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Woodland restoration on mineral extraction waste tips: a comparison of tree performance over eight years

by Jonathan Millett & John R. Healey

SUMMARY: Woodland restoration on hard-rock waste tips is often unsuccessful because of inappropriate species selection for planting and poor growth of planted trees after initial establishment. In this study we monitored the growth of three native (*Betula pendula*, *Sorbus aucuparia* and *Quercus petraea*) and one exotic (*Alnus cordata*) tree species planted in a restoration scheme on slate-waste tips in North Wales. The objectives of this study were to: (i) investigate the relationship between the relative performance of planted species, and the species' relative success in natural regeneration at the same site; and (ii) investigate how well early performance is correlated with the subsequent success of the species. The growth of the native species did reflect the extent of their natural regeneration at the site (*B. pendula* > *S. aucuparia* = *Q. petraea*). However, while not present as a natural coloniser, *A. cordata* was the best performing species. Species' relative growth performance after eight years was predicted by that during the first few months after planting, suggesting that short-term monitoring has value in predicting longer-term establishment success. Of the species studied *A. cordata* is recommended for planting on slate waste. However, *B. pendula* is the best of the native species studied.

Introduction

Hard rock quarrying often takes place in areas of high landscape and ecological importance. For example, in the UK 61% of carboniferous limestone quarries are located inside areas nationally designated for their conservation value (British Geological Survey, 2006). However, the post-quarrying landscape is generally considered to be aesthetically unappealing and to have little biodiversity value. Mitigation of these impacts is essential if sustainable development targets are to be met and can be achieved through the appropriate creation/restoration of high-value functioning ecosystems once industrial activity has ceased (Winter-Syndor and Redente, 2002). The establishment of woodland on hard-rock waste tips increases their environmental value by restoring ecosystem processes and increasing biodiversity (Rowe et al., 2006) and reduces their negative impact on the visual amenity of the landscape. Consequently, this is the focus of many restoration schemes.

On the waste tips of hard-rock quarries woodland will often establish naturally through primary succession. Nonetheless, the time scales involved are long (even more than 100 years). Restoration often aims to speed up succession (Luken, 1990); on waste

tips this is achieved by planting trees directly into the waste. However, blocky hard-rock waste poses a significant impediment to transplant establishment due to its poor water holding capacity and low levels of nutrient availability (Davis et al., 1985; Wheater and Cullen, 1997; Rowe et al., 2005). In particular slate and limestone waste share a profound shortage of nitrogen (N), which will tend to be the nutrient limiting plant growth in such early successional sites (Chapin, 1993; Marrs and Bradshaw, 1993; Wardle et al., 2004). Consequently, restoration must concentrate on either reducing these limitations to tree establishment and/or using species that are able to establish despite these poor conditions for growth (Williamson et al., 2003).

A range of species are recommended for woodland restoration on slate waste tips (Williamson et al., 2003) with the obvious choice being those that naturally colonise a site. Being able to predict the suitability of a species for planting from its success as a natural coloniser would be a valuable tool for restoration practitioners. However, the niche requirements early in a tree's life tend to be narrower than those of adult stages (Young et al., 2005). As such, species colonising a site will be those that are

not excluded by constraints on seed dispersal, germination and seedling establishment. This may be independent of species' relative survival and growth rates at later juvenile and mature stages. If the use of nursery-raised transplants for ecological restoration does overcome the initial barriers to establishment then a wider range of species might be suitable for transplanting than will naturally colonise a site (Palmer et al., 1997). However, it is not clear if this is in fact the case.

The English Woodland Grant Scheme (EWGS) (Forestry Commission England, 2009) and Scottish Rural Development Programme determine the success of woodland restoration 5-10 and 10 years respectively after planting. However, it might be expected that short-term plant establishment success is representative of these longer-term monitoring requirements. For example, constraints on their initial growth might have a long lasting legacy on the trees. Additionally, plant growth in the first few years will reflect the prevailing biotic and abiotic constraints to growth and it might be expected that these plant-environment interactions will continue to affect growth in a similar way for a much longer period of time. If this is the case, short-term monitoring might be considered a useful tool for predicting the likelihood of meeting longer-term restoration goals, facilitating management planning. However, the lack of longer-term studies prevents the validity of short-term measures to be determined.

In this study we monitored the growth of trees planted in an existing restoration scheme on blocky slate-waste at Penrhyn Slate Quarry in North Wales over a period of eight years. This long period of study

provides a more robust test of transplant success than is generally available. The objectives of this study were to: (i) investigate the relationship between the relative performance of planted species, and the species relative success in natural regeneration at the same site; and (ii) investigate how well early performance is correlated with the subsequent success of the species.

Methods

Site

Penrhyn Quarry (lat 53°9'N, long 4°4'W) is the largest slate quarry in Europe. The site is located in Bethesda, North Wales and covers approximately 265ha, approximately half of which comprises the waste tips alone. These tips are typically composed of 'blocky' waste (i.e. 95% of waste >10cm diameter; Rowe et al., 2006) with very little fine material and are typically greater than 10m in depth. This substrate is particularly hostile to tree establishment and growth, in particular due to low nutrient and water availability (Williamson et al., 2003; Rowe et al., 2005; Rowe et al., 2006). Vegetation cover is generally restricted to tips older than 1975 and areas where fine material has collected (e.g. the base of slopes) (Wheeler, 2001). There is a moderate level of herbivory on vegetation within the quarry boundaries, mainly from sheep which have 'escaped' from surrounding farms.

Three sites within an existing restoration project on the northern waste tips of Penrhyn Quarry (using nursery-raised tree transplants to restore woodland) are used in this study. Each had been subject to different combinations of two contrasting site

Table 1. Site characteristics and the number of individuals of each species measured at the two survey times.

Site	UK Grid Ref	Altitude (a.s.l.)	Site		Number of individuals measured							
			restoration treatment	Date of planting	<i>Alnus cordata</i>		<i>Betula pendula</i>		<i>Sorbus aucuparia</i>		<i>Quercus petraea</i>	
					1998	2006	1998	2006	1998	2006	1998	2006
1	SH 626653	170 m	-topsoil -fencing	May 1997	9	9	28	18	4	1	5	1
2	SH 628652	160 m	-topsoil +fencing	February 1997	14	5	13	11	17	17	23	23
3	SH 642653	150 m	+topsoil +fencing	October 1997	27	19	44	29	15	10	9	9

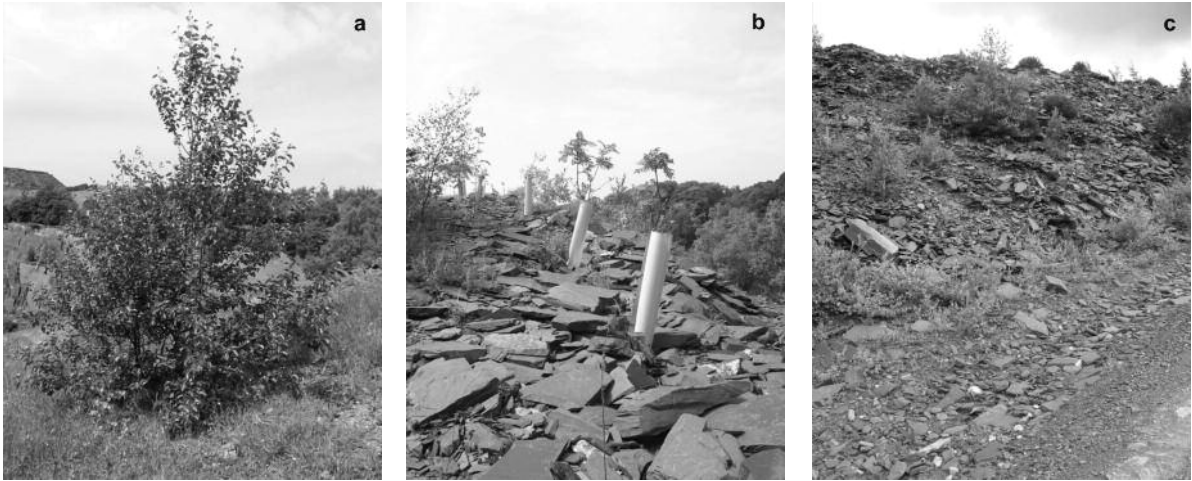


Figure 1. Woodland restoration on slate waste 8 years after planting - trees planted at three sites with different amelioration: a. *Alnus cordata* at Site 3 (herbivore fencing and topsoil addition); b. *Sorbus aucuparia* at Site 2 (fenced with no topsoil); c. *Alnus cordata* (upper centre) and *Betula pendula* at Site 1 (no fencing or topsoil).

amelioration methods that were used to try to improve the success of tree planting: the addition of top soil and fencing against sheep. The three sites occur within a short distance of each other (i.e. <500m), are at a similar elevation (i.e. within 150-170m a.s.l.) (Table 1), and were planted with a similar range of tree species (Fig. 1). As such, they provide a good opportunity to compare tree species performance.

Naturally regenerated vegetation on the site was investigated by Wheeler (2001). Trees were absent from younger tips (created after 1975), whereas the average cover was 34% on tips created before 1975 (Table 2). A total of six angiosperm and two gymnosperm tree species were found growing in pre-1975 tips. Half of this tree cover comprised *Betula pendula*, which also dominated the mature woodland occurring adjacent to the quarry (Table 2). The second ranked species was *Salix x reichardtii* and the third *Alnus glutinosa* both of which were notably less abundant in the adjacent woodlands. Species that were more abundant in mature semi-natural woodland close to the tips than on the

tips included the animal-dispersed *Quercus petraea* and *Sorbus aucuparia* and the wind-dispersed *Acer pseudoplatanus*.

Table 2. Composition of tree natural regeneration on slate waste tips and adjacent mature woodland at Penrhyn Quarry in 2001: mean % cover of tree species in circular plots (20 m radius, 1256 m² area). No trees were present in the post-1975 tips that were surveyed. Adapted from Wheeler (2001) with permission.

Species	Slate waste tips older than 1975 (53-206 m a.s.l., 5-45° slope, 12 plots)	Mature woodland (138-212 m a.s.l., 5-30° slope, 6 plots)
<i>Betula pendula</i>	17.7	48.0
<i>Salix x reichardtii</i> *	7.2	3.5
<i>Alnus glutinosa</i>	3.3	1.4
<i>Acer pseudoplatanus</i>	2.3	10.6
<i>Quercus petraea</i>	2.3	12.7
<i>Fraxinus excelsior</i>	1.3	0
<i>Larix kaempferi</i>	0.1	0
<i>Crataegus monogyna</i>	0	0
<i>Sorbus aucuparia</i>	0	7.8
<i>Ilex aquifolium</i>	0	3.8
<i>Prunus laurocerasus</i>	0	1.5
<i>Pinus sylvestris</i>	0	0
Total tree cover	34.2	89.2

*The *Salix* population on this site comprises both *S. caprea* and *S. cinerea* and their natural hybrid *S. x reichardtii* (E. Rowe pers. comm.); all are referred to here as the hybrid.

Study design

Site amelioration techniques

All trees were planted between February and October 1997 (Table 1) approximately 1m apart into a hessian sack (of approximately 3l volume) filled with peat-free compost and placed in a ‘pocket’ created in the surface of the waste slate. Direct planting into the slate results in almost 100% mortality; this pocket planting technique prevents the tree being planted into an ‘air space’ in the slate (Sheldon, 1975). Furthermore, all trees were protected to some extent from herbivory with 60cm high tree guards. On one site (Site 3) before tree planting, imported top soil had been added to the entire surface of the slate tip to a depth of 1-1.5m. On two of the sites (Site 2 and Site 3) sheep had been excluded with 1m high fencing. Specifically, the sites had the following amelioration: 1. No topsoil, no fencing; 2. No topsoil, fenced; and 3. Topsoil added, fenced (Table 1).

Tree species

Amongst the variety of tree species planted, four were planted at all three sites. These are: the locally native *Betula pendula* Roth. (silver birch), *Quercus petraea* L. (pendunculate oak) and *Sorbus aucuparia* L. (rowan) and the exotic *Alnus cordata* (Loisel.) Duby (Italian alder). The number of each species planted was uneven between the sites (Table 1); however each species is represented by at least four individuals in each site. Furthermore, the tree species are reasonably well interspersed within the three sites, supporting the treatment of each tree as independent of all others.

Measurements

The height of all trees in each site was measured in August 1998. In August 2006 the height and stem diameter of the same trees were measured. Due to the presence of naturally regenerated trees, only those trees that could be identified as having been planted (i.e. through the presence of planting guards or remaining planting media) were measured. Therefore, some surviving trees that had been measured in 1998 will not have been re-measured in 2006. We consider this to be more acceptable than the possibility of measuring trees in 2006 that were not present in 1998.

Data analysis

Tree heights in 1998 were converted into estimated growing-season monthly height growth, due to

differences in planting dates (Table 1). An initial height at planting of 30cm (based on observations at time of planting (T. Rendall, Ecology Company, North Wales, personal communication) and a growing season of April-October were assumed for this calculation. Stem diameters were converted into above-ground dry mass using site-specific allometric equations developed for planted trees at Penrhyn Quarry (M. Nason, University of Wales, Bangor, personal communication). The same model was used for all four tree species because there was very little difference in the model when the species were assessed separately. Height growth between 1998 and 2006 was calculated by randomly pairing trees in the same species/site combination from the August 1998 and August 2006 data sets. Therefore, we were able to assess initial establishment success, height growth over the eight year period and above-ground biomass after eight years of growth. Data were analysed with a Randomised complete blocks design ANOVA using GLM in Minitab 14 (Minitab Inc. 2005). Site was used as a blocking factor and each tree treated as an independent replicate for between-species contrasts. Post-hoc pairwise comparisons between each site x

Table 3. Results of the univariate analysis of three measures of sapling establishment success on blocky slate waste. Presented are the main effects of species and site (blocking factor) from the output of a complete blocks design one-way ANOVA.

	df	F	p
Early establishment^{1,2}			
Spp	3, 196	11.95	<0.001
Site	2, 196	15.75	<0.001
Spp x Site	6, 196	4.43	<0.001
Height growth 1998-2006²			
Spp	3, 143	77.46	<0.001
Site	2, 143	9.29	<0.001
Spp x Site	6, 143	2.00	0.061
Dry mass 2006³			
Spp	3, 143	120.58	<0.001
Site	2, 143	7.72	<0.001
Spp x Site	6, 143	2.07	0.382

¹Mean monthly height growth up to August 1998.

²Square root transformed before analysis.

³Log₁₀ transformed before analysis.

species combination were made using Fisher's LSD test. For all analyses dry mass data were \log_{10} transformed before analysis in order to conform to the assumptions of the tests used.

To assess the relationship between initial establishment and medium-term establishment success we ran a regression analysis with height growth (up to 1998) as the independent variable and dry mass or height in 2006 as the dependant variables. Measurements of the mean for each site x species combination were used in the regression model. Because of the high growth rate of *A. cordata* the regression was skewed to some extent by its inclusion. Therefore, the regression was also run with *A. cordata* excluded.

Results

There were significant effects of both species and site for all measured and calculated variables (Table 3). In 2006 *Alnus cordata* was consistently the largest tree, showing greatest height growth over the eight year period and having the greatest biomass in 2006 (by two to three orders of magnitude) consistently for all three sites (Fig. 2b and 2c). The difference in growth (1998-2006) and in 2006 biomass between *A. cordata* and the other species was greater than any other effects (i.e. amongst species or sites) (Fig. 2b and 2c). *Betula pendula* was the second largest species in 2006; there were no significant differences between *Sorbus aucuparia* and *Quercus petraea*. These differences in tree biomass are also reflected in the height growth between 1998 and 2006 (Fig. 2b). For initial height growth (up to 1998) species differences were similar, with the fastest growing being *A. cordata* and *B. pendula*, and the slowest being *S.*

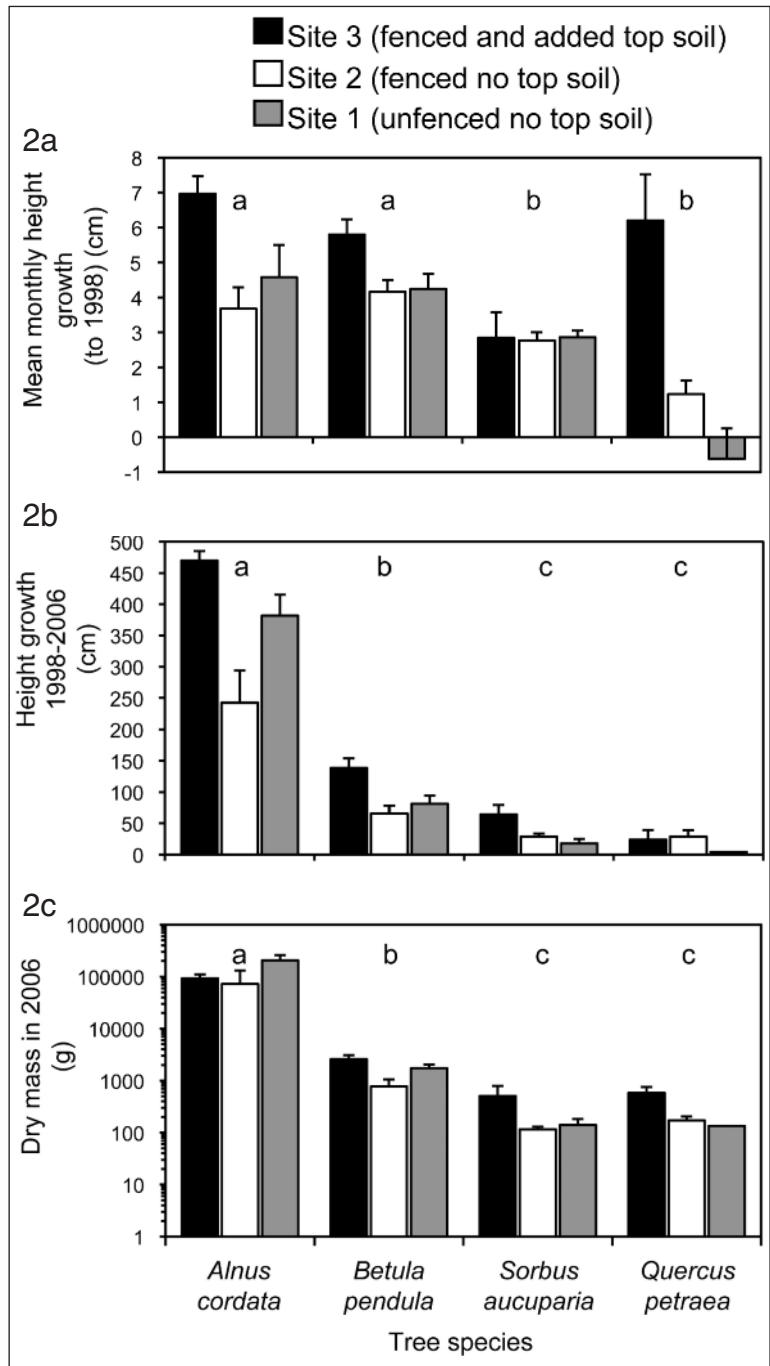


Figure 2. Differences in the establishment success of four planted tree species amongst three sites in Penrhyn Slate Quarry: a. mean monthly height growth between time of planting and August 1998 (see text and table 1 for details of calculation of height growth); b. height growth between 1998 and 2006 and c. biomass in 2006 (log scale). Means \pm SE are shown; species with different letters above them are significantly different from each other (Fisher's LSD).

aucuparia and *Q. petraea* (Fig. 2a).

There were significant ($p < 0.001$) positive relationships between the height growth of species x site combinations in the first months after planting (1998 measurements) and height growth 1988-2006 and tree size in 2006 (Fig. 3a and 3b). The very notable development from 1988 to 2006 was the accelerated growth of *A. cordata* relative to the other three species. If the regression analysis of growth to 1988 against biomass in 2006 is repeated with just the three native tree species, there remains a weakly significant relationship between early and longer-term establishment success (Fig. 3c).

Site differences were less consistent than species differences. Early establishment success was greater in Site 3 than Sites 1 and 2 (height growth to 1998: Site 3 > Site 2 = Site 1 - $P = 0.05$). However, eight-year establishment success differed amongst the sites depending on the measure used. Height growth between 1998 and 2006 was greatest in Site 3 (Site 3 > Site 2 = Site 1 - $P < 0.05$) whereas dry mass in 2006 was highest in Site 2 (Site 2 > Site 1 = Site 3 - $P < 0.05$).

Discussion

There is a relatively long period between the implementation of forest restoration and the evaluation of the success of that restoration (e.g. EWGS = 5-10 years). In this study we were able to show that measurements of tree transplant growth soon after planting were predictive of tree size after eight years. This may be due to one of two reasons. First, constraints on growth in the first few months may have had long-lasting impacts on tree performance. Second, environmental factors limiting growth in the first few months may have continued to act over the following eight years if there were not major shifts in the species' relative niche requirements during this phase of their life cycles (Young et al., 2005). We therefore suggest that measurements of early tree transplant establishment might be of value for predicting the likely success of the restoration, at least in terms of the requirements of the EWGS and the Scottish rural

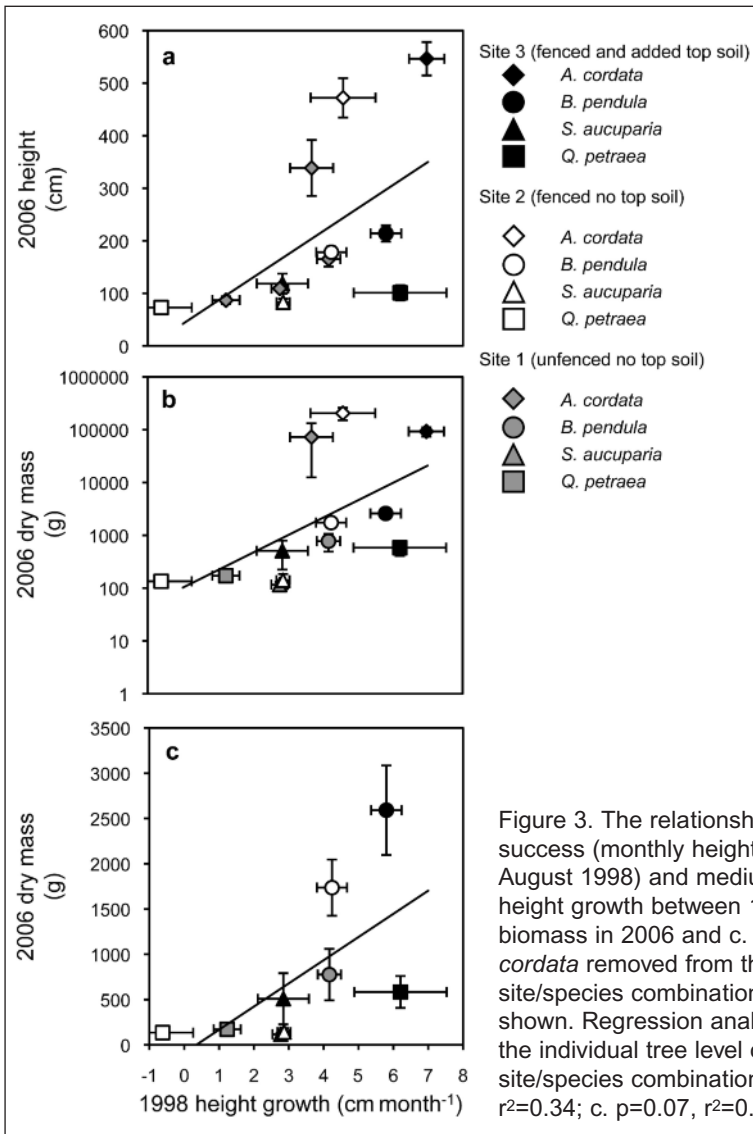


Figure 3. The relationship between initial establishment success (monthly height growth between time of planting and August 1998) and medium term establishment success (a. height growth between 1998 and 2006 b. individual tree biomass in 2006 and c. individual tree biomass in 2006 with *A. cordata* removed from the analysis). Mean \pm SE for each site/species combination and the fitted regression line are shown. Regression analysis was performed on calculations at the individual tree level on randomly paired trees within site/species combinations. a. $p = 0.04$, $r^2 = 0.35$; b. $p = 0.05$, $r^2 = 0.34$; c. $p = 0.07$, $r^2 = 0.41$.

Development Programme. Though we must add the proviso that these monitoring periods are short when considering the restoration of a fully functioning forest ecosystem. Decades or centuries may be required before the success or otherwise of a restoration scheme can be accurately evaluated.

In relating the abundance of species as natural regeneration on the site to their performance as transplants, the most striking result is that the exotic species *Alnus cordata* (which is completely absent from natural regeneration on the waste tips or from adjacent woodland) was the best performing transplanted species by far: height and biomass were considerably greater than the three native species after the eight year study period. *Alnus cordata* is often used on mineral waste due to its ability to tolerate dry and infertile soils (Matthews, 1987) in part associated with its origin in dry habitats in southern Italy and Corsica and its N-fixing symbiosis. However, previous studies which have included *A. cordata* have had mixed results. For example, Rawlinson et al. (2004) found that mortality and growth were relatively high when compared with 20 other tree species planted on closed landfill sites. Furthermore, Rowe et al. (2005) found that over three years its mortality was intermediate and growth was not different to that of *A. glutinosa* or *Salix caprea* planted on very well drained slate waste tips. In contrast our results show the good potential of *A. cordata* in the longer term, which may be related to its low nutrient and water requirements. It is notable that natural regeneration of the native *A. glutinosa* is relatively abundant on the waste tips, even though that species is usually associated with moist habitats such as riverine woodlands (McVean, 1953). However, further controlled trials are required to fully understand the potential of *A. cordata* for restoration of hard-rock waste in comparison with species such as *A. glutinosa* under contrasting climate conditions.

Of the three locally native species, we found that their performance as transplants was well correlated with their abundance as natural regeneration on the site. *Betula pendula* had the greatest height growth rate and mass in 2006 and was the most abundant natural coloniser of this site (as on other slate waste tips in UK (Sheldon 1975). Natural regeneration of *Quercus petraea* and *Sorbus aucuparia* was comparatively rare and their transplant growth rate was less, especially in Site 1, which had no added

topsoil and was unfenced. *Q. petraea* and *S. aucuparia* are relatively abundant in adjacent mature woodlands and these results together imply that their scarcity of as natural colonisers of the slate waste tips may not simply be attributable to limitations of seed dispersal. However, the much greater growth rate of the transplanted, locally absent, exotic species *A. cordata* indicates an important qualification. Where species have no capacity to be dispersed naturally to the site, their absence as natural regeneration has no capacity to predict their capacity as transplants. This indicates the importance for data interpretation of carrying out an inventory of vegetation adjacent to the site (containing potential seed parent trees) as well as on the site itself (as Wheeler (2001) did).

The performance of *B. pendula* and *Q. petraea* was compatible with previous accounts. *B. pendula* is well known as an early coloniser in UK due to its combination of wind seed dispersal and its capacity to grow in conditions subject to nutrient and water shortage (Atkinson, 1992) whereas *Q. petraea* is particularly sensitive to water shortage and has slow growth under low nutrient conditions (Jones, 1959). This was reflected by the very slow growth rate of *Q. petraea* up to 1998 in Sites 1 and 2 (with no added top soil) compared with Site 3 (with added top soil), though this between-site difference became much less for the surviving trees in 2006. The situation for *S. aucuparia* is less clear: even though it is considered to be tolerant of nutrient and water shortage (Raspé et al., 2000), our results show it to have much lower growth rates on slate waste tips than *B. pendula*, especially on Sites 1 and 2.

Added topsoil would be expected to have greater water holding capacity and nutrient capital than the slate waste – ameliorating these constraints to plant growth. However, while tree establishment in the site (3) with imported top soil was initially best, surprisingly, dry mass was highest in a non-top soil site (2) by 2006. Any generalisations regarding differences between sites must be treated with caution due to the lack of replication of site treatments. Furthermore, the different planting dates for the three sites might impact on the early establishment measurements, for example due to planting shock. However, Site 3 had the latest planting date but the fastest growth rate, suggesting that the potential impact of time since planting was small relative to other impacts on tree growth. Nonetheless, in this

study the use of additional topsoil did not improve sapling performance any more than the effect of the pocket planting technique (Sheldon, 1975; Sheldon and Bradshaw, 1975) used on the other two sites or inherent environmental differences amongst the sites. This might be explained through compaction of the topsoil, which can make it an inhospitable substrate for root growth, or alternatively increased competition from weed growth might offset any benefits from the topsoil.

In conclusion, in terms of biomass accumulation *A. cordata* was by far the best species of the four studied for planting on blocky slate waste at this site. It might be assumed that a functioning forest ecosystem would develop most quickly where *A. cordata* is planted. However, in restoration schemes the establishment of woodland of a semi-natural character is often preferred. Therefore, in the UK the use of *A. cordata* is less acceptable than native tree species, of which *B. pendula* was the best performing in this study. However, when choosing species for planting on such sites, the most abundant early natural colonisers are not necessarily the most suitable for the establishment of a woodland ecosystem. For early successional sites these may often be 'ruderal' species (i.e. those with a great capacity for dispersal and quick establishment; sensu Grime, 2002). In contrast on sites subject to such shortages of water and nutrients, the fastest growing species may be 'stress tolerators' (sensu Grime, 2002). From this study, these differences in species relative growth are apparent from a very early stage after transplanting.

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