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Responses of an ancient woodland field layer to soil translocation: methods and timing

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Compaction; *Hyacinthoides non-scripta*; Loose-tipping; *Rubus fruticosus*; Seed bank; Turfing

Nomenclature
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Abstract

Questions: Do woodland field layer communities reassemble effectively after being translocated from a donor to a receptor site? Does the method of soil handling (turfing or loose-tipping) or the season of transfer determine the outcome?

Location: A deciduous ancient coppice woodland in Kent, southeast England.

Methods: Two methods of soil translocation were compared: loose-tipping, involving stripping a layer of topsoil and re-spreading it at a receptor site; and soil placement, in which soil turves were cut and laid intact. The operation also provided an opportunity to observe the effects of seasonal translocation, respectively, in autumn and spring. Soil compaction, texture and chemistry were assessed after translocation, while vegetation cover changes were followed over a 10-yr period.

Results: Immediately after soil transfer, the vegetation community reverted to an earlier successional stage, with bare ground rapidly colonized by annuals and biennials, many germinating from the soil seed bank. By the third year after translocation, a perennial vegetation cover had developed comprising grasses and woodland-edge opportunists as well as most of the original woodland species. After 10 yr, bramble (*Rubus fruticosus* agg.) dominated the receptor sites, along with the woodland geophyte bluebell (*Hyacinthoides non-scripta*), while some woodland species became less abundant. Localized soil compaction radically altered species composition. Autumn translocation promoted better immediate recovery of forest species than spring translocation, while soil placement caused less initial damage than loose-tipping. Convergence in vegetation composition in all soil handling treatments was observed towards canopy closure.

Conclusions: Results illustrate the importance of conducting translocation operations in the autumn when the vegetation is dormant and soil moisture is low, and point to the need for sensitive soil handling. After 10 yr the field layer at the receptor site still resembled that of the donor, but with differences in abundance of some individual species.

Introduction

To facilitate building development, it has become increasingly common to physically re-locate semi-natural vegetation and its underlying soil profile using heavy earth-moving machinery, aiming to reproduce a functioning ecosystem similar to the donor site. Several examples of grassland, heathland and wetland translocations are given in the literature (Buckley 1989; Byrne 1990; Pywell et al. 1995; Jefferson et al. 1999; Bruelheide & Flintrop 2000; Anderson 2003; Vécrin & Muller 2003; Trueman et al. 2007; Kiehl et al. 2010; Török et al. 2011), some of which are reported to have retained their nature conservation value at the new site (Helliwell 1996; Bruelheide & Flintrop 2000; Box 2003). Less information is available on woodland translocation, as fewer such translocations have been attempted or adequately monitored. In the UK, significant areas of woodland have been translocated to compensate for new airports, railways and roads (Helliwell et al. 1996; Roberts 2000; Anderson 2003; Cawley & Corney 2012). Detailed long-term studies comparing different translocation protocols are needed to demonstrate

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whether 'restored' woodland habitats are likely to function as the original habitat.

Two main habitat translocation methods, moving soil profiles rather than trees, are used: placement, involving moving the soil and vegetation as intact 'turves', and loose-tipping (also termed soil transfer) in which soil and vegetation are stripped together and re-spread, mixing the soil profiles. Matching of suitable receptor sites (in terms of soil types, nutrient status, aspect and hydrology), mitigation of damage to vegetation, the potential ingress of undesirable species and subsequent management (Bullock 1998; Box 2003) must be considered. Woodland translocation uniquely involves a sudden long-term change in the light, temperature and humidity regime following canopy removal, creating conditions less suitable for woodland species and opportunities for light-responsive, non-woodland species. Loose-tipping has been the main method adopted, as trees and large embedded roots complicate soil removal compared to grassland and heathland transfers (Helliwell et al. 1996; Anderson 2003).

This investigation describes the impact of soil translocation on the field layer from an ancient woodland located near Canterbury, Kent, UK, over a 10-yr period. The operation provided an opportunity to compare different translocation methods (loose-tipping and soil placement) and contrasting seasons of transfer (autumn and spring). It was hypothesized that light-demanding ruderal species would initially increase at the expense of woodland species, but that the latter would begin to recover towards canopy closure. Soil placement was also expected to cause less disturbance than loose-tipping and therefore favour the survival of woodland species. Similarly, translocation in autumn, when the vegetation was dormant, was expected to be less damaging than translocation in spring.

Methods

The 5-ha donor site, Brickhouse Wood, was an outlier of the 3000 ha Blean Woods complex near Canterbury, Kent, UK (51°17′ N, 1°6′ E, 50 m a.s.l.), situated on Eocene London Clay with a variable covering of silty or sandy drift material (Foster 2002). The climate is temperate and oceanic, with mean maximum and minimum temperatures of 13.8 and 6.3 °C, respectively, and annual precipitation of 728 mm. Tree cover consisted mainly of *Castanea sativa* and *Fraxinus excelsior* coppice with tall standards of *Quercus robur*, unmanaged for at least 60 yr, over a field layer dominated by *Hyacinthoides non-scripta* and locally abundant *Anemone nemorosa*, *Lamiastrum galeobdolon* and *Rubus fruticosus* agg., typical of the *Fraxinus excelsior–Acer campestre–Mercurialis perennis* community (W8b) described in the British National Vegetation Classification (Rodwell 1991). The site was listed in Natural England's Ancient Woodland Inventory, recording woods considered to have been in continuous existence for at least 400 yr (Spencer & Kirby 1992).

Adjacent to the woodland was a working quarry and landfill site with planning permission to expand across the southern portion of the Brickhouse Wood (Fig. 1). As a condition of this permission, 2.8 ha of woodland soil were relocated to adjacent receptor sites. A soil survey carried out in the spring of 1998 distinguished two main soil types in the donor area: 1.7 ha of shallow heavy clay loams over London Clay and 1.1 ha of deeper silty-clay loams over Brickearth drift. Soil depths averaged 22 cm in London Clay, but extended up to 1 m in silty Brickearth. Three nearby receptor sites under permanent grassland were found to have matching soil types, which were prepared beforehand by stripping topsoil layers of 20 and 25 cm on shallow clay and deeper silts, respectively, using a bulldozer. Donor soil was then translocated to the receptor sites in two phases: Phase I in February 1998 (clay soil only), and Phase II in September 1998 and April 1999 (both clay and silty-clay soils; Fig. 1). The entire Phase II translocation was intended to be carried out in September, but wet weather interrupted the operation and it was resumed in the following April, providing an opportunity to compare spring and autumn.

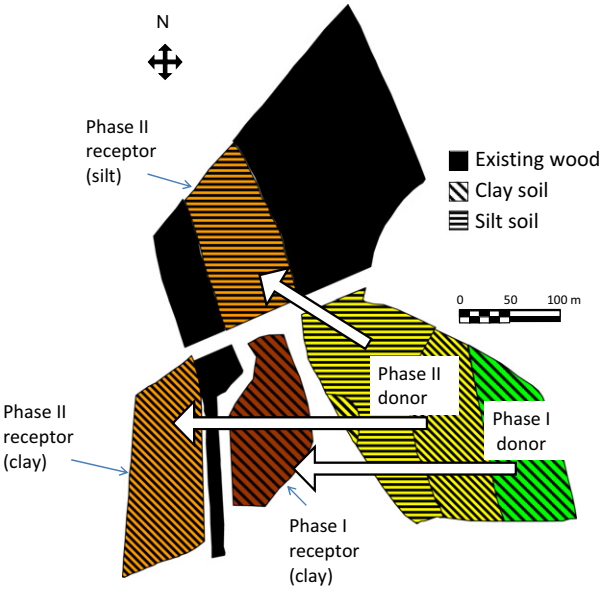


Fig. 1. Layout of the Brickhouse Wood experimental site in spring 1998. Planning permission had been granted to translocate soil from Brickhouse Wood (South) to the proposed Phase II donor site. Soil had previously been moved from the Phase I donor site to its receptor site in January/February 1998. Soil from the proposed Phase II donor site was then to be moved to two receptor sites with matching subsoil types, clay and silty clay.

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Site operations

The donor woodland was worked in 6-m strips, the trees being felled and cleared before removing 20-cm (clay soil) or 25-cm (silty-clay soil) layers of topsoil. The general method employed loose-tipping (McRae 1997; Moffat & Bending 2000), where soil was stripped using a 360° excavator, loaded into dumper trucks and transported to receptor sites. Here it was re-spread immediately as 25- and 30-cm layers for clay and silty-clay soils, respectively, allowing for approximately 20% settlement. To minimize soil compaction, machinery avoided travelling over the woodland topsoil at any point before or after translocation.

During Phase I, receptor site soil was loose-tipped as described, apart from a restricted area underneath electricity pylons where bulldozer spreading was necessary. Translocation at the Phase II receptor sites used loose-tipping either in autumn or spring, and soil placement in spring. Placement involved excavating and lifting, 4 × 1 m intact soil monoliths of 20–25-cm thickness on a horizontal bulldozer blade with extended bucket plates, which were then transported on wooden pallets to the receptor site where they were re-laid. The Phase II operation included 21 4 × 4-m plots located randomly along six transects in the donor woodland, where vegetation cover had been previously surveyed in the spring and summer of 1998, prior to translocation (Table 1). Soil from these same plots was then moved into new random positions at the receptor sites, using either loose-tipping or soil placement. Due to delays in the operation, all placement treatment plots were eventually moved in April. In addition to the 21 pre-surveyed plots, 30 4 × 4-m plots were delineated in random positions at the Phase II receptor sites in May 1999 to record vegetation development, giving a total of 51 permanent plots.

Outside these permanent plots, the receptor sites were planted with trees and shrubs in December/January 1998 (Phase I) and 1999 (Phase II), at 2.1-m spacing using a species mix consisting mainly of *Q. robur*, *F. excelsior*, *A. campestre* and *C. betulus*. No herbicides were used on any of the receptor sites.

Table 1. Numbers of plots in different experimental treatments in Phase I and II of the Brickhouse Wood-South soil translocation experiment. Pre-surveyed plots in the donor woodland are shown in parentheses.

	Clay Soil	Silt Soil
Phase I Soil Translocation		
Loose-Tipping (Lt)	5	
Bulldozer Translocation (BD)	5	
Phase II Soil Translocation		
Autumn Loose-Tipping (Au Lt)	8 (3)	8 (3)
Spring Loose-Tipping (Sp Lt)	7 (2)	7 (2)
Spring Placement (PL)	6 (6)	5 (5)

Vegetation and soil monitoring

Vegetation in the sample plots was surveyed by the same surveyor (MC) twice each year, in May and mid-June, between 1999 and 2002. The survey was repeated in 2008, when only 19 of the original 21 pre-surveyed plots could be re-located. A full species list for these permanent Phase II plots is given in Appendix S1. Percentage cover of all vascular species was estimated visually, including bare ground. Soil samples were taken in February 2000 from each plot to analyse soil texture (Bouyoucos 1962), organic matter (based on loss on ignition), pH (measured in water), total Kjeldahl nitrogen and extractable Olsen phosphorus (Ministry of Agriculture, Fisheries & Food 1986). Three soil cores were taken randomly within each plot to a depth of 14 cm, using a 5-cm diameter Dutch auger, and bulked. Bulk density samples were also recovered from 15 of the 21 pre-surveyed plots in February 1999 prior to translocation, using brass cylinders 7.5-cm long and 5.0-cm diameter at two depths, 0–7.5 and 7.5–15.0 cm, and again after two seasons' settlement in October 2000. Additional soil compaction measurements from all plots were taken in November 2001, using a hand-held penetrometer (Anderson et al. 1980). Eight separate readings, using 3.5 cm increments to a maximum depth of 52 cm, were taken from each plot and averaged.

Data analysis

Total vegetation percentage cover, species number and percentage bare ground were analysed for the Phase II plots ($n = 41$) using linear mixed-effect models (SPSS v 22; http://people.math.aau.dk/~rw/Undervisning/PhD-MixedModelsF13/Material/IBM%20SPSS%20Advanced%20Statistics.pdf; IBM, SPSS v. 22.0., Armonk, NY, USA) to determine the effect of translocation method, season, soil type and categorical year with all interactions, for the first 4 yr (1999–2002) after translocation, with plot as a random effect and year as a repeated measure. Vegetation functional traits were also analysed (Table 2), including species of ancient woodland habitats in the UK (Kimberley et al. 2013), established plant strategies (Grime et al. 1988), life forms, clonality and Ellenberg indicator values based on their proportionate cover values (Hill et al. 2004). These traits were chosen to investigate the response in time to translocation treatments by vegetation life strategy, and in particular for ancient woodland species (including geophytes such as *H. non-scripta*), ruderals developing from buried seed, competitor and intermediate strategy species, and those species developing vegetatively through rhizomes and stolons that might have a competitive advantage in bare-soil conditions. Prior to all analyses, species data (% cover abundance and vegetation func-

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Table 2. Vegetation variables and vegetation functional traits incorporated in LMM and ordination analysis of species cover. Species cover was used for all variable groupings. Ellenberg indicator values were weighted proportionally according to the species composition per plot.

Variables/Trait	Abbreviation
Ancient Woodland Indicator	Number (AWI no.) and cover (AWI cover)
Number of Species	no. spp
Total Vegetation Cover	tot cover
Bare Ground	bare gd
Plant Strategy	Competitors (C); intermediate (CSR); ruderals (R); stress tolerators (S); annual/biennial (ann/bien)
Life Form	Geophyte (geo); phanerophyte (phan)
Ellenberg Indicator Value	Light (Ell L), reaction (Ell R), moisture (Ell F), based on proportionate cover values for each species
Clonality	Extensive rhizome spread (rhiz 2); stolon spread (stol)

tional traits) were tested for normality of distribution and were $\log_{10}$ transformed where required to improve normality and homogeneity of variance across samples; penetrometer readings were also $\log_{10} + 1$ transformed. Paired or two-sample *t*-tests were also performed to compare (1) pre-surveyed plots with subsequent years following translocation, (2) differences between clay and silt soils and (3) differences between loose-tipped and bulldozer treatments in Phase 1.

A multivariate analysis of the 2002 species cover data was performed on the combined 51 plots of the Phase I and II receptor sites to examine the effect of soil properties and soil translocation methods after 4 yr (5 yr for Phase 1), using CANOCO v 4.5 (Microcomputer Power, Ithaca, NY US). Detrended correspondence analysis (DCA) produced short species turnover gradients of 2.465 and 1.933 for axes 1 and 2, respectively, and a unimodal approach was chosen (ter Braak & Prentice 1988; Hirst & Jackson 2007; http://cc.oulu.fi/~jarioksa/opetus/metodi/vegantutor.pdf). Correspondence analysis (CA) of vegetation cover (using sample scaling, species centring and standardization) was then used to interpret translocation methods (loose tipping, placement and bulldozer), season of movement (autumn and spring). Site-related factors (phase, soil type and soil properties), key species abundance and vegetation functional traits were supplementary. To test whether observed differences in vegetation establishment were due to the treatments and site factors, canonical correspondence analysis (CCA) and partial CCA (to adjust for plot differences) were used. Significance testing was carried out using an unrestricted Monte Carlo permutation test in both cases.

A similar procedure was used to follow changes in species composition over time, based on 19 of the 21 original pre-surveyed Phase II plots for all years, including 1998 (before translocation), 1999–2002 and 2008. An initial DCA of these data also gave moderate gradients (3.347 and 2.932 on axes 1 and 2, respectively) and the CA ordination, based on cover values, showed key species (defined as species with >20% fit to the ordination space) and supplementary vegetation functional traits to visualize changes in time that relate to translocation method and season. To test whether the change over time was independent of the treatments, a pCCA on the treatment × year interaction (adjusted for 'plot' and 'year') was run for nominal years with plots permuted together in blocks of year (ter Braak & Wiertz 1994; Leps & Smilauer 2003). An unrestricted Monte Carlo test was used, as the time intervals were uneven. To determine the influence of time alone on species change, a pCCA for all nominal years was run, adjusted for 'treatments' and 'plot.'

Results

Soil properties

In Phase II, total nitrogen and extractable phosphorus were similar in both clays and silts, but soil pH was significantly lower in the clay soils, and organic matter was significantly lower in the silts (Table 3). Phase II soils previously classified as clays or silty-clays were confirmed by textural analysis, the former having mean particle size distributions of clay, silt and sand of 63:27:10%, respectively, and the latter 40:41:19%. In Phase I, soils spread by bulldozer rather than loose-tipping had higher levels of total nitrogen and organic matter than loose-tipped areas, but were more heavily compact at 1.41 MPa. Bulk densities in the upper 0–7.5 cm layers of both clays and silts were slightly higher in samples of both soil types taken after translocation by loose-tipping (Table 4).

Penetrometer readings taken in 2001 showed progressive soil resistance with depth, markedly increasing around 30 cm, at the base of the spread topsoil layer (e.g. Appendix S2, for Phase I). The mean penetration resistance for all depths was significantly lower (0.754 MPa) at the $P = 0.05$ level in the Phase II spring placement treatments compared with spring loose-tipping (0.898 MPa, $t = 2.699$, $n = 25$) but there were no differences between clays and silts.

Vegetation response: years 1–4 (1999–2002)

Differences between autumn- and spring-translocated areas were most noticeable in the first year. Spring loose-tipping plots had 97% bare ground, compared with 57% after placement and 25% after autumn loose-tipping. In the first two growing seasons the vegetation cover increased rapidly, irrespective of soil type, with a commensurate decline in bare ground (Table 5). In all treatments, except for spring loose-tipping on clay where vegetation

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Table 3. Differences in properties between clay and silt soils in the Phase II translocations and compacted (bulldozer) and loose-tipped areas in the Phase I translocation: means with SE in parentheses. (Av. P, available phosphorus; Pen MPa, penetrometer resistance). Two sample *t*-tests were applied.

Phase II	*n*=	Total N^1	pH	Org. Matter[2]	Av. P^1	Pen MPa	% Sand	% Silt	% Clay
Clay	20	2471 (137)	4.47 (0.02)	7.66 (0.31)	10.06 (0.60)	0.87 (0.04)	10.5 (0.7)	26.6 (1.1)	63.4 (1.6)
Silt	21	2218 (96)	4.887 (0.07)	6.41 (0.27)	10.36 (0.51)	0.86 (0.02)	19.2 (0.8)	41.1 (1.4)	40.1 (1.9)
t		1.51	−5.33	3.05	−0.40	0.25	−8.02	−7.93	9.54
Significance		ns	***	**	ns	ns	***	***	***
Phase I									
Loose-Tipped	5	2487 (224)	4.64 (0.20)	7.54 (0.65)	9.19 (1.51)	1.15 (0.08)	10.9 (2.24)	30.2 (1.91)	58.9 (3.80
Bulldozer	5	3314 (153)	4.50 (0.08)	10.26 (0.56)	10.44 (1.25)	1.41 (0.01)	10.5 (0.48)	28.3 (0.53)	61.2 (0.44)
t		−.05	0.45	−.16	−0.605	−3.50	0.18	0.92	−0.60
Significance		*	ns	*	ns	**	ns	ns	ns

*$P < 0.05$; **$P < 0.01$; ***$P < 0.001$.

[1]$\mu g \cdot g^{-1}$ oven dry soil.

[2]Organic matter content determined based on loss on ignition.

Table 4. Comparisons between mean soil bulk densities ($g \cdot cm^{-3}$) before (1999) and after (2000) loose-tipping in clay and silt soils of Phase II. A paired *t*-test was applied for both soil types based on logged values.

	Clay soil				Silt soil				Clay + Silt			
Soil Depth	Before ($n = 7$)	After	*t*	Sig	Before ($n = 8$)	After	*t*	Sig	Before ($n = 15$)	After	*t*	Sig
0–7.5 cm	0.71	1.00	−4.23	**	1.03	1.22	−2.32	ns	0.88	1.11	−4.37	***
7.5–15.0 cm	1.06	1.00	0.55	ns	1.27	1.33	−1.69	ns	1.17	1.17	−0.01	ns

*$P < 0.05$; **$P < 0.01$; ***$P < 0.001$.

cover appeared still to be increasing, cover had stabilized by the third growing season. More species were present on silt soils, but these declined significantly in the year 4, with relatively greater losses in the autumn, loose-tipped and placement treatments compared with spring loose-tipping, where species numbers remained constant over the monitoring period. Ancient woodland indicators (AWI) also declined over the same period, by an average of about one species per 4 × 4-m plot (out of an initial three to four species), with slightly lower numbers on clays. Conversely, AWI cover increased over 2–3 yr, particularly on clays and in spring loose-tipped plots, largely due to the rapid expansion of the geophyte *H. non-scripta*; this cover had stabilised by year 4. Responses of key species to the translocation treatments in each contrasting soil type are illustrated in Appendix S3.

With most species traits, loose-tipping in spring showed a delayed but parallel response to the other translocation methods, resulting in significant method × year interactions (Table 5). Both competitor and competitor–stress tolerant–ruderal (CSR) vegetative cover increased initially before stabilzsing or declining. Competitors included species such as *Holcus mollis*, *Arrenatherum elatius*, *Juncus effusus* and *Cirsium arvense*, while the intermediate CSR group consisted mainly of the grasses *Holcus lanatus* and *Poa trivialis* and the herb *Silene dioica*, which persisted at low level of cover. Annuals, biennials and other ruderals responded to the initial disturbance, but then declined very rapidly. More cover of annuals and perennials occurred on silt soils, but their overall cover was less in spring loose-tipping treatments in the first year. Stress tolerant species such as the AWIs *Anenome nemorosa*, *Milium effusum* and *Lamiastrum galeobdolon* declined more gradually but persisted better on silt soils.

By contrast, *R. fruticosus* and *H. non-scripta*, showed strong increases in cover over the 4 yr. *H. non-scripta*, the dominant geophyte, expanded rapidly after translocation, but developed more slowly on silt than on clay. By year 4 it had achieved 25–50% cover and was still expanding in loose-tipped treatments on clay soils. The phaneroptype group, mostly consisting of *R. fruticosus* (and a few ash seedlings), also increased to 50–70% cover by 2002, an expansion influenced neither by translocation method nor soil type, but it developed faster with soil placement than loose-tipping.

Translocation methods had relatively little influence on mean community Ellenberg indicator values, although marginally higher reaction (R) values occurred in silt and in the soil placement treatment. Soil type was also influential, with higher light (L) values on silts compared with clays. Over time, both light and moisture values (F) fluctuated, with many interactions determined by the particular dominant species. For example, vegetation indicating higher light values occurred in soil placement plots, while

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Table 5. Results of linear mixed model analysis of vegetation traits, recorded over 4 yr, 1999–2002. Estimated mean values for individual years are shown for each translocation method: autumn and spring loose-tipping (LTAu, LTSp) and soil placement (PLSp).

Vegetation Trait	Soil type	Translocation Method												*F*-Values, Significance[1] and df, 1999–2002						
		LTAu *n* = 16				LTSp *n* = 14				PLSp *n* = 11				Method	Soil	Year	Method × Soil	Method × Year	Soil × Year	Method × Soil × Year
		1999	2000	2001	2002	1999	2000	2001	2002	1999	2000	2001	2002							
Total Species	Clay	3.6	149.1	170.2	155.8	21.7	127.4	129.9	142.1	15.5	129.8	148.0	133.6	105.3***	0.088	368.0***	0.034	76.99***	1.3536	1.632
Cover %	Silt	22.5	148.0	145.9	149.6	12.3	123.5	129.5	118.8	25.8	131.2	139.6	140.0	df 2,51	df 1,51	df 3,52	df 2,51	df 6,52	df 3,52	df 6,52
Bare Ground	Clay	26.5	1.6	3.5	2.4	97.0	6.0	5.6	3.6	58.0	2.2	4.2	2.2	27.75***	3.604	472.2***	0.122	7.38***	0.206	0.925
Cover %	Silt	20.8	1.5	2.5	1.3	96.7	3.3	5.9	3.6	56.5	1.7	2.5	1.8	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Total Number	Clay	18.6	14.4	13.1	11.3	13.6	16.6	14.6	13.9	20.0	18.8	17.6	13.4	2.360	47.59***	18.06***	6.985**	4.769***	2.010	0.412
of Species	Silt	31.3	31.3	27.4	25.3	19.3	22.4	21.0	19.3	21.8	21.0	21.7	18.5	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Number of	Clay	3.5	2.6	2.5	2.0	3.3	3.0	3.1	2.1	3.4	3.2	3.2	2.8	0.865	6.953*	22.08***	0.260	3.022*	0.224	1.004
AWI Species	Silt	4.8	3.4	3.8	2.9	3.4	3.9	3.4	2.9	4.3	4.2	3.5	3.0	df 2,51	df 1,51	df 3,41	df 2,51	df 6,41	df 3,41	df 6,41
AWI Species Cover %	Clay	36.0	72.7	70.3	51.8	11.6	32.0	45.6	27.4	2.4	12.1	24.3	23.6	26.72***	20.68***	89.38***	3.122	26.95***	2.429	2.382*
	Silt	3.2	34.3	25.6	27.3	12.3	5.0	15.8	17.3	6.4	31.0	20.3	23.9	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Competitor	Clay	14.4	25.6	25.4	8.2	2.0	28.8	32.6	10.6	2.8	11.7	27.6	27.4	0.606	0.016	129.9***	0.465	8.039***	2.403	1.611
spp. Cover %	Silt	4.5	13.5	15.4	17.6	1.6	16.4	31.3	24.4	2.3	9.1	16.1	17.9	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
CSR. Cover %	Clay	22.2	54.3	54.8	39.4	1.5	63.8	31.9	31.1	19.1	83.9	40.4	14.2	12.62***	0.158	204.1***	0.080	33.52***	3.308*	3.586**
	Silt	44.3	59.1	39.5	23.0	2.8	74.8	45.4	17.7	18.6	82.8	40.6	20.6	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Ruderal spp.	Clay	2.2	1.5	0.5	0.2	0.2	1.0	0.1	0.4	1.7	1.4	0.1	0.0	2.645	3.632	20.21***	0.220	6.015***	2.091	2.139
Cover %	Silt	1.6	2.6	2.1	1.1	0.7	1.4	0.4	0.5	3.4	1.1	0.8	0.4	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Stress-Tolerant	Clay	8.1	8.5	5.9	2.9	0.5	3.7	2.9	1.3	7.9	5.8	4.0	1.4	15.42***	0.389	8.837***	1.116	4.673**	4.553**	1.023
spp. Cover %	Silt	10.5	8.1	14.5	10.0	0.5	2.1	3.0	4.7	4.4	5.4	8.1	2.0	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Annual & Biennial	Clay	12.5	1.5	0.3	0.4	0.9	1.7	0.3	0.5	6.7	5.1	1.2	0.4	16.83***	13.49***	69.74***	4.858*	8.259***	2.415	4.235***
Cover %	Silt	14.2	9.7	3.6	3.2	1.4	3.6	0.3	0.4	8.9	4.2	1.8	2.7	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Geophyte Cover %	Clay	20.6	42.4	46.8	50.0	0.4	6.4	20.5	28.1	1.2	8.3	21.7	24.1	19.47***	0.182	217.2***	2.109	24.03***	4.974***	0.502
	Silt	19.0	37.0	32.9	24.8	0.8	9.6	17.3	17.6	3.4	14.1	27.3	27.2	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Phanerophyte	Clay	1.7	14.0	33.8	54.3	3.1	23.6	39.1	68.9	0.4	18.6	56.2	70.2	1.354	0.217	315.2***	0.075	8.824***	0.564	0.603
Cover %	Silt	1.5	17.4	35.8	58.2	2.6	18.6	30.9	53.6	1.9	17.8	50.3	69.1	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Ellenberg L	Clay	5.54	5.91	5.95	5.94	5.78	6.17	6.05	6.01	5.73	6.17	6.11	6.01	3.277*	10.27**	17.37***	0.151	2.948*	8.92***	3.63**
	Silt	6.06	6.04	6.06	6.04	5.99	6.21	6.43	6.17	6.10	6.33	6.14	6.18	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Ellenberg R	Clay	5.41	5.33	5.47	5.69	5.56	5.44	5.41	5.90	6.06	6.06	6.07	6.08	4.338*	12.624***	3.822*	2.858	1.650	3.208*	0.984
	Silt	6.07	5.90	5.94	6.00	6.23	6.07	6.00	6.00	6.18	6.02	6.03	6.03	df 2,294	df 1,294	df 3,61	df 2,294	df 6,61	df 3,61	df 6,61
Ellenberg F	Clay	5.46	5.61	5.58	5.54	5.79	5.91	5.78	5.72	5.54	5.75	5.64	5.58	2.064	21.00***	19.30***	6.340**	2.229	3.982*	0.982
	Silt	5.90	6.00	6.19	6.13	5.75	5.94	5.95	5.91	5.75	5.85	5.78	5.74	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6.35
Rhizome Cover %	Clay	13.5	22.7	18.5	7.2	1.4	24.0	24.6	0.9	0.9	1.5	1.5	0.7	3.164	0.030	21.62***	2.112	4.111**	5.161**	1.328
	Silt	3.5	11.4	10.4	10.9	0.5	4.0	4.1	1.6	0.0	1.2	1.2	1.9	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35
Stoloniferous	Clay	0.6	0.6	2.3	1.5	0.1	2.4	3.1	4.0	0.1	1.5	2.0	0.3	0.890	9.354**	21.86***	3.952*	1.128	1.767	2.423*
Cover %	Silt	3.4	11.4	13.0	11.4	0.7	2.0	6.4	2.6	3.6	3.1	5.6	5.4	df 2,35	df 1,35	df 3,35	df 2,35	df 6,35	df 3,35	df 6,35

[1]Analyses calculated using log+1 data *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$.

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higher moisture values occurred in the second year following translocation in spring loose-tipping treatments. For species capable of vegetative spread by rhizome or stolon growth, there were no universal responses to translocation method, although stoloniferous species such as *L. galeobdolon* and *Ranunculus repens* were more abundant on silt soils. Both clonal strategies showed initial increases in cover, stabilizing or declining by year 4 (Table 5).

Multivariate analysis

The CA performed on species cover values of the combined 51 Phase II and Phase I plots in 2002 explained 14.5% of the species variance on axis 1 and 9.9% on axis 2 of the model (Fig. 2). CCA was then applied, reducing species variance to 12.6% on axis 1 and 7.3% on axis 2, with 55.3% of the species–environment variance explained. An unrestricted Monte Carlo permutation test confirmed a significant model ($P < 0.001$). The first CA axis represented a combination of the contrast between clay and silt soils, the latter more species-rich and with a higher pH, and the amount of wetland species cover. The second axis was primarily a response to compaction (Lpen), resulting from the bulldozer-translocated plots moved in Phase 1 (Buld PI; Appendix S4, CCA), contrasted with species in loose-tipping and placement treatments in Phase I and Phase II. Bare ground and ruderals (R) such as *J. effusus* and *R. repens* were associated with the more open conditions and areas of higher soil compaction. Testing individual treatments in the CCA showed that pH, bulldozer, loose-tipping autumn, loose-tipping spring and placement spring were all significant in 2002 ($P = 0.001$, except for placement $P = 0.002$), with bulldozer having a high impact and accounting for 14% of the species variability, whereas the other treatments each explained only 6% or 7%.

For the cover data of all years, 1999–2008, on 19 of the pre-surveyed Phase II plots, the CA explained 15.1% of species variance on the first two axes. The pCCA of the treatment × year interaction, adjusted for 'plot' and 'year' accounted for only 7.1% of the total species variance and 47.1% of species–environment variance on the first two axes (Appendix S5). However, Monte Carlo permutation tests indicated a highly significant model ($P < 0.001$). The translocation treatments, over time, had very little effect on vegetation composition and were only significant for loose-tipping in autumn treatment in the first 2 yr (1999 and 2000) and for soil placement in 1999 and 2008 (Appendix S5).

The first axis of the CA ordination diagram (Fig. 3) illustrates the initial disruption in 1999 following translocation, with bare ground and opportunistic ephemeral species such as *Moehringia trinervia*, *Persicaria maculos*a and

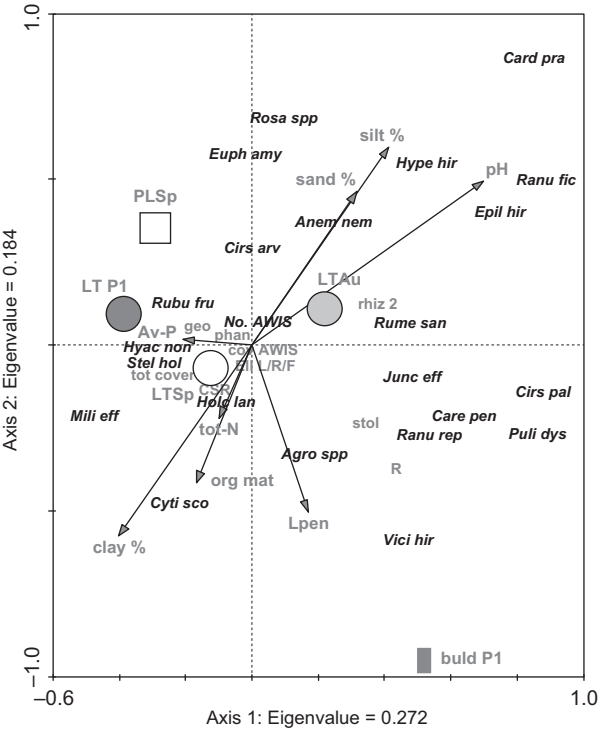


Fig. 2. Correspondence analysis biplot of $\log_{10} + 1$ transformed species cover developing after translocation in 2002 ($n = 51$), showing species and supplementary vegetation functional traits (both with a fit of 35% or more to the ordination space), handling methods (as centroids) and supplementary soil variables displayed for interpretation. Species codes: *Arrh ela, Arrhenatherum elatius*; *Agro* spp, *Agrostis* species; *Anem nem, Anemone nemorosa; Card pra, Cardamine pratensis; Care pen, Carex pendula; Cirs arv, Cirsium arvense; Cirs pal, Cirsium palustre; Cyti sco, Cytisus scoparius; Epil hir, Epilobium hisutum; Euph amy, Euphorbia amygdaloides; Holc lan, Holcus lanatus*; *Hyac non*, *Hyacinthoides non-scriptus*; *Hype hir, Hypericum hirsutum*; *Junc eff*, *Juncus effusus*; *Mili eff*, *Milium effusum*; *Puli dys, Pulicaria dysenterica*; *Ranu fic, Ranunculus ficaria*; *Ranu rep*, *Ranunculus repens; Rosa* spp., *Rosa canina/arvensis; Rume san, Rumex sanguineus*; Rubus, *Rubus fruticosus; Stel hol, Stellaria holostea; Urti dio =Urtica dioica; Vici hir =Vicia hisutum*. Species traits are given in Table 2. Soil handling codes: buld PI, bulldozer spreading; LTAu, autumn loose-tipping; LTSp, spring loose-tipping; PLSp, spring placement; LTP1, loose-tipped Phase 1 plots; Environmental codes: Tot cov, total cover; Org mat, organic matter; Tot-N, total nitrogen; Lpen, penetrometer compaction. % clay, % sand and % silt are supplementary in the model.

Sonchus spp. present, especially in spring loose-tipped plots. By 2000 the receptor sites were being dominated by hemicryptophytes, especially grasses such as *H. mollis,* here expanding rapidly through rhizomatous growth, and other CSR species. Total vegetation cover increased up to 2002, after shade-tolerant perennials such as *H. non-scripta* and *R. fruticosus* became dominant. The second axis shows small differences between spring and autumn-tipped plot centroids, with placement treatments intermediate, with a tendency of autumn tipping to retain more cover of *H. non-scripta* and to gain less

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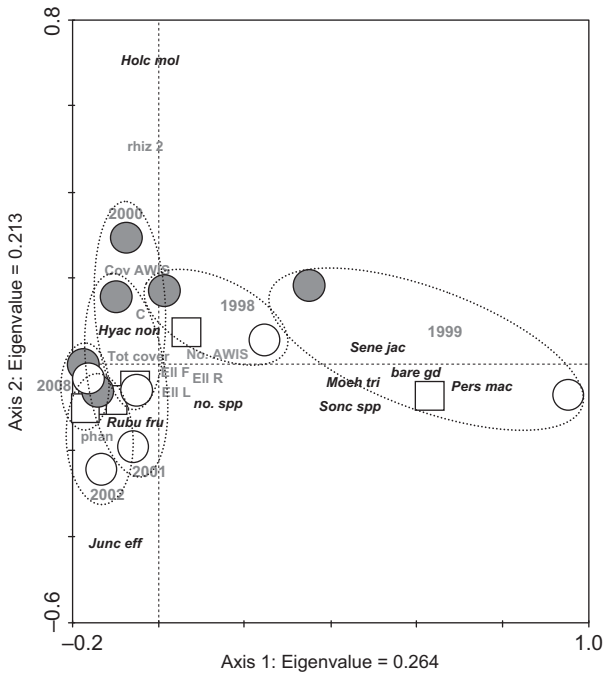


Fig. 3. Correspondence analysis ordination of the species cover data from 1998–2008 (n = 114) using 19 of the pre-surveyed plots in 1998, showing changes in the vegetation community at the Phase II receptor sites. Supplementary variables are survey year and soil handling method (filled and unfilled circles, autumn and spring loose-tipped plot centroids, respectively; squares are soil placement plots). Species with a fit of 25% or more to the ordination space are shown, with vegetation traits as supplementary and the same fit. Species codes: *Holc mol*, *Holcus mollis*; *Hyac non*, *Hyacinthoides non-scriptus*; *Junc eff*, *Juncus effusus*; *Moeh tri*, *Moehringia trinervia*; *Pers mac*, *Persicaria maculosa*; *Rubu*, *Rubus fruticosus*; *Sene jac*, *Senecio jacobaea*; *Sonc spp*, *Sonchus* species. Species traits are given in Table 2. Environmental codes: tot cov, total cover; no. spp, species number; bare gd, bare ground; rhiz 2, extensive rhizome spread.

R. fruticosus. After 10 yr (2008) all three translocation treatments were closely clustered but still quite far from the original (1998) state, being less species-rich, having fewer ancient woodland indicator species but more overall cover. The pCCA model testing years (with translocation treatments and plots as covariables, permuted in blocks) explained 23.72% of the overall variation in cover. All years formed a highly significant model ($P < 0.001$; Appendix S6).

A comparison of the pre-surveyed plots in 1998 with the final recording in 2008 indicated some major species shifts during the recording period (Table 6). Three ancient woodland indicator species *A. nemorosa, L. galeobdolon* and *M. effusum* had declined in cover and frequency, while *H. mollis* expanded slightly and *H. non-scripta* retained its original cover. Some incipient scrub of *F. excelsior* and *Rosa* spp. had developed, but the largest expansion was the three-fold increase in cover of *R. fruticosus*.

Discussion

Effects of soil disturbance on vegetation recovery

Following translocation, the vegetation recovered rapidly, even in the severely disrupted spring loose-tipping treatment. Much of this early cover was generated from the soil seed bank, with annual, biennial and ruderal species dominating. These soon declined and were succeeded by perennial grasses such as *H. lanatus* and *H. mollis* and in turn by *R. fruticosus* scrub, while the geophyte *H. non-scripta* recovered to almost its original abundance level.

Woodland seed banks contain several non-target competitive species potentially capable of swamping specialist forest plants during the open-canopy phase, many of which were present at this site (Hietalahti 2003). In coppiced woods such as this, regular clearances tend to promote high seed bank densities, especially species of forest edges and clearings (Bossuyt et al. 2002; Erenler et al. 2010; Plue et al. 2010). However, compared with the short open phase after coppicing (about 5 yr; Brown & Warr 1992; Mason & Macdonald 2002), complete canopy removal during soil translocation allows competing light-responsive species to persist for 10 yr or more before any planted trees can cast significant shade. Despite the dilution of the seed bank after soil stripping and loose-tipping, the rapid expansion of ruderal species observed here is typical of translocation sites (e.g. Pywell et al. 1995; Helliwell et al. 1996; Jefferson et al. 1999).

Under normal forest conditions, the recovery of shade-tolerant perennial species can be rapid even after significant soil disturbance, for example following catastrophic windthrow (Clinton & Baker 2000) or rooting of wild boar (*Sus scrofa*) (Kotanen 1995; Sims 2005); indeed the altered micro-topography may create additional species niches (Beatty 1984). In the case of this soil translocation, declines in the abundance of ancient woodland indicator species *A. nemorosa, L. galeobdolon* and *M. effusum* over 10 yr are probably the result of prolonged exposure to competitive species and the homogenization of soil structure and topography caused by soil translocation. Of these competitors, the most effective was *R. fruticosus* with its persistent seed bank, effective vegetative spread (Brown & Oosterhuis 1981; Warr et al. 1994; Buckley et al. 1997; Decocq et al. 2004) and ability to form an interim canopy under shade, augmented by a tendency to trap falling litter (Barkham 1992). Nevertheless, even under this dense cover, *H. non-scripta* in particular persisted into year 10. Vickers et al. (2000) have also noted its ability to colonize at forest edges under patches of *R. fruticosus* or *Pteridium aquilinum*.

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Table 6. Changes in percentage cover of species in 19 permanent sample plots at the donor site in 1998 and at the receptor site in 2008. Species with <1% cover in either year are not shown; data are means with SE in parentheses, compared using paired *t*-tests. Ancient woodland indicator species in bold.

	Life Form	Clonal Spread	Primary Strategy	1998 (n = 19)	2008 (n = 19)	Paired *t*-Test (t)	Significance
Declining Species							
Anemone nemorosa	geo	rhiz1	S	4.15 (1.37)	0.17 (0.11)	−2.89	**
Lamiastrum galeobdolon	cham	stol2	S	3.29 (0.95)	0.17 (0.16)	−3.23	**
Milium effusum	hemi	0	S	15.37 (3.73)	1.84 (1.11)	−3.36	***
Poa trivialis	hemi	0	CSR	4.47 (1.86)	0.18 (0.11)	−2.28	*
Increasing Species							
Chamerion angustifolium	geo	root	C	not recorded	1.16 (1.05)	1.11	ns
Holcus mollis	hemi	rhiz2	C	0.31 (0.27)	2.58 (1.20)	2.11	*
Rosa spp.	phan	0	SC	0.13 (0.05)	5.37 (2.47)	2.11	*
Rubus fruticosus	phan	tip	SC	21.53 (4.45)	76.84 (3.25)	9.17	***
Species showing no change							
Arrhenatherum elatius	hemi	0	C	0.58 (0.32)	2.26 (0.84)	1.83	ns
Corylus avellana	phan	0	SC	2.16 (1.35)	0.37 (0.26)	−1.28	ns
Fraxinus excelsior	phan	0	C	0.71 (0.24)	1.74 (0.53)	1.85	ns
Galium aparine	ther	0	CR	6.26 (4.11)	1.49 (0.40)	−1.13	ns
Glechoma hederacea	hemi	node2	CSR	0.97 (0.56)	2.12 (1.27)	0.91	ns
Holcus lanatus	hemi	0	CSR	1.34 (0.45)	5.79 (2.28)	1.87	ns
Hyacinthoides non-scripta	geo	Otb	SR	25.32 (5.75)	18.58 (3.75)	−1.09	ns
Silene dioica	hemi	0	CSR	3.14 (0.90)	6.33 (1.30)	2.00	ns
Stellaria holostea	cham	rhiz1	CSR	5.62 (2.63)	2.59 (1.83)	−0.89	ns

Life form: cham, chamaephyte; geo, geophyte; hemi, hemicryptophyte; phan, phanerophyte; ther, therophyte.

Clonal spread: 0, negligible spread; Otb, bulbous clones; rhiz 1, limited rhizome spread; rhiz2, extensive rhizome spread; root, root suckers; stolon2, extensive stoloniferous spread; tip, tip rooting.

Primary strategy: C, competitor; CR, competitive ruderal; CSR, intermediate competitor–stress tolerant–ruderal; S, stress tolerant; SC, stress-tolerant competitor.

*P < 0.05; **P < 0.01; ***P < 0.001.

Effects of translocation methods and season on species composition

In the very few case studies where soil placement and loose-tipping methods have been compared at the same translocation site, results suggest that placement better maintains the plant community and the integrity of the soil structure (Byrne 1990; Box 1999; Jefferson et al. 1999; Anderson 2003). In this investigation, however, no strong evidence was found in favour of placement, perhaps because of large unsevered roots tearing and loosening the soil during the operation. Problems with lifting woodland soil effectively as intact turves have been cited by others as a reason for adopting loose-tipping (Helliwell et al. 1996; Roberts 2000; Anderson 2003).

As expected, autumn translocation gave better immediate recovery of several woodland species and geophytes in particular, compared with loose-tipping in April. Tipping in the spring, when the plants were actively growing, left many bulbs and rhizomes damaged on or near the surface, where they would have been susceptible to desiccation. Such conditions explain the poor recovery of *Anemone*, which cannot tolerate drying of its rhizomes (Shirreffs 1985; Cummings & Cook 1992), and the slower recovery of *Hyacinthoides* in spring-translocated areas. Damaged plants would also have less opportunity to allocate biomass to underground organs, while the lack of flowering for a whole season reduced seed production in that year. However, all translocation treatments had largely stabilized after 10 yr, with only small effects on the vegetation cover persisting (Fig. 3, Appendix S5), although autumn-spread plots remained better vegetated than those loose-tipped in spring. Time, the changes between 1998 and 2008, explained more of the variation than treatment differences, the vegetation cover reassembling in roughly the same way in all translocation treatments over (Fig. 3, Appendix S6).

As many woodland species have only transient seed banks (Grime et al. 1988; Thompson et al. 1997; Van Calster et al. 2008), spring translocation would have been particularly damaging to any seedlings that had already germinated. Stripping and spreading woodland topsoil during a wet winter period has also been shown to depress seedling recruitment compared with autumn translocation when soils are dry (Rokich et al. 2000). Conversely the timing of translocation was apparently less critical for shade-tolerant species with persistent soil seed banks, even on spring-translocated plots, which in the second year

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were distinguished by profusely flowering *S. dioica* and *Euphorbia amygdaloides*, species able to germinate in almost bare ground with little competition.

Influence of soil factors

Soil texture and chemistry were highly influential in determining the outcome of translocation, underlining the importance of matching soil types and conditions at donor and receptor sites. Multivariate analysis in 2002 confirmed that the plant community was heavily influenced by soil compaction in the high clay content of these surface-water gleys, with impeded drainage over winter. Soil penetrometer and bulk density measurements, however, showed that compaction was minimal, although significant (Table 4), where machinery operators had correctly followed a set code of practice for loose-tipping. In contrast, where soil had been spread with a bulldozer in Phase 1, significant compaction occurred due to the low reach of the machine and the necessity for it to travel backwards and forwards over subsoil in winter on soil already at field capacity. The association of species such as *J. effusus, Carex pendula* and *R. repens* with these plots suggests waterlogging was caused by compaction and probably also by the loss of a transpiring tree canopy. In the bulldozer-translocated area *Hyacinthoides* recovery was poor, probably due to its intolerance of compaction and waterlogging (Blackman 1954; Knight 1964; Pigott 1982).

Conclusion

As predicted, soil translocation created opportunities for ruderal and competitive vegetation to dominate some, but not all, ancient woodland indicator species. The original aim was to move woodland soil and its associated flora so that at least some key species would be maintained and the habitat, given time, would begin to resemble the donor woodland. In this narrow sense the objective was achieved, as many cosmopolitan shade-tolerant woodland species persisted at the receptor sites up to the point of canopy closure, and no species were actually lost over the 10-yr period. However populations of some ancient woodland indicator species were much reduced while others were resilient and some, such as the geophyte *H. non-scripta*, regained cover values similar to those in the donor wood. The vegetation community moved closer to the pre-translocation state after 3 yr, but by year 10 some divergence in cover, pattern and composition was still apparent, with increases in open-ground and woodland edge species. Differences in vegetation composition due to translocation treatment were much less important than its recovery through time.

Soil compaction is clearly an issue with translocation, as the influence of bulldozer-moved soil on penetrometer readings and vegetation in Phase 1 clearly showed. Loose-tipping and placement were more benign methods, but this study found only minor differences between them, for the reasons given. The more profound differences were between autumn and spring operations and in the recovery time of the vegetation moved by different methods.

To expect a closer match to the donor site within 10 yr, prior to full canopy closure, would be unrealistic, not only because woodland vegetation is affected by translocation methodology, microtopography and soil type, as acknowledged by other practitioners working on open habitat translocations (Bullock 1998; Box 1999; Anderson 2003), but also by the fundamental change in light regime. Although one alternative to translocation would have been to allow natural recolonization of the open land from the fragments of woodland left behind (Brunet et al. 2012), it is likely that the dispersal rates of ancient indicator species such as *A. nemorosa* and *H. non-scripta* would have been much lower.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. Trends in percentage species frequency in the permanent Phase 2 plots over the 10-year period, with species traits indicated: ancient woodland indicators in bold type.

Appendix S2. Soil penetrometer pressure data (MPa) taken at increasing profile depths, at the Phase I receptor site in 2001.

Appendix S3. Mean percentage cover values for key woodland species showing different patterns of development after Phase II translocation in clay and silt soils.

Appendix S4. CCA across site significance testing: 2002 vegetation cover and environmental data from 51 plots.

Appendix S5. pCCA significance for the model translocation method–season × year with covariables plot and year.

Appendix S6. pCCA significance, testing the effects of individual years, 1998–2008, on vegetation cover.

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