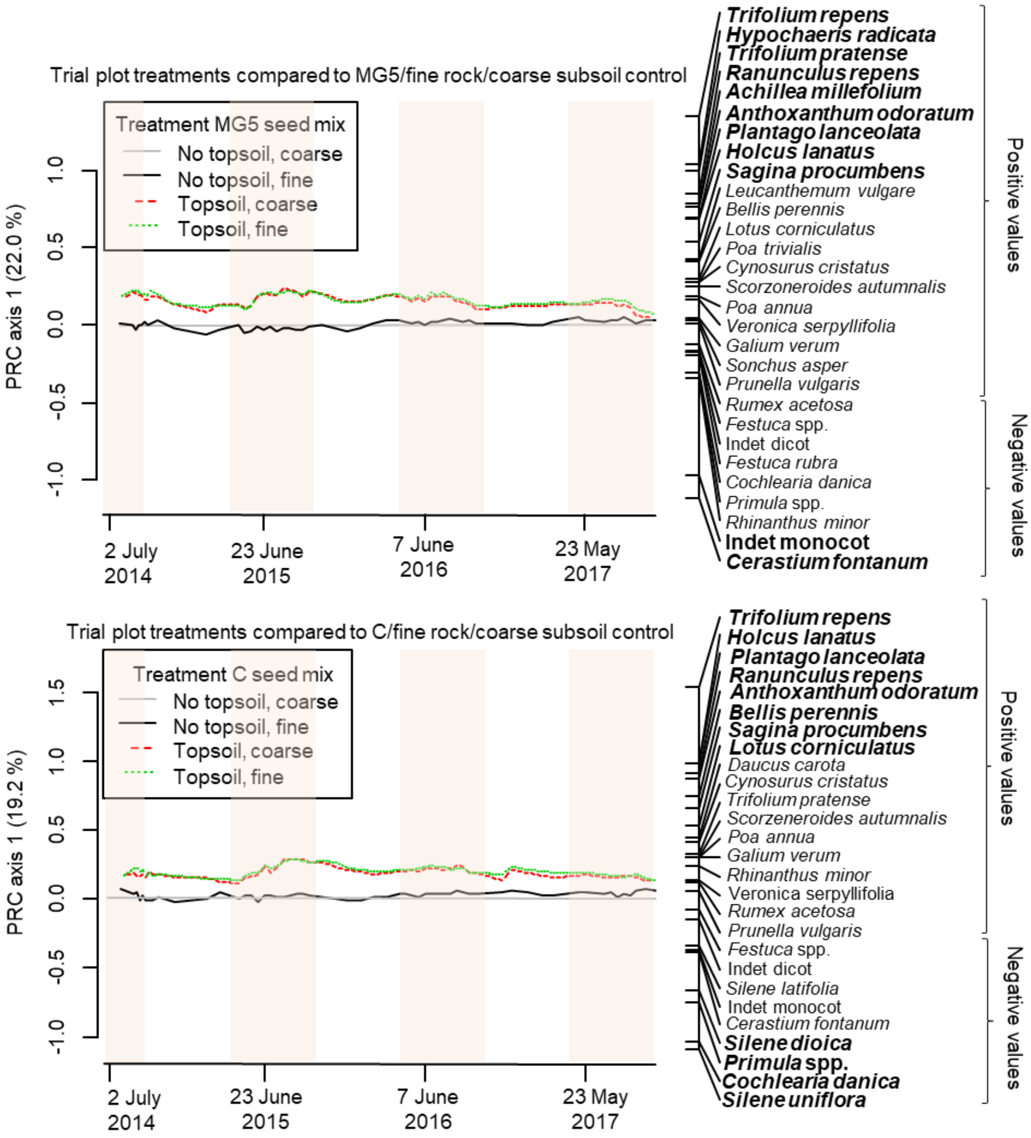

Figure 5. Principal response curves (PRC's) regarding vegetation composition with the MG5 seed mix (top) and coastal seed mix (bottom). The lines (representing the treatments) deviating from the "0" line (the fine rock/coarse subsoil reference treatment) show the behaviour of the other treatments over time. The greater the distance between the treatments and the reference line equates to a greater dissimilarity in terms of species composition. On the right are the plant species (abbreviated names) which create these differences, and their scores (representing correlations) are on the left axis. Those scores > 0.5 or < -0.5 are shown in larger typeface in bold. Species with a positive score are positively correlated with the treatments above the line, 0 = no correlation, and negative scores are negatively correlated with the treatments above the 0 line. Sections of the figure with a vertical coloured background show the growing season(s) (start of April to end of September, for each year).

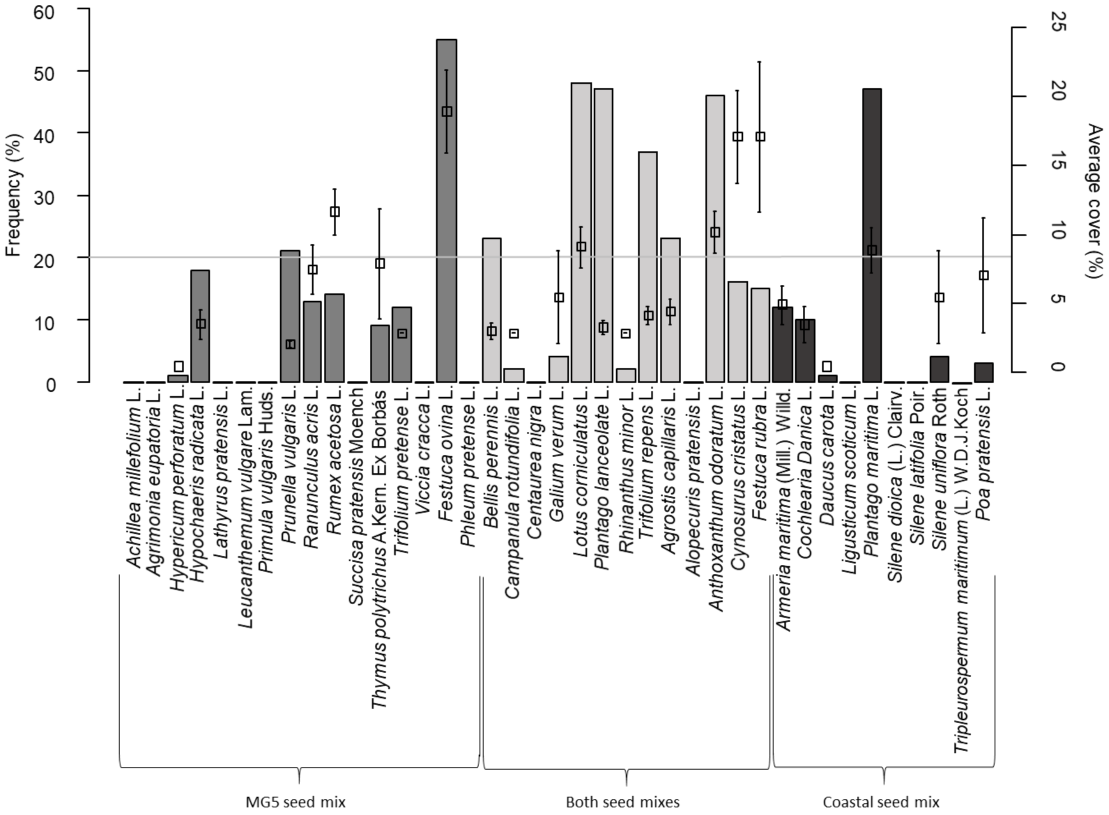

Figure 6. Frequency of occurrence (left y-axis) and cover (right y-axis) of seed mix trial plot species in non-peat reference site quadrats, grouped by seed mix. The frequency (bar chart) shows the percentage of occurrence of these species on non-peat reference sites (n = 109); the average cover is represented as a square (average) with ± SE.

The mean fresh and dry weights from the treatments with topsoil were 481.2 g and 149.5 g (respectively), compared to 72.9 g and 26.8 g (respectively) from treatments with fine crushed rock. These differences were significantly different for both fresh weight ($F = 116.3$, $p < 0.001$) and dry weight ($F = 122.2$, $p < 0.001$). The seed mix and subsoil used did not have a significant impact on the results.

R. minor grew taller on treatments with topsoil than fine crushed rock, mean heights being 320 mm and 125 mm respectively, and this difference was significant ($F = 70.99$, $p < 0.001$).

3.2 Invertebrates

On treatments with topsoil, there were significantly more individuals (mean richness = 31.9) and orders (mean = 5.9) than when there was just fine rock (mean richness = 11.5, mean orders = 4.2; F_{ind} = 43.8, $p < 0.001$ and F_{order} = 69.8, $p < 0.001$) Figure 7. The diversity of orders was greater on treatments with topsoil than on those with fine rock (F = 17.8, $p < 0.001$). Seed mix and subsoil did not have a significant effect on the number of orders.

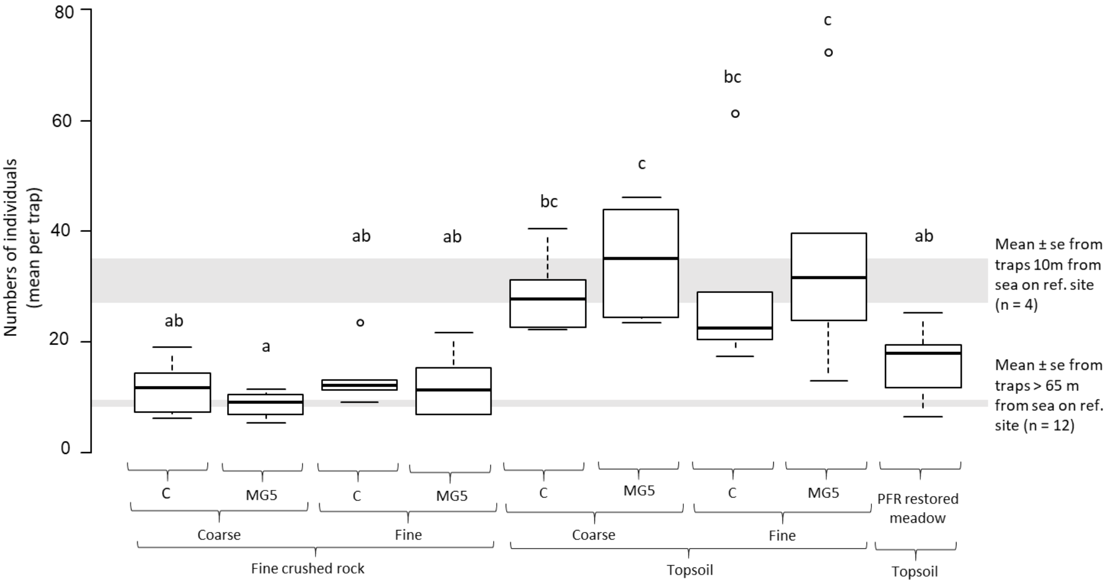

Figure 7. Boxplots of numbers of individual invertebrates recorded on the trial plots and the zone 1B restored meadow. The grey bands across the figure show the mean ± se of numbers of invertebrates collected from the reference site at Borrowston Mains.

The mean number of individuals on the reference site (at >65 m from the coast) reflected the numbers recorded on the fine rock treatments (no topsoil), while numbers from closer to the sea (at traps ca. 10 m from the cliff edge) reflected those treatments with topsoil.

Further, there were more individuals and orders recorded on the trial plot treatments with topsoil than on the previously restored meadow at Dounreay (zone 1B; F_{ind} = 6.46, $p < 0.001$ and F_{order} = 10.01, $p < 0.001$, respectively), though these values did not differ between the restored meadow and the treatments with fine rock.

3.3 Soils

Using principal component analysis (PCA), the first three principal components (PC1-3) explained 69% of the variance in the soil physico-chemical data between the soils on the reference sites and those on the trial plots, with PC1 explaining 39.2% (*F* = 211.2, $p < 0.001$) of the variance, PC2 a further 19.2% (*F* = 8.66, *p* < 0..001) (Figure 8), and PC3, 10.8% (*F* = 8.0, *p* < 0..001).

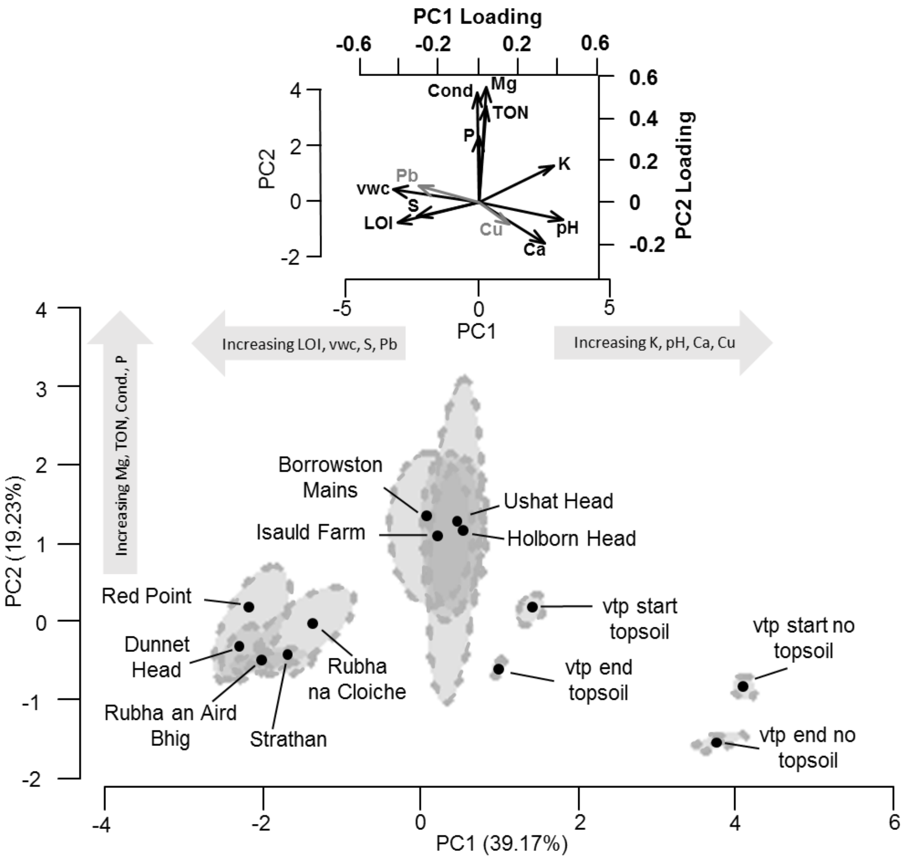

Figure 8. Principal component analysis regarding the soil chemistry data from the trial plots (divided here between those with/without topsoil – and status at the start (2014) and end of the trial (2016)) and the reference sites. The PC loadings illustrate which of the soil properties are most associated with each other, and the directions in which they are associated with the sites. For each site, the plot shows the mean and 95% (standard error confidence interval) ellipses for the soil chemistry. There are obvious groupings of sites according to soil chemistry, with the trial plots also showing changes in soil chemistry during the period of the trials.

Along PC1 (Figure 8) the main division was between sites with higher soil water, organic matter (%LOI), S and Pb (on the left of the figure), and more alkaline soils with higher Ca, K and Cu (on the right). The second axis highlighted a gradient in N, P, Mg, and conductivity, with those soils with higher concentrations represented further up the axis. Between 2014 (start of trial) and 2016, for the trial plot soil material, there was a decrease in the concentrations of P and Mg, and also reductions in levels of TON (total oxidised nitrogen) and conductivity, whilst other factors remained similar (i.e., a shift along PC2, but little change along PC1 – see right side of Figure 8).

4. Discussion

The research described here aims to seek an "environmental (and financial) cost-balance solution" to cliff-top industrial site (in this instance Dounreay) remediation; and in so doing realize a balance between successful remediation without reciprocated negative impact elsewhere (i.e., due to large scale topsoil importation).

4.1 Vegetation regenerative power

We observed higher richness and cover in treatments utilising topsoil, which may (at least in part), be a consequence of more successful germination and establishment of both species within the seed mixes and opportunistic unsown species. Notably, the latter contributed up to 49% of the total species number recorded during the first growing season. Over time, the proportion of non-sown species decreased in treatments with topsoil, but, increased on the fine crushed rock treatments, possibly because of competitive interactions relating to the nutrient deficient and xeric nature of these treatments which limited initial competitive ability (Callaway and Walker, 1997). In fine crushed rock treatments there was potentially more space for early colonisers and opportunistic species to establish, when compared to treatments with topsoil where vegetation both established and reproduced more rapidly (Ballesteros et al., 2012).

Over time diversity, richness and cover in fine rock treatments reached similar levels to that of local reference sites, albeit at a slower rate than on treatments with topsoil. A possible inference from this is that topsoil may not be required (provided sufficient time is given) to support the desired vegetation growth on the capping layer (though it accelerates initial growth). Obtaining more rapid early cover is often desirable in ecological restoration, as prerequisites may be set regarding the need for the site to meld with its surroundings quickly (Luken, 1990; Perrow and Davy, 2002; in Prach and Hobbs, 2008). At Dounreay though, in the context of needing to upscale the remediation to the whole site, fine rock as a surface layer may be a suitable strategy, given limited local topsoil availability. As organic matter (from litter) builds up on these treatments, and a richer soil begins to accumulate, a more diverse community would likely establish across the site, aiding in the creation of the more natural appearance desired. Importantly though, the final species composition also has to be right. During these field trials, the treatments all maintained a predominance of the species exclusive to the seed mix used (over that of the alternate seed mix). Desired vegetation cover was also reached not just by the successful survival of perennials (such as Silene spp., Trifolium spp., *L.vulgare*, *A.millefolium* and *P.lanceolata*), but also the successful regeneration of annuals (including *R.minor* and *H.radicata*).

A seasonal trend is exaggerated in the topsoil treatments (at least initially) due to the rapid establishment of seedlings at the outset, combined with little competition from non-sown species. The seasonality seen may also be enhanced due to location, with winds bringing in high levels of salt spray and increasing plant dieback at the trial site. Our results support the idea that during field trials or restoration monitoring it is imperative to record data either regularly, or, within the same few weeks every year (if only occasional monitoring can be achieved) to allow meaningful comparisons to be made (Ruiz-Jaen and Aide, 2005). Later reductions in seasonal variability may have been due to the creation of suitable microhabitats, with more resilient species and plant litter providing increased shelter, as observed previously from forests (Ellis et al., 2015) through to mires (Goodyer, 2014).

Whilst suitable plant assemblages clearly emerged on the fine crushed rock treatments in the absence of topsoil, evidence from the above ground biomass data shows that a significant difference persisted between topsoil and fine rock treatments, suggesting fewer longer term constraints to growth where topsoil is present. Through the sowing of target species, i.e. proactive restorative action, (akin to the 'fast-forward' method of restoration, whereby succession and ecosystem development are accelerated; Hilderbrand et al., 2005), levels of richness, cover and diversity on fine crushed rock treatments continued to progress towards the levels where topsoil was present – at least during these trials to date. This is in contrast to passive cliff top restoration projects, where successional dynamics occur over longer time frames than those tested in this study (Sawtschuk et al., 2010; Shiflett et al., 2013). In the context of upscaling to site-scale remediation, these results suggest that establishing "islands" of vegetation, or strip seeding, may well be a good strategy, as seed production will then allow large-scale re-vegetation in the longer term (Grubb, 1977; Donath et al., 2003; Holl et al., 2011) .

4.2 Invertebrate associations with vegetation

The Society for Ecological Restoration (SER) recommends that indicative factors tested in restoration success include several indicators from their 'recovery wheel' - including invertebrate species composition (McDonald et al., 2016). Indeed, Andersen et al. (2004) considers that biodiversity monitoring cannot be construed as effective if invertebrate surveys are omitted. In many cases, vegetation remains the sole response that is regularly measured and considered (Ruiz-Jaen & Aide, 2005). Testing invertebrate recovery is entirely feasible and desirable, given their ability to respond quickly to changes in both microclimate and microhabitat. In contrast, meso and macro fauna may take many years to respond and return (Déri et al., 2011). Here, a functionally wider ecosystem was considered - and we focussed on trying to attain knowledge regarding a wider range of species, rather than focusing on a single taxa or functional group, as this is only possible by knowing which taxa qualify success (Stork and Eggleton, 1992; Weibull et al., 2000; Andersen et al., 2004).

The greater number of individuals, orders and diversity of ground-dwelling invertebrate orders recorded on topsoil treatments mirrors the results for the vegetation. The heterogeneity (i.e. variety of structure, micro-habitats and food sources) of the trial plots (when compared to that of the surrounding land and the restored meadow) may well have helped generate the elevated invertebrate numbers found. At a small and medium scale, heterogeneity has often been shown to positively correlate with invertebrate numbers and diversity (Weibull et al., 2000; Noreika et al., 2015), though this is not necessarily the case at larger scales (Barsoum et al., 2014).

Despite the proximity of the traps to one another, and their primary use being to assess ground dwelling invertebrates (e.g. beetles), the functional ecology of the orders recorded did seem to match typical invertebrate habitat preferences. For example, two orders, Dermaptera (earwigs) and Opiliolines (harvestmen), occurred more frequently on fine rock treatments, and these both tend to be nocturnal, seeking out protective crevices during the day (Tilling, 2014). Further, detritivores (such as Collembola; springtails) were recorded three times more frequently on topsoil treatments, where a higher amount of vegetation litter was found, a habitat well suited to this group (Hättenschwiler et al., 2005). As more vegetation was present on topsoil treatments, this attracted more herbivores, and subsequently detritivores as vegetation died back. In turn, this provided increased opportunities for omnivores and carnivores.

The final area to be restored at Dounreay is ~1000 times larger than that of the trial plots, and levels of homogeneity will inevitably occur, but overall, a degree of heterogeneity will remain, and this should support the desired diverse range of invertebrates. While we only focussed on ground-dwelling invertebrates, we can speculate that targeted selective seeding of plant species could also create suitable habitat for locally rare invertebrate species more broadly, including *Bombus distinguendus* Morawitz (great yellow bumblebee), and *Cupido minimus* Fuessly (small blue butterfly). Plant species such as Scabious (*Knautia arvensis* (L.) Coult. and *Succisa pratnesis* Moench) and Knapweed (*Centaurea* spp.), and Kidney Vetch (*Anthyllis vulneraria* L.) would be beneficial in this regard. In turn, invertebrates are key prey items for a number of higher taxa (small mammals, birds, amphibians, reptiles) that may all benefit from successful restoration in the long-term.

4.3 Soil, fine rock, or, a mixture?

Assessing changes in levels of nutrients and other variables from the start to end of the trials allows us to comment on their suitability as growing media for the desired vegetation, return of a functional invertebrate community, and if amendments to the materials may be required. The concentrations of several nutrients, including Ca, P, Mg, K, and Na, were lower on trial plots and reference sites than mean levels for other soils (including agricultural land) across northern Scotland (Paterson et al., 2011). The trialled materials tested became marginally more similar to the reference materials (along PC1; see Figure 8) over the course of the trial, though some nutrients exhibited a decline during the trial period (along PC2), being leached out through run-off or possibly taken up by plants. Replenishment through rainfall, aerial particulate deposition and via the turnover of decaying vegetation may occur, but the decrease is a concern. Monitoring foliar uptake would have been a way to determine whether elements were absorbed and possibly recycled, or lost. Longer term experiments/testing on these materials is required to determine if this continues – and if they do it this matters to vegetation assemblages - or if levels stabilise at. The fine rock treatments differed to topsoil with respect to two key elements required by plants, i.e., there were much lower levels of P and much higher levels of Ca. Although the low P recorded did not prevent vegetation growth, it may have inhibited it at the start, and in the longer term this may be a limiting factor affecting the success on these treatments (Chapin et al., 1986). Suding (2011) states that '*recovery relative to reference sites often occurs […] where soils and physical features remain largely intact*'. Use of local materials, and especially topsoil, in the remediation process, could help provide desirable soil seed banks and microorganisms for restoration (Harris, 2009), i.e., providing initial inoculation of mycorrhiza to help overcome low or declining N and/or P as observed in the PCA (Figure 8) (Neuenkamp et al., 2018).

How long desirable habitat recovery takes depends upon methods, but is unlikely to be a linear process (Suding, 2011); intensive restorative activities will help reach critical thresholds more rapidly, and accelerate the return of key attributes and functions, especially here where the site is not surrounded by desired species (Prach and Hobbs, 2008). Remediation of land degraded by mining or chemical spills (Taggart et al., 2004) or remediation of farmland (Wade et al., 2008) will take longer than remediation of less contaminated areas if similar effort is exerted. A loss or absence of soil biota/community may impact on the success of any restoration (Harris, 2009), and although the addition of commercially available products have failed in some experimental settings (Perkins and Bennett, 2017), the transfer of vegetation with sods (to provide that wider soil community/ecosystem) have proved successful (van der Bij et al., 2018). Additions of local materials may help provide the necessary starting blocks to successful remediation in this instance. All these elements should therefore be considered in the development of a large-scale remediation strategy for Dounreay.

5. Conclusions

We have shown through experimental trials, that to tackle a scarcity of topsoil for restoration, fine crushed rock can initially act as a suitable alternative. The successful establishment and persistence of vegetation on the trial plots in a relatively hostile environment (i.e., cliff top/exposed/sea-spray) may bode well for restoration and reclamation of sites in other unfavourable habitats. Even if topsoil was available, the negative ecological impact of topsoil removal (on the donor area) may constrain its use. To minimise these impacts and when topsoil is only limited in small quantity and a very thin layer of topsoil, the establishment of "islands" or the mixing of topsoil with crushed rock may provide realistic, ecologically suitable, alternatives.

Whilst the final goal of having native vegetation and supporting invertebrate communities may not be achieved as quickly with fine crushed rock without any topsoil, if expectation are realistic at the outset (by government agencies/planners/regulators, etc.), this slower approach may still be entirely viable.

In targeting suitable locally relevant species to use in remediation, more successful germination and growth of species can be ensured, but also, aesthetically, the site will then blend into its surroundings more favourably. Expert advice from local botanical institutions, relevant government agencies and NGOs, along with data from reference sites, can help clarify further which species are most suitable. Island or strip sewing may aid in providing the levels of heterogeneity commonly found in natural systems, whilst simultaneously reducing ecological and financial costs.

Differences in soil chemistry between reference sites and trialled materials largely remained at the end of the trials, and amelioration with a phosphorus enriched compost on the fine crushed rock surface could enhance initial growth and support it in the longer term. This would act to provide nutrients (present at low bioavailable levels in the fine rock), and, aid water retention in the initial stages (in what is otherwise a xeric medium). However, there are environmental and economic costs associated with the use of fertiliser, which could weigh in a decision-making process.

Upscaling these trials to the area to be restored in coming years will provide further evidence as to the long-term feasibility of these treatment options. Simple calculations regarding costs, suggest that in reducing topsoil use (by using topsoil mixing or fine rock), 25 – 50% cost reductions could be made. In this remediation project, this could reduce estimated topsoil costs by ~£350K over a 1 ha area. Cost savings involved in using seed islands (thus reducing seed purchase costs), could also be >£100K for example if only 80% of the remediated area were seeded (based on seed cost for the trial plots). Ecological and financial savings in remediation/restoration of larger areas would clearly be even greater, and more than cover the cost of site specific trials of this type (i.e. the costs of a PhD project). Should evidence be attained that certain approaches will work across a wide range of systems and areas, fewer trials may ultimately be required - and as such, a collective goal could be to promote more studies based on the approach utilised here.

Acknowledgements

We thank Dounreay Site Restoration Limited (DSRL) for funding this work as part of a restoration ecology PhD research project, and DSRL staff, particularly Mark Kime, who helped with enabling regular access to the trial plots. We thank Dr Nick Littlewood for provision of training in invertebrate identification, and Alan MacDonald for help with identification and sorting of invertebrates. We thank anonymous reviewers for their feedback in helping us to make improvements to the paper.

6. Appendices

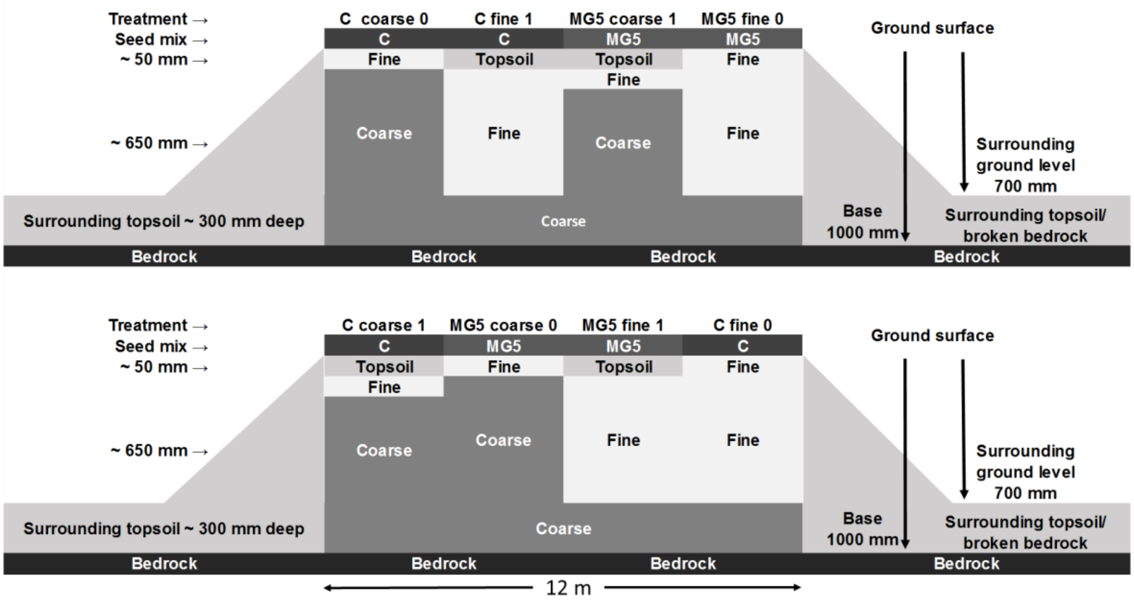

Figure 9. Cross section showing the construction of the eight different treatments trialled in the experiment. The treatments were given abbreviations, e.g., C coarse 0 = coastal seed mix, coarse subsoil, no topsoil; MG5 fine 1 = MG5 seed mix, fine subsoil, topsoil present.

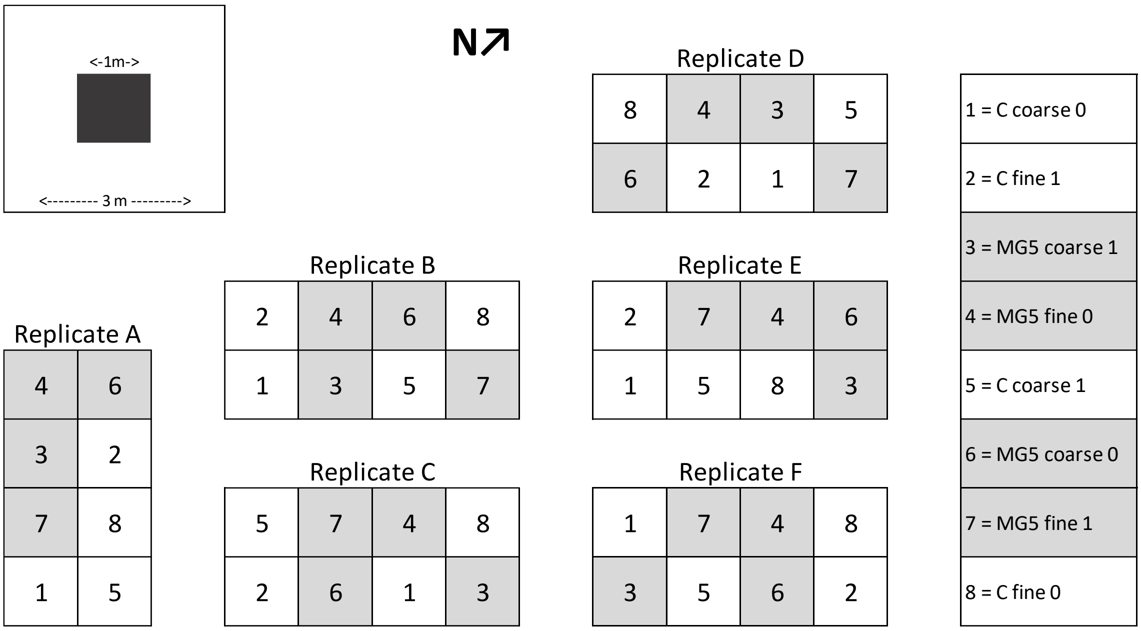

Figure 10. Schematic showing plan layout of the replicate blocks for the vegetation trials. To the top left of the figure is one of the replicates, each 3 m x 3 m, with the central 1 m x 1 m surveyed for vegetation during the trials. In the table to the right the first number indicates where each treatment was on the replicates; the 'C' and MG5' indicate seed mix (coastal mix and mesotrophic grassland mix respectively); 'coarse' and 'fine' indicate the lower rock type in the treatment (coarse crushed or fine crushed respectively); and the '0' and '1' indicate if there was fine crushed rock (0) or topsoil (1) on the surface.

Table 2. Species in seed mixes on trial plots

Species in seed mix and quantities used in vegetation trial plots

Binomial	Common name	Coastal Mix (%)	MG5 Mix (%)
Wildflowers		20.0	20.0
Achillea millefolium L.	Yarrow		1.0
Agrimonia eupatoria L.	Agrimony		2.0
Armeria maritima (Mill.) Willd.	Thrift	0.2	
Bellis perennis L.	Daisy	0.1	0.1
Campanula rotundifolia L.	Harebell	0.1	0.1
Centaurea nigra L.	Common Knapweed	2.0	1.0
Cochlearia danica L.	Danish Scurveygrass	1.0	
Daucus carota L.	Wild carrot	1.0	
Galium verum L.	Lady's Bedstraw	3.0	2.0
Hypericum perforatum L.	St. John's wort		1.0
Hypochaeris radicata L.	Cat's ear		0.5
Lathyrus pratensis L.	Meadow vetchling		1.0
Leucanthemum vulgare Lam.	Ox-eye daisy		1.0
Ligusticum scoticum L.	Scot's lovage	3.0	
Lotus corniculatus L.	Birdsfoot trefoil	1.0	0.5
Plantago lanceolata L.	Ribwort plantain	1.9	1.3
Plantago maritima L.	Sea plantain	0.2	
Primula veris L.		1.0	1.0
Primula vulgaris Huds.	Primrose		0.5
Prunella vulgaris L.	Selfheal		1.0
Ranunculus acris L.	Meadow buttercup		1.0
Rhinanthus minor L.	Yellow rattle	1.0	1.0
Rumex acetosa L.	Sheep's sorrel		1.0
Silene dioica (L.) Clairv.	Red campion	1.0	
Silene latifolia Poir.	White campion	1.0	
Silene uniflora Roth	Sea campion	0.5	
Succisa pratensis Moench	Devil's-bit scabious		1.2
Thymus polytrichus A.Kern. Ex Borbás	Wild thyme		0.5
Trifolium pratense L.	Red clover		0.3
Trifolium repens L.	White clover	1.0	0.5
Tripleurospermum maritimum (L.) W.D.J.Koch	Sea mayweed	1.0	
Vicia cracca L.	Tufted vetch		0.5
Grasses		80.0	80.0
Agrostis capillaris L.	Common Bent	5.0	10.0
Alopecurus pratensis L.	Meadow foxtail	3.0	2.0
Anthoxanthum odoratum L.	Sweet vernal grass	10.0	12.0
Cynosurus cristatus L.	Crested dog's tail	19.0	15.0
Festuca ovina L.	Sheep's Fescue		15.0
Festuca rubra L.	Red Fescue	28.0	21.0
Phleum pratense L.	Timothy grass		5.0
Poa pratensis L.	Smooth-stalked meadow grass	15.0	

7. References

Andersen, A.N., Fisher, A., Hoffmann, Ben, D., Read, J.L., Richards, R., 2004. Use of terrestrial invertebrates for biodiversity monitoring in Australian rangelands, with particular reference to ants. Austral Ecol. 29, 87–92.

Ballesteros, M., Cañadas, E.M., Foronda, A., Fernández-Ondoño, E., Peñas, J., Lorite, J., 2012. Vegetation recovery of gypsum quarries: short-term sowing response to different soil treatments. Appl. Veg. Sci. 15, 187–197. https://doi.org/10.1111/j.1654-109X.2011.01166.x

Barber, H.S., 1931. Traps for cave-inhabiting insects. J. Elisha Mitchell Sci. Soc. 46, 259–266.

Barsoum, N., Fuller, L., Ashwood, F., Reed, K., Bonnet-lebrun, A., Leung, F., 2014. Forest Ecology and Management Ground-dwelling spider (Araneae) and carabid beetle (Coleoptera : Carabidae) community assemblages in mixed and monoculture stands of oak (Quercus robur L ./ Quercus petraea (Matt .) Liebl .) and Scots pine (Pinus sylv. For. Ecol. Manage. 321, 29–41. https://doi.org/10.1016/j.foreco.2013.08.063

Biaggini, M., Consorti, R., Dapporto, L., Dellacasa, M., Paggetti, E., Corti, C., 2007. The taxonomic level order as a possible tool for rapid assessment of Arthropod diversity in agricultural landscapes. Agric. Ecosyst. Environ. 122, 183–191. https://doi.org/10.1016/j.agee.2006.12.032

Borůvka, L., Kozák, J., Mühlhanselová, M., Donátová, H., Nikodem, A., Němeček, K., Drábek, O., 2012. Effect of covering with natural topsoil as a reclamation measure on brown-coal mining dumpsites. J. Geochemical Explor. 113, 118–123. https://doi.org/10.1016/j.gexplo.2011.11.004

Callaway, R.M., Walker, L.R., 1997. Competition and Facilitation : A Synthetic Approach to

Interactions in Plant Communities. Ecology 78, 1958–1965.

Chapin, F., Vitousek, P., van Cleve, K., 1986. The Nature of Nutrient Limitation in Plant Communities. Am. Nat. 127, 48–58.

Déri, E., Magura, T., Horváth, R., Kisfali, M., Ruff, G., Lengyel, S., Tóthmérész, B., 2011. Measuring the Short-term Success of Grassland Restoration: The Use of Habitat Affinity Indices in Ecological Restoration. Restor. Ecol. 19, 520–528. https://doi.org/10.1111/j.1526-100X.2009.00631.x

Donath, T.W., Hölzel, N., Otte, A., 2003. The Impact of Site Conditions and Seed Dispersal on Restoration Success in Alluvial Meadows. Appl. Veg. Sci. 6, 13–22.

Ellis, C.J., Eaton, S., Theodoropoulos, M., Elliot, K., 2015. Epiphyte Communities and Indicator Species. An ecological guide for Scotland's woodlands. Royal Botanic Garden Edinburgh, Edinburgh.

European Commission, 2004. Living with coastal erosion in Europe - Sediment and space for sustainability. Luxembourg office for official publications of the European Union.

Goodyer, E., 2014. Quantifying the Desmid Diversity of Scottish Blanket Mires. University of Aberdeen.

Grubb, P.J., 1977. The maintenance of species-richness in plant communities: the importance of the regeneration niche. Biol. Rev. 52, 107–145. https://doi.org/10.1111/j.1469-185X.1977.tb01347.x

Harris, J., 2009. Soil Microbial Communities and Restoration Ecology: Facilitators or Followers? Science (80-.). 325, 573–574. https://doi.org/10.1126/science.1172975

Hättenschwiler, S., Tiunov, A. V., Scheu, S., 2005. Biodiversity and litter decomposition in terrestrial ecosystems. Annu. Rev. Ecol. Evol. Syst. 36, 191–218. https://doi.org/10.1146/annurev.ecolsys.36.112904.151932

Hilderbrand, R.H., Watts, A.C., Randle, A.M., 2005. The myths of restoration ecology. Ecol. Soc. 10. https://doi.org/19

Holl, K.D., Zahawi, R.A., Cole, R.J., Ostertag, R., Cordell, S., 2011. Planting Seedlings in Tree Islands Versus Plantations as a Large-Scale Tropical Forest Restoration Strategy. Restor. Ecol. 19, 470–479. https://doi.org/10.1111/j.1526-100X.2010.00674.x

Lenth, R. V, 2016. Least-Squares Means: The {R} Package {lsmeans}. J. Stat. Softw. 69, 1–33. https://doi.org/10.18637/jss.v069.i01

Lepš, J., Smilauer, P., 2003. Multivariate Analysis of Ecological Data using CANOCO. Cambridge University Press.

Luken, J.O., 1990. Directing ecological succession. Chapman and Hall, London, UK.

Maiti, S.K., Maiti, D., 2015. Ecological restoration of waste dumps by topsoil blanketing, coir-matting and seeding with grass-legume mixture. Ecol. Eng. 77, 74–84. https://doi.org/10.1016/j.ecoleng.2015.01.003

Malloch, A., 1972. Salt-Spray Deposition on the Maritime Cliffs of the Lizard Peninsula. J. Ecol. 60, 103–112. https://doi.org/10.2307/2258044

McDonald, T., Gann, G.D., Jonson, J., Dixon, K.W., 2016. International Standards for the Practice of Ecological Restoration - Including Principles and Key Concepts.

Mehlich, A., 1984. Mehlich 3 soil test extractant: A modification of Mehlich 2 extractant. Commun. Soil Sci. Plant Anal. 15, 1409–1416. https://doi.org/10.1080/00103628409367568

Merino-Martín, L., Commander, L., Mao, Z., Stevens, J.C., Miller, B.P., Golos, P.J., Mayence, C.E., Dixon, K., 2017. Overcoming topsoil deficits in restoration of semiarid lands: Designing hydrologically favourable soil covers for seedling emergence. Ecol. Eng. 105, 102–117. https://doi.org/10.1016/j.ecoleng.2017.04.033

Met Office, 2013. Met Office [WWW Document]. URL http://www.metoffice.gov.uk/ (accessed 3.20.11).

Neuenkamp, L., Prober, S.M., Price, J.N., Zobel, M., Standish, R.J., 2018. Benefits of mycorrhizal inoculation to ecological restoration depend on plant functional type, restoration context and time. Fungal Ecol. 1–10. https://doi.org/10.1016/j.funeco.2018.05.004

Noreika, N., Kotiaho, J.S., Penttinen, J., Punttila, P., Vuori, A., Autio, O., Loukola, O.J., Kotze, D.J., 2015. Rapid recovery of invertebrate communities after ecological restoration of boreal mires 23, 566–579. https://doi.org/10.1111/rec.12237

O'Brien, P.L., DeSutter, T.M., Ritter, S.S., Casey, F.X.M., Wick, A.F., Khan, E., Matthees, H.L., 2017. A large-scale soil-mixing process for reclamation of heavily disturbed soils. Ecol. Eng. 109, 84–91. https://doi.org/10.1016/j.ecoleng.2017.09.015

Oksanen, A.J., Blanchet, F.G., Kindt, R., Legen-, P., Minchin, P.R., Hara, R.B.O., Simpson, G.L., Solymos, P., Stevens, M.H.H., 2012. Community Ecology Package, … Ecology Package …. https://doi.org/10.4135/9781412971874.n145

Paschke, M.W., 2008. Establishing shrubs to achieve bond release on Colorado coal mines – A demonstration of methods. Final Report.

Paterson, E., Campbell, C.D., Coull, M.C., Shand, C.A., 2011. Geochemical Atlas for Scottish Topsoils.

Perkins, L.B., Bennett, J.R., 2017. A field test of commercial soil microbial treatments on native grassland restoration. Restor. Ecol. 1–7. https://doi.org/10.1111/rec.12639

Perrow, M.R., Davy, A.J. (Eds.), 2002. Handbook of ecological restoration. Cambridge University Press, Cambridge.

Poore, M.E.D., 1955. The use of phytosociological methods in ecological investigations. Br. Ecol. Soc. 43, 226–244.

Poulin, M., Andersen, R., Rochefort, L., 2013. A new approach for tracking vegetation change after restoration: A case study with peatlands. Restor. Ecol. 21, 363–371. https://doi.org/10.1111/j.1526-100X.2012.00889.x

Prach, K., Hobbs, R.J., 2008. Spontaneous succession versus technical reclamation in the restoration of disturbed sites. Restor. Ecol. 16, 363–366. https://doi.org/10.1111/j.1526-100X.2008.00412.x

Prach, K., Pyšek, P., 2001. Using spontaneous succession for restoration of human-disturbed habitats: Experience from Central Europe. Ecol. Eng. 17, 55–62. https://doi.org/10.1016/S0925-8574(00)00132-4

R Core Team, 2016. R: A language and Environment for Statistical Computing.

RStudio Team, 2015. RStudio: Integrated Development for RStudio. Inc.

Ruiz-Jaen, M., Aide, M., 2005. Restoration success: How is it being measured? Restoration Ecology. Restor. Ecol.

Sawtschuk, J., Bioret, F., Gallet, S., 2010. Spontaneous Succession as a Restoration Tool for Maritime Cliff-top Vegetation in Brittany, France. Restor. Ecol. 18, 273–283. https://doi.org/10.1111/j.1526-100X.2010.00653.x

Sawtschuk, J., Gallet, S., Bioret, F., 2012. Evaluation of the most common engineering methods for maritime cliff-top vegetation restoration. Ecol. Eng. 45, 45–54. https://doi.org/10.1016/j.ecoleng.2010.12.019

Scottish Natural Heritage, 2017. Sites of Special Scientific Interest [WWW Document]. URL http://www.snh.gov.uk/

Shiflett, S.A., Zinnert, J.C., Young, D.R., 2013. Changes in plant community composition and structure following restoration of maritime communities 1. J. Torrey Bot. Soc. 140, 89–100. https://doi.org/10.3159/TORREY-D-12-00008.1

Smith, R.K., Dicks, L. V, Mitchell, R., Sutherland, W.J., 2014. Comparative effectiveness research : the missing link in conservation online. Conserv. Evid. 11, 2–6.

Stork, N.E., Eggleton, P., 1992. Invertebrates as determinants and indicators of soil quality. Am. J. Altern. Agric. 7, 38–47. https://doi.org/10.1017/S0889189300004446

Suding, K.N., 2011. Toward an Era of Restoration in Ecology: Successes, Failures, and Opportunities Ahead. Annu. Rev. Ecol. Evol. Syst. 42, 465–487. https://doi.org/10.1146/annurev-ecolsys-102710-145115

Taggart, M.A., Carlisle, M., Pain, D.J., Williams, R., Osborn, D., Joyson, A., 2004. The distribution of arsenic in soils affected by the Aznalcollar mine spill , SW Spain 323, 137–152. https://doi.org/10.1016/j.scitotenv.2003.10.008

Tilling, S.M., 2014. A key to the major groups of British terrestrial invertebrates, Second edi. ed. Aidgap.

Van den Brink, P., Ter Braak, C.F., 1998. Multivariate analysis of stress in experimental ecosystems by Principal Response Curves and similarity analysis. Aquat. Ecol. 32, 163–

178. https://doi.org/10.1023/A:1009944004756

Van den Brink, P.J., Ter Braak, C.J.F., 1999. Principal response curves: Analysis of time-dependent multivariate responses of biological community to stress. Environ. Toxicol. Chem. 18, 138–148. https://doi.org/10.1897/1551-5028(1999)018<0138:PRCAOT>2.3.CO;2

van der Bij, A.U., Weijters, M.J., Bobbink, R., Harris, J.A., Pawlett, M., Ritz, K., Benetková, P., Moradi, J., Frouz, J., van Diggelen, R., 2018. Facilitating ecosystem assembly: Plant-soil interactions as a restoration tool. Biol. Conserv. 220, 272–279. https://doi.org/10.1016/j.biocon.2018.02.010

Wade, M.R., Gurr, G.M., Wratten, S.D., 2008. Ecological restoration of farmland: progress and prospects. Philos. Trans. R. Soc. B Biol. Sci. 363, 831–847. https://doi.org/10.1098/rstb.2007.2186

Weibull, A.C., Bengtsson, J., Nohlgren, E., 2000. Diversity of butterflies in the agricultural landscape: Role of farming system and landscape heterogeneity. Ecography (Cop.). 23, 743–750. https://doi.org/10.1111/j.1600-0587.2000.tb00317.x

Yanahan, A.D., Taylor, S.J., 2014. Vegetative communities as indicators of ground beetle (Coleoptera: Carabidae) diversity. Biodivers. Conserv. 23, 1591–1609. https://doi.org/10.1007/s10531-014-0688-4

Younginger, B.S., Sirová, D., Cruzan, M.B., Ballhorn, D.J., 2017. Is Biomass a Reliable Estimate of Plant Fitness? Appl. Plant Sci. 5, 1600094. https://doi.org/10.3732/apps.1600094

Zahawi, R.A., Reid, J.L., Holl, K.D., 2014. Hidden Costs of Passive Restoration. Restor. Ecol. 22, 284–287. https://doi.org/10.1111/rec.12098

Zhang, H., Hardy, D., Mylavarapu, R., Wang, J., 2014. Mehlich-3, in: Sikora, F. (Ed.), Soil Test Methods from the Southeastern United States. Clemson University, pp. 101–110.