Final Report on the Afforested Environments Study (2011-2015):

The Feasibility of Mitigating Hardwood Forest Removal through Afforestation of Farmland

Paul J. Richardson* & Stephen D. Murphy

Centre for Ecosystem Resilience & Adaptation, University of Waterloo

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*Corresponding author: dr.paul.j.richardson@gmail.com

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-PJR & SDM

Part 1: Executive Summary

Context:

1.1 Businesses seeking to develop mineral aggregate resources through extraction in southern Ontario must, on occasion, remove mature hardwood forest cover from the landscape to access underlying deposits. While forest cover can sometimes be returned to these areas through rehabilitation, efficient resource development often requires extraction proceed below the water table, resulting in final ecosystems which are aquatic rather than terrestrial. Given guiding principles and in some cases legislation demanding no net forest loss from the landscape, licences for new extraction operations increasingly demand compensation (mitigation) for proposed forest removals in the form of new off-site forest plantings (usually on marginal-value farmland), in order to functionally replace stands slated for removal.

1.2 As such, managers of forest creation (afforestation) must be able to establish new self-sustaining forest ecosystems that will eventually provide all of the ecological "goods and services" supplied by the original forest cover. These include ecosystem processes such as tree growth, wood production and removal of greenhouse gasses from the atmosphere, as well as nutrient and water cycling through the environment. Perhaps even more importantly, mature natural forests provide high-quality habitat for diverse ecological communities representing the natural heritage of the landscape. These span multiple forest strata and divisions of life, from microorganisms in the soil to mosses, lichens and higher plant guilds in the understorey (including ferns, herbs, shrubs and saplings), to the canopy-forming tree community. Diverse fauna is in turn supported by the varied vegetation, from beetles to butterflies, salamanders to songbirds, badgers to bats. Indeed, ecologists increasingly believe such diversity of flora, fauna and their interactions help forest processes to operate productively and with high resilience in the face of environmental change.

1.3 The need to accurately replicate vital features of mature natural hardwood forests within typical afforested environments constitutes a major problem for ecosystem planners and managers. This is because although forestry practitioners have developed effective strategies for growing trees in the context of wood production, ecologists lack basic knowledge about how well conventional afforestation methods capture the wide biological diversity inherent within mature natural hardwood forests (referred to hereafter as target or reference forests). For example, we do not know how closely planted forests resemble target forests with respect to many key plant-community and ecosystem properties. Nor do we know how much time is needed to reach different thresholds or aspects of similarity, or what management steps may impede or accelerate the pacing of this. These knowledge gaps beg the practical question of how forest development processes playing out over many decades can even be studied effectively within a typical 3-5 year research window.

1.1 Executive Summary

Investigation:

1.4 The Afforested Environments Study (AES) was initiated in 2011 to work towards answering these questions, with the goal of closing knowledge gaps currently preventing successful mitigation of forest removal through afforestation. We selected 36 representative planted forest stands for study, i.e. conventional plantation stands consisting of spatially-uniform plantings typically corresponding to only one or a few softwood species. Collectively, the plantation stands comprised a chronosequence of forest development because they spanned a 30-90 year gradient with respect to the duration passed between initial stand planting and the beginning of the study ("time since afforestation"). Stands additionally varied with respect to the degree of tree-thinning previously experienced – either thinned regularly ("T") or left under-thinned ("U") – and with respect to the types of trees initially planted (i.e. receiving either a single softwood species (monoculture, "M"), a mixture of softwood species ("X"), or a mixture of hardwood and softwood species ("H"). In addition to the 36 plantation stands, we included 5 mature natural hardwood stands (typically referred to as "old growth" or "older growth" by the forest managers) to represent the best-case mitigation reference state ("R").

1.5 All stands were selected from a larger pool of potential sites (N=123), at random but meeting criteria so as to ensure useful variance in stand age and type. This step was essential for maximizing the degree to which results from the study sites could be generalized to the wide swathe of southwestern Ontario represented by the pool of potential sites.

1.6 With respect to 42 features characterizing the diversity and structure of forest ecosystems, we compared planted stands to natural ones and analyzed the degree to which similarities and differences were related to aspects of site history, including management practices and time since afforestation. Features compared ranged from aspects of the canopy-forming tree community (i.e. canopy closure, stand density, species diversity and composition) to understorey vegetation (i.e. amount of area covered, number of species present, composition of species) to duff-layer properties (e.g. cover by conifer needles vs. broad-leaves) to coarse woody debris (i.e. snags, stumps and fallen logs), micro-topography (i.e. pits and mounds), and soil composition (i.e. texture, organic matter, bulk density, moisture and nutrients).

1.7 We solved the problem of how to study a multi-decade process in just a few years in part by crossing the chronosequence approach with a "plant indicator" or phytometer experiment. A phytometer is any species of plant capable of reliably signalling information about the local environment through its growth, or failure to grow, following experimental relocation to that environment. This is analogous to sending a canary into a coal mine, but is particularly important for plant ecology because it can help distinguish between the rate at which target habitat conditions develop versus the rate at which species capable of taking advantage of such habitat immigrate. Knowledge about both types of processes is important for ecosystem planning, but focussing solely on spontaneous colonization may result in the random-chance aspect of immigration success obscuring the potentially more orderly and predictable process of habitat development. Spontaneous occurrence of typical hardwood-forest understorey herbs in a mature plantation would certainly indicate that vital habitat requirements have been met, for example, but the absence of target species could be due to fundamentally unsuitable habitat conditions *or* to immigration barriers such as small and fragmented source populations. Experimentally introducing the target species to test sites and assessing how well it survives relative to performance within the home environment neatly overcomes this difficulty.

1.8 By experimentally relocating two target-forest herb species to all plantation stands, we were able to determine rates at which habitat equivalence to R forests develop over time, independent of immigration. We selected one species expected to be particularly sensitive to the light environment (the shade-tolerant *Asarum canadense*, or wild ginger) and one species expected to be sensitive to other conditions such as soil quality (the ephemeral spring herb *Allium tricoccum*, or wild leek). While both chronosequence and phytometer approaches have previously been used to great effect separately independently, the AES was the first study to use both approaches simultaneously to calculate time-lags between forest plantings and emergence of habitat equivalence to target forests.

1.9 For each forest feature investigated, we determined whether average levels within different plantation types differed from those in R forests, and whether the degree of difference changed as stands developed over several decades. Based on the magnitudes of the differences and the degree to which these were projected to shift over time, we classified the feasibility of capturing each target feature through afforestation, noting whether any one plantation type performed significantly better than the others.

1.10 We classified feasibility using a scale whereby a target feature was considered “Likely” to be achieved within a plantation stand if its occurrence was consistent with R forests within 30-40 years of stand planting and this similarity did not decrease as the plantations continued to age. We considered a feature’s replacement “Feasible” if it gradually emerged spontaneously in at least one type of plantation and converged with R forests within 60 years of planting. Likewise, a feature was “Possible” if it emerged spontaneously 61-90 years after tree planting. “Challenging” features were those where convergence with R values was not projected to occur until at least 91-150 years after planting. Finally, features were classified as “Unlikely” if they differed significantly from R forests regardless of plantation age, showing no sign of increasing similarity as stands developed.

Findings:

1.11 In total, 42 target forest features were measured, compared and classified using the feasibility scale. The results are summarized in Table 2.1 of the Executive Report. Fourteen of the target features (33%) are Likely to be achieved within conventionally afforested farmlands; 7 (17%) are Feasible; 12 (29%) are Possible; 3 (7%) are Challenging; and 6 (14%) are Unlikely under current management practices. Put another way, if a new forest were planted today using typical methods, in 90 years this would most likely be indistinguishable from natural hardwood forests with respect to nearly 80% of investigated features. An additional 60 years of stand development would be required to increase the success rate to 86%, and a residual 14% of features will probably remain distinct from R forests no matter how much time passes.

1.12 Features classified as Likely included canopy-level features (e.g. canopy closure, stand density), understorey vegetation properties (e.g. cover by ground flora), and microhabitat properties (e.g. surface cover by bare soil and fine woody debris; soil texture, cation exchange capacity). For many such features, plantations and reference forests were not distinguishable primarily because both forest types exhibited considerable site-to-site variation. This highlights the fact that the reference state in this context is not a single, narrowly defined and inflexible entity but rather a diverse and dynamic ecological system. The goal of mitigative afforestation should thus not be construed as perfect replication of any given forest, but rather development of new forests that are themselves variable but falling within the range of variation exhibited by the reference forests.

1.13 Target forest features classified as Feasible were mainly properties that did change substantially as plantations aged, generally reaching R levels 50-60 years after tree planting. These included the diversity of tree species growing, but only in H plantations. In T plantations, surface cover by fallen conifer needles steadily declined with plantation age while that by fallen broadleaves increased, both reaching R levels nearly 60 years after stand planting. The diversity of trunk-diameter size-classes represented similarly increased and reached target levels over this period, but only in T plantations. Some soil micronutrients (e.g. calcium, magnesium) likewise changed predictably and converged with R values over this duration, but these bear watching because they tended to continue to shift even after target conditions were met, “overshooting” targets eventually.

1.14 Features classified as Feasible tended to follow a well-defined trajectory of increasing similarity to R forests over time, converging with targets 70-90 years after tree-planting. These included the volume, diversity, composition and decay state of coarse woody debris (CWD) objects (e.g. snags, stumps and logs), as well as the frequency of pit-and-mound structures on the forest-floor. Importantly, the capacity for both CWD and micro-topographic features (MTF; e.g. pits and mounds) to directly support a high diversity of plant species was also realized over this timespan, indicating biodiversity-promoting habitat features can spontaneously develop within afforested environments. Perhaps most importantly, both wild leek and wild ginger phytometers indicated that plantation stands developed highly suitable habitat over this same duration (70-90 years post-planting), though such effects were only evident in plantations where hardwood species were *not* planted.

1.15 It may seem counterintuitive that planting hardwood species is not advantageous when trying to establish a hardwood forest, but the results of this study suggest this is indeed the case, at least when employing conventional methods. The reason is likely that planted softwood species grow faster than hardwoods and create a semi-closed canopy relatively quickly, conditions supportive of native hardwood species from other forests in the landscape which immigrate. The relatively quick development of target habitat conditions (as indicated by the phytometers) within T plantations compared to other types indicated that regular thinning of planted softwoods complements such development, likely by opening-up valuable growth space and eliminating potential competitors once suitable environmental conditions have developed.

1.16 Aside from phytometers, an important spontaneous change in vegetation composition projected to occur between 60 and 90 years after stand planting is a drastic drop in the proportion of understorey species corresponding to non-natives. Given that the total number of species was constant, this decline indicated that initially thriving exotics were slowly but steadily replaced by native species as the planted stands developed.

1.17 The three Challenging features discovered – i.e. those projected to eventually converge with R forests but requiring up to 150 years to do so – were the species composition of the canopy tree community (requiring 105 years in H plantations but 130 years in T plantations), the species composition of the understorey (requiring 150 years regardless of plantation type), and the frequency of small pits occurring on the forest floor (requiring 130 years). Of these, pit frequency is of relatively low concern since near-target levels are reached within 90 years, pit structures could likely be created by managers fairly easily, and we found mound to be more important than pits with respect to providing habitat for diverse plant species.

1.18 The slow emergence of species composition is of much greater concern. While only representing 5% of the investigated features, species composition is tightly linked to forest capacity to support natural-heritage plant communities (i.e. those including iconic species characterizing Ontario hardwood forests and unlikely to find safe habitat elsewhere, such as trilliums). Our discovery that planted stands are on slow but nevertheless progressive trajectory towards compositional equivalence to R forests bodes well for afforestation managers in that it suggests that the goal can eventually be reached even if management practices remain unchanged. However, the relatively long duration required seems problematic, as the ecological processes governing forest development are themselves undergoing changes related to land-use transformations and climate change. If source populations of target species are in decline, it is difficult to imagine pressures will be the same over the next century as they were over the one. Plant communities within plantation stands were considered to be highly similar to R forests if they were just as similar to the group of all R forests as were each of the individual R stands to the group of R stands, on average (i.e. a measure accounting for the natural compositional variation among R stands, as well as between plantations and R).

1.19 Three of the target features determined to be Unlikely to converge with R forests in plantations are directly related to compositional differences in the understorey, while the other three are soil differences which may be indirectly related to understorey vegetation. Specifically, the number of plant species present per site (species richness), the number of species present per square-metre plot (species density), and the relative equality in abundances of different species per plot (species evenness) were all lower in plantations than reference forests and showed no sign of increasing as the stands developed. Species richness, for example, was 26% lower in T and U than in R stands (i.e. 34 vs. 46 species per site). The soil features deemed Unlikely to be replaced were bulk density, organic matter, and moisture content. They are related in that soils underlying areas of human activity often become compacted and disturbed, resulting in high bulk density and low organic material, which in turn can reduce soil moisture-retention capacity, nutrient content, and capacity to support plant growth.

Significance & Applications

1.20 Our analysis of the feasibility of achieving within planted forests the 42 target-forest features highlighted many forests properties which are likely to be successfully replaced through conventional afforestation, as well as targets that managers can anticipate will be more challenging. Of vital importance is the light shed on opportunities for overcoming the challenges and reaching targets more quickly, more completely, and with greater cost efficiency. Some such opportunities still need careful research and development at a management-relevant scale, but others are ready to be incorporated into afforestation planning immediately. We have discovered that conventional uniform planting of a low diversity of quick-growing softwood species can indeed eventually yield naturalized forest ecosystems with high ecological similarity to heritage hardwood forests, recovering up to 79% of desired features within 90 years of planting (a report-card "B+"). However, this is best achieved by planting mixtures rather than monocultures, keeping hardwood species out of the mix (or at least in low abundance and scattered among the softwoods) and maintaining regular stand-thinning (e.g. every 7-15 years after the first 30 years of development).

1.21 Crucially, 60-80 years after stand planting a leverage point arises whereby concerted management intervention in the form of assisted species immigration may yield major gains in ecological similarity to target forests. Introducing target species at the right juncture may help plantations cross thresholds of species diversity and composition that otherwise prevent planted stands from fully reflecting the target state. This discovery was the valuable product of combining phytometer with chronosequence methods: phytometers showed assisting immigration to stands younger than 60 years would likely be ineffective due to inappropriate habitat conditions, but after this point plantation habitats were indistinguishable from reference forests from the phytometers' perspective. This is significant because if such an opportunity were to be ignored, an additional 60-90 years of stand development would be needed to reach equivalence with target forests.

1.22 The discovery of persistent soil differences between planted and natural forests indicates there is potential for strategic soil amendments to further accelerate the pace of emerging similarity to reference forests. Overcoming soil compaction, poor water retention and low organic matter can be straightforward, simply requiring scraping into the subsoil and intermixing low-nutrient, vegetation-based compost during initial tree planting and again 30-60 years after planting (perhaps coinciding with stand-thinning). Focussing on several 10-100 m^2 patches of forest floor to receive multiple interventions (e.g. soil and plant introductions) rather than the entire stand would make intervention management logistically feasible, easily testable, and likely to promote greater physical and biological diversity within treated stands.

1.23 Results of the AES point to opportunities for significantly advancing the state of the art and science of mitigative afforestation. Chief among these is new research aimed at developing innovative and efficient methods for capturing numerous features of target forests over a much faster timescale: 30-50 years after tree-planting. The hypothesized method would be ideally suited for afforestation in the context of mitigating forest removal because it would effectively "recycle" valuable biological material already present within mature natural stands slated for removal. Bulk transfer of topsoil – including the intrinsic stores of high-quality soil features and plant biodiversity in the form of propagule roots and seeds — from such stands to very recently planted afforestation fields would be the first step in such a method, but one that is unlikely to be fully effective on its own given that most target plant species require at least moderate shade and other conditions of closed-canopy forests. This is why the critical second step would be to quickly develop different strategies for artificially mimicking semi-closed canopy habitat conditions and maximizing ecological "yields" from imported topsoil.

1.24 The chief significance of the 2011-2015 AES findings is that managers can now plan and implement mitigative afforestation with high confidence that the outcome of forest planting will be a nearly complete ecological replacement of the forest ecosystems undergoing removal. Even better, managers can now accurately predict how long this process will take, plan for time-lags associated with different specific features and similarity goals, and implement appropriate management strategies for minimizing time-lags and maximizing the degree of similarity to reference forests. The work also serves to highlight the importance of explicitly defining and prioritizing goals at the afforestation planning stage, since methods which are most effective for meeting one goal may be least effective for meeting another (e.g. planting hardwoods species accelerates canopy-level similarity but impedes understorey similarity to target forests). Perhaps most valuably, the AES provides managers in the aggregates industry with tools for demonstrating that not only is it possible to develop a natural resource with *no net loss* of forest ecosystems from the landscape, but appropriate afforestation measures can even ensure net ecological *gains* by improving connectivity among forest fragments and providing essential refuge for heritage biodiversity.

Part 2: Executive Report

2.1 The Problem of Mitigating Forest Loss

2.1.1 Businesses seeking to develop natural resources in Canada are socially and legally bound to minimize negative impacts on the environment. For sand, stone and gravel producers, licenses to develop new pits and quarries can be difficult to acquire. Applications may require comprehensive plans showing capacity to not only rehabilitate extraction sites, but also to fully mitigate the loss of ecological structures and functions unlikely to be recovered on the extraction site. For example, removing mature natural hardwood forests to extract underlying aggregates followed by rehabilitation to wetland or other non-forest ecosystem may require creation of new forests ("afforestation") on surrounding lands (most often marginal-value crop or pasture lands). These new forests must eventually become ecologically equivalent to the removal forests – e.g. with respect to biological diversity in the canopy, understorey, ground and soil layers – in order for mitigation to be successful.

2.1.2 The capacity for aggregate producers to develop new resources is thus tied to their capacity to successfully carry out ecosystem replacement, including identifying the critical ecological features than must be recovered and the management steps that will accomplish this. Planners must also be able to confidently predict how long this process will take, in order to set expectations and standards for success as well as to ensure that lengthy gaps between forest removal and effective replacement are compensated for (e.g. by establishing even larger or higher-quality forests than those slated for removal).

2.1.3 Such standards targeting no net loss of ecosystems are highly beneficial to Canada but can unfortunately pose major problems for companies responsible for mitigation, as substantial gaps separate existing scientific knowledge from that needed to comprehensively replace whole ecosystems. This is particularly true for complex, slow-developing ecosystems like the diverse hardwood forests constituting much of southern Canada's natural heritage (i.e. in both the Carolinean and Great Lakes forest regions). It seems unlikely that the full array of ecological structures associated with mature natural hardwood forests can be adequately recovered in forests established using conventional forestry methods, which include planting low-diversity rows of softwood tree species typically selected for their quick growth and utility in wood production rather than historical ecological importance in local forests.

2.1.4 Here, we seek to address several key questions which must be answered to obtain the knowledge necessary to successfully convert farmland to forests which are ecologically equivalent to local mature natural hardwood stands. If conventional methods are used, how similar can planted forests become to natural ones? Which properties converge, and which do not? How long does this process take? What tree-planting and thinning strategies produce the shortest time-lags to success?

2.1.5 See Appendix A for detailed background on these questions and the wider context of ecosystem mitigation.

2.2 Investigative Approach Taken

2.2.1 We comprehensively compared planted woodlots (n=36 sites) to mature naturally-occurring hardwood forests in southern Ontario (i.e. the reference, or target, state; n=5 sites). This included comparisons between planted and target forests with respect to 42 ecological features representing 6 different "layers" within forests: i) the community of canopy-forming trees (i.e. species composition, stem density, percent canopy closure); ii) the understorey plant community (i.e. shrubs, saplings, ferns, mosses, grasses, sedges, wildflowers); iii) coarse woody debris (CWD; i.e. volume and decay status of downed logs, stumps and standing dead trees); iv) the duff layer, including surface cover by fallen leaves, needles and fine woody debris; v) micro-topographic features (MTF; i.e. frequency of encountering small pits and mounds on the forest floor; capacity of these to serve as habitat for diverse plant species); vi) the physical, chemical and biological properties of the soil

2.2.2 In order to assess not just whether planted stands differ from natural ones but also how areas of similarity and distinction shift over the long timescale of forest development, we investigated planted forests which had been established at different points in time ranging from 30 to 90 years before the study began. This "chronosequence" approach is powerful because if the similarity to reference forests increases predictably with site age then the time-lag necessary for planted and target forests to become ecologically equivalent can be inferred. Sites spanned the age-gradient fairly evenly. To maintain as broad a geographic scope as possible (as well as to capture the full range of sites needed), the study region spanned a 200 km x 200 km area centred on Waterloo, Ontario (see Fig. 3.1 in the Scientific Report section of this document for a map illustrating this).

2.2.3 Planted forests additionally varied with respect to which tree species were planted and the intensity with which these were subsequently harvested or thinned over the life-cycle of the plantation. This enabled us to determine how a few broadly different conventional planting and thinning strategies were associated with different levels of similarity to target forests and different rates of similarity-emergence. For example, while many sites began as uniform plantations of a single quick-growing softwood species (e.g. red pine), others were planted as mixtures of 2-4 softwood species (e.g. pines, spruces, larches) and still others were established by intermixing 1-2 softwoods with 1-2 hardwood species (e.g. white ash, black walnut). Thinning differences consisted of sites that were either thinned regularly (e.g. row and selection thinned every 7-15 years after the first 30 years of stand development) or left under-thinned (e.g. never thinned, or more than 25 years since last thinning) for a variety of reasons (e.g. new highway construction prevented easy access for equipment).

2.2.4 One-letter codes used here for the three planting compositions are: M (monocultures), X (softwood-only mixtures) and H (hardwood-softwood mixtures). Codes for the thinning regimes are T (thinned) and U (under-thinned). Two-letter codes refer to the combination of planting and thinning treatments (e.g. MT=thinned monoculture stands). Other codes for site types include R (reference natural forests), P (the group of all plantation stands), S (the group of plantation stands initially planted with softwood species only), ST (the subset of S corresponding to regularly-thinned stands), and SU (the subset of S corresponding to under-thinned stands).

2.2.5 In addition to measuring numerous individual environmental variables at each site directly, we used an experimental approach to indirectly assess a complex suite of interconnected variables determining habitat quality. We selected two native herb species characteristic of the target forests but not present in the plantation forests, wild leek (*Allium tricoccum*) and wild ginger (*Asarum canadense*). We located wild populations of each species within the reference forests, extracted hundreds of individual *Allium* bulbs and *Asarum* rhizomes, and replanted these to new locations within the reference forests as well as to each plantation stand. We monitored survival of individuals over 3 years as an indication of how ecologically similar the new environment was to the original one. Plant death following relocation within reference sites provided a baseline expectation for impacts of transplantation shock (i.e. stress associated with the process of plant removal and replanting rather than habitat differences between new and original locations). This “phytometer” strategy of manipulating plants whose survival can be taken as indicative of particular environmental conditions has long been valuable in ecology but has never previously been crossed with a chronosequence approach. This innovative steps enabled us robustly and comprehensively determine how plant-relevant habitat similarity of created ecosystems to mitigation targets emerges over decades of forest development.

2.2.6 Beyond simply relocating phytometers from reference to plantation forests, we experimentally altered some environmental conditions at sites receiving transplants in order to test ideas about the nature of environmental differences separating planted and natural forests. To test the hypothesis that substantial differences in the chemical, physical and biological properties of soils in planted forests impedes phytometer survival, for every phytometer that was planted as a bare-root only, a second was planted as a bare root plus a small volume (1 L) of soil from the home reference forest. To test the idea that living components of the soil were particularly important in this context, for every phytometer added as a bare root combined with home soil, another was planted as bare root plus home soil which had been steam-sterilized (autoclaved) prior to out-planting. If phytometers in this treatment were to perform significantly worse than those receiving the not-sterilized home soil, it could be inferred that a lack of living soil organisms in plantation soils is responsible. Given the chronosequence of sites under investigation, this approach also allowed determination of whether soil-based barriers to phytometer survival became weaker as planted forests developed over time. Phytometer relocation was carried out in autumn 2011 and transplant survival was monitored in the summers of 2012, 2013 and 2014.

2.2.7 All reference and plantation stands were sampled identically using standard protocols either borrowed directly from the literature or adapted slightly to meet unique requirements of the study. The adult tree community was assessed using a plot-free method (the point-centred quadrant), requiring measurement of species identity, trunk diameter and distance to nearest neighbours for 80 trees per site. This enabled calculation of tree community composition, aspects of tree-size distribution, and stand density. The degree of canopy closure was assessed at random locations above the forest floor using a spherical densitometer (n=5 per site). Randomly-located 1 m^2 temporary plots were laid out on each forest floor to assess the species composition of understorey saplings, shrubs, herbs, ferns and grasses (n=21 plots per site) as well as percent surface cover by different living and non-living substrates (e.g. mosses, lichens, bare soil, leaf litter, needle litter, etc.; n = 5 plots per site). Five soil cores (2 cm diameter x 20 cm deep) were collected from each site and sent out for extensive laboratory analysis, including measurement of bulk density, moisture content, organic matter, texture, pH, and various macro- and micro-nutrients.

2.2.8 Three Y-shaped transects, each composed of three 10 m segments, were placed at random locations within each site and used to assess both coarse woody debris (CWD) and micro-topographic features (MTF). At each location where transects were intersected by a woody debris object (i.e. a stump, standing dead tree (“snag") or fallen log with diameter > 6 cm), the object’s dimensions were measured (length or height and diameter at each end) and the extent of decomposition was evaluated based on the 5-point “decay state” scale. This enabled calculation of the total volume of CWD per site as well as aspects of its ecological diversity, such as the degree to which different types and decay-states of CWD were represented. Furthermore, for every CWD object encountered, the number of vegetative species using the object as habitat was assessed. This involved assessing both how many different life forms were growing directly on the object (including trees, shrubs, grasses, herbs, ferns, mosses, liverworts, lichens, fungus and algae), and how many dissimilar organisms were present within each group (e.g. lichens could not be easily identified to species, but the number of species colonizing a given CWD object would be recorded as “3” if only black, yellow and orange coloured lichen colonies were distinctly present). The purpose of this evaluation was to help determine whether the role of CWD in providing habitat for plant diversity could be as strong in planted forests as in natural ones, and how much time is needed for such functionality to emerge.

2.2.9 MTFs such as the occurrence of small pits and mounds (often caused by tree upheaval by wind or other forces) were assessed by inspecting points spaced 1 m apart along each Y-transect (N=30 points per transect) and classifying the relative elevation at each point as either “pit”, “mound” or “matrix”. Matrix refers to the baseline or average elevation of the forest floor while pits are relative depressions with bottoms at least 30 cm below the matrix level and mounds are small hummocks with peaks at least 30 cm above the matrix level. For each point inspected, the number of vegetative species present within a 0.5 m diameter circular plot centred on the point was evaluated, using the same approach as for CWD. This enabled evaluation of not only how the frequency of pits, mounds and matrix patches compared between reference forests and planted forests over time, but also the utility of these patches in providing habitat for biodiversity.

2.2.10 Field activities were carried-out between the summer of 2011 and that of 2014. Work in 2011 focussed on locating appropriate study sites and sources of biological material for the phytometer experiment, as well as carrying-out the phytometer relocation. Most of the soil, duff-layer, understorey vegetation and canopy-layer sampling was completed in 2012, while sampling for CWD and MTFs was completed in 2013. Phytometer survival was monitored through early summer 2014.

2.2.11 We independently analyzed each of the 42 forest features using general and generalized linear models to first assess how well variation in the feature across sites could be explained by the site’s stand type (i.e. R, ST, SU, or H). Secondly, for features where R sites differed significantly from other stand types, we calculated how dissimilar each plantation was from the average R site and then investigated whether the degree of dissimilarity was related to the amount of time passed since tree-planting, alone and through interaction with stand type. The third step of the analysis looked at those features that did change significantly with plantation age and calculated for each how much time would be required before projected values in planted forests became consistent with the average observed in R stands. Fourth, for features where these projected time-lags differed among ST, SU, and H plantations, the plantation type with the shortest predictable time-lag was used to inform the best-practice strategy for meeting mitigation targets for that feature via afforestation.

2.2.12 Finally, based on the calculated time-lags, each feature was classified with respect to the estimated feasibility of accurately capturing target-forest conditions using conventional afforestation. This scale ranges from Likely (i.e. plantations equivalent to references within 30 years of planting) to Unlikely (i.e. plantation not projected to ever converge with reference forests). Possible, Achievable, and Challenging rankings were designated for features requiring up to 60, 90, or 150 years to reach reference values, respectively. The list of features, their feasibility rankings, and recommended best practices was tabulated as the Feasibility Checklist while comparative timelines until target convergence were drawn for groups of related features as the Convergence Timelines.

2.2.13 See Appendix B for a detailed account of the methods used to carry-out this research.

2.3 Major Findings

2.3.1 Of the 42 forest ecosystem features investigated, 34 (79%) are expected, in planted forests, to reach levels consistent with reference forests within 90 years of planting. The proportion increases only marginally (to 86%) when expected developments over 150 years are considered, suggesting 14% of the features may never show convergence when traditional afforestation methods are used (Table 1). However, it must be cautioned that the quality of recovered features is more important than the quantity. Similarity of the understorey plant community to reference forests, for example, represents only 2% of the list of mitigation targets, but achieving this may be the most important task of mitigative afforestation, given the goal of preserving particular species and community types intrinsic to Canada's natural biological heritage. While spontaneous recovery on this front is relatively slow, at least the goal appears to be achievable if sufficient time is allowed.

2.3.2 Soil features in the afforested farmlands showed some of the most rapid rates of convergence with reference conditions. Models of soil texture, phosphorus and cation exchange capacity created from the collected field data showed that these variables generally converge with values observed in R stands within the first 30 years of forest development. Modelled magnesium, aluminum, calcium and potassium levels exhibited gradual convergence with R over the subsequent 30 years, while soil acidity followed 10 years later (i.e. 70 years after stand-planting; Fig. 1A). Similarly, duff-layer features including surface cover by leaf litter, fallen conifer needles, fine woody debris and bare soil showed convergence with references within 60 years of tree-planting, while the volume, composition, diversity and decay state of coarse woody debris (CWD) were found to require 70-80 years (Fig. 1B). The capacity for CWD to provide habitat for diverse vegetation also increased over time, with equivalence to R forests projected to occur 95 years after stand-planting. Modelled micro-topographic features (MTF) also exhibited increasing similarity to R forests over time, with the frequency of encountering small mounds reaching target levels 90 years after stand-planting while that of encountering small pits was projected to reach target levels 130 years after planting (Fig. 1B).

2.3.3 With respect to spontaneously-colonizing understorey vegetation (Fig. 1C), modelled percent surface cover by herbs and bryophytes (i.e. mosses and liverworts) in planted forests exhibited equivalence to R forests within 30 years of afforestation. However, the identities of the plant species comprising the understorey herbs, grasses, ferns, shrubs and saplings overlapped very little between the planted and natural forests. The modelled degree of dissimilarity decreased over time, but only very slowly, showing that at least 150 years may be required for full compositional similarity to develop. Fortunately, results from the phytometer experiment indicate it may be possible to cut this time-lag in half by assisting migration of target species from target forests to maturing plantations. Both *Allium* (wild leek) and *Asarum* (wild ginger) phytometer species exhibited increased survival when relocated to progressively older plantations, with some types of mature (i.e. 80-85 year-old) planted stands supporting both phytometer species just as well as the R forests. Thus, from the perspective of two species adapted to different aspects of the target-forest understorey, reasonably old and appropriately managed plantations stands provide high-quality habitat essentially equal to that of mature natural forests, and such habitat can be readily taken advantage of once barriers to dispersal are overcome.

2.3.4 We expect that a broad swathe of the other plant species adapted to mature hardwood forests would follow suite if they were similarly relocated or dispersed, suggesting species composition of reference forests may be captured in such plantations under a multispecies assisted- immigration program. This is particular relevant where afforestation is planned to mitigate forest removal, as the forests to be removed could be rich sources for propagules of appropriate species (i.e. seeds, nuts, roots) which would otherwise be wasted. Survival patterns across sites showed that while both *Allium* and *Asarum* exhibited peak survival in similarly-aged plantation stands, *Allium* could additionally survive well in somewhat younger plantations (e.g. 50-70 years old) while *Asarum* could not. This makes sense in light of the fact that *Allium* is a spring ephemeral species, meaning it is primarily biologically active in the early spring, under the full-sun conditions characterizing hardwood forests before leaves emerge. As such, the relatively more intense light environment of younger plantations may not impose significant stress on the species. *Asarum*, on the other hand, grows all summer and fall, strongly adapted to the understorey shade but unable to flourish under a more open canopy.

2.3.5 Thus, factors largely unrelated to light (e.g. soil nutrient and water status) likely limit *Allium*, and these appear to converge with R levels after only 50-60 years. Shade levels and associated environmental features are likely important to *Asarum*, though, and these may require at least 80 years to develop to a state equivalent to the target forests. This suggests that assisting species immigration to planted forests can potentially begin at sites somewhat younger than 80, but age limitations do exist and the biology of the targeted species must be considered. Finally, although compositional similarity with respect to understorey species was shown to emerge relatively slowly, and the richness and density of such species may never reach R forest levels in planted stands under current practices, one ray of hope comes from the pattern of change over time exhibited by the ratio of native to exotic species comprising the understorey plant community. Young plantations predominantly supported an understorey of exotic herb species, however, as the modelled community developed over time, these were progressively replaced by native herbs and young trees. The proportion of total species corresponding to exotics dropped sharply from 30% in the youngest plantations to 5% in 90 year-old plantations, a low level consistent with the R forests.

2.3.6 A similar pattern was found in the responses of canopy-layer features (Fig. 1D). The degree of overhead canopy closure and the density of trees in the stand developed to a level consistent with the R forests. However, far fewer tree species were present in the planted forests, and the identities of these were not very representative of R. Over time, though, both modelled tree species diversity and compositional similarity to R increased significantly. Under best practices, the tree diversity in planted forests exhibited equivalence to R forests 50 years after planting, but an additional 55 years was projected to be required before the composition of the tree community converges completely with the R forests. This is a shorter time-lag than that for understorey composition, but still a long time to wait before mitigation success can be claimed. As no tree phytometers were used, it is unknown how well trees relocated from source hardwood forests may fair if relocated to middle-aged and older plantation forests. Further research should address this.

2.3.7 Thinned softwood (ST), under-thinned softwood (SU) and hardwood-softwood (H) stands did not differ significantly from one another with respect to 25 of the 42 forest features investigated. ST stands exhibited significantly shorter or more reliable time-lags than SU or H sites with respect to 8 features, SU stands performed best for 3 features, and H stands exhibited fastest convergence with R with respect to 6 forests features. The suggests that ST, SU and H management strategies can each do a good job recovering most of the target features over time, but focussing on planting softwoods and thinning regularly will recover the greatest number of features overall. However, failing to thin regularly can facilitate replacement for some features, while intermixing rows of hardwood species alongside softwoods can help recuperate others. Deciding which types of species to plant and how intensely to thin them thus depends on which features are the highest priority for recovery. This highlights the importance of creating clear mitigation success targets at the earliest stages of afforestation planning. Recovering the greatest number of features is likely not as important as recovering the most highly-valued features as quickly and completely as possible, but just what these features are will likely always be context-dependent and a matter of some debate among stakeholders.

2.3.8 ST management may be the fastest and most reliable way to recover most target features, but SU afforestation resulted in soil organic matter and moisture content that was closer to target levels than were ST stands. Furthermore, both the diversity and composition of tree species in the canopy layer converged with R approximately 25 years sooner where hardwood species had been initially (i.e. H plantations). In contrast, ST afforestation clearly created the most reference-like conditions with respect to the understorey vegetation and habitat features. The proportion of understorey species richness corresponding to exotics only dropped significantly with time in ST stands, for example, and both *Allium* and *Asarum* phytometers exhibited significantly shorter time-lags to mitigation success in ST than in SU stands – by 25 and 40 years, respectively. In contrast, phytometer convergence is not expected to occur at all in H plantations. Similarly, the balance of surface cover by needles and broadleaf litter observed in R stands only emerged in ST plantations. Most soil, micro-topography and coarse woody debris features developed similarity to R stands equivalently in ST, SU and H plantations.

2.3.9 The Feasibility Checklist (Table 1) indicates that within 90 years, conventional afforestation methods can successfully replicate nearly 80% of the vegetation and habitat features characterizing mature, natural hardwood forests. While this is certainly better than some potential outcomes of afforestation, it is likely that parties requiring complete mitigation of forest removal will wish to recover even more features, and do so more rapidly, completely and cost-effectively than possible under current practices. In particular, given rapid ongoing changes to biodiversity and the environment, it will be essential to capture a much larger proportion of the R forest plant communities sooner the 150 years required under current management practices.

2.3.10 Phytometer results showed that R-like habitat conditions emerged in plantation stands at a significantly faster pace than spontaneous immigration by typical R species. As such, assisting immigration of species from natural to planted forests may strongly help mitigation success. However, analysis of the environmental data suggests a few key physical features of afforested farmland may prevent many species arriving in this way from flourishing, unless steps are taken to overcome these constraints. Specifically, soils underlying planted forests tend to have less organic matter (2.75%) than soils in the reference forests (4%). Organic matter provides essential nutrients to decomposer species such as soil bacteria and fungi, which in turn produce essential nutrients for plant growth; even a 1% drop in organic matter could cause a 10% reduction in nutrient capacity. Furthermore, organic matter facilitates critical water storage and air flow through soil, both of which were found to be lacking in afforested environments in general. Soil moisture in planted forests was about two-thirds that of reference forests while bulk density (which increases as a soil becomes compacted and pore spaces fill in) was 20% higher in planted than reference forests. Each of these gaps showed no signs of closing as the plantation forests aged, suggesting active interventions early-on may be needed.

2.3.11 A second suite of related soil features may compound upon this problem. As the plantations aged, soils tended to become more acidic – a chemical change associated with changes in availability of certain micro-nutrients to plants. While the range of pH values reached was not out of line with those of reference forests (which included fairly acidic sites located on sand plains), levels of soil calcium, magnesium and aluminum showed a trajectory that led to substantially increasing *dissimilarity* to reference values as sites aged beyond 40-60 years. Changes to soil biochemistry resulting from acidification has been implicated in dramatic canopy loss in older red pine plantations ("red pine decline") and thus could easily have impacted understorey biodiversity and habitat features. Such effects were clearly not strong enough to prevent the gradual increase in habitat suitability over time detected by phytometers, but we must not overlook the possibility that greater success may be achieved faster if soils were managed differently. Addition of lime, organic matter or other nutrient amendments at strategic points during afforestation may improve capacity to achieve vegetation-based targets sooner, though field-testing of different possible approaches is needed.

2.3.12 Aspects of both the forest floor micro-topography and the physical heterogeneity created by coarse woody debris (CWD) and duff-layer properties represent forest features where early manipulation may have a big impact on achieving mitigation success more rapidly. The pattern of increasing habitat suitability over time shown by phytometers resonates closely with patterns of spontaneous gradual change exhibited by these environmental features. Specifically, surface cover by needle litter decreased while that by leaf litter increased to target levels over 60 years; the availability of variable types and advanced decay states of coarse woody debris increased to target levels within 80 years, and the frequency of encountering small soil mounds climbed to target levels over 90 years. Both of the latter trends were associated with a greater diversity of plant species directly using such features as habitat. While these correlations do not prove causation, it would not be too difficult for mitigation managers to intervene at much early stages of stand development to create mounds and import leaf litter and coarse woody debris from mature natural forests (especially of forests destined for removal are available). The impacts of such measures on reaching biodiversity targets – particularly when combined with assisted species migration – would need to be monitored and tested using an appropriate research design, but results from the present study provide hope that major progress can be achieved.

2.3.13 See the Results in the Scientific Report (section 3.4) for detailed evidence supporting this broad pattern of results, including figures illustrating the individual relationships discovered.

Table 1: Feasibility checklist for creating natural-forest conditions using conventional afforestation strategies

#	**Forest layer**	**Target feature***	**Feasibility of converging with reference forests**					**Best strategy**	**Evidence**
			Likely	Feasible	Possible	Challenging	Unlikely		
			Equivalence within ≈30 years	*Equivalence within ≈60 years*	*Equivalence within ≈90 years*	*Equivalence within ≈150 years*	*Failure to achieve equivalence*	*T=Thinned softwoods (SW)* *U=Under-thinned SW* *H=Hardwood-SW mix* *A=All strategies equal*	*Supporting figure in this Report*
1	Canopy	Canopy closure	**X**					**U**	Fig. 3.2A
2		Stand density	**X**					A	Fig. 3.2B
3		Basal density	**X**					**T**	Fig. 3.2C
4		Tree species diversity		**X**				**H**	Fig. 3.2E
5		Tree species evenness	**X**					A	Fig. 3.2D
6		Trunk-diameter diversity		**X**				**T**	Fig. 3.2G
7		Community composition				**X**		**H**	Fig. 3.3F
8	Under-storey	Cover by herbs, ferns, shrubs, saplings	**X**					A	Fig, 3.3C
9		Cover by mosses & liverworts	**X**					A	Fig. 3.3B
10		Number of vascular species per m^2					**X**	**H**	Fig. 4.4
11		Number of vascular species per site					**X**	**H**	Fig. 3.3D
12		Vascular species evenness per site					**X**	A	Fig. 3.3E
13		Community composition				**X**		**T**	Fig. 3.3F
14		Proportion of species not native to ON			**X**			**T**	Fig. 3.3G
15		Phytometer: spring ephemeral			**X**			**T**	Fig. 3.7B
16		Phytometer: shade-tolerator			**X**			**T**	Fig. 3.7D
17	Duff layer	Cover by bare soil	**X**					A	Fig. 4.6A
18		Cover by broadleaf litter		**X**				**T**	Fig. 4.7D
zz19		Cover by needle litter		**X**				**T**	Fig. 4.7C
20		Cover by fine woody debris	**X**					A	Fig. 4.6B

*The reference condition; i.e. the first line can be read: “planted stands developed equivalent canopy closure to reference forests within 30 years of planting, but only in Underthinned stands (U); details can be found in Fig. 3.2A of this Report”

21	Coarse woody debris (CWD)	CWD volume			**X**			A	Fig. 3.5B
22		CWD decay status			**X**			A	Fig. 3.5C
23		Composition of CWD types			**X**			A	Fig. 3.5E
24		Diversity of CWD types			**X**			A	Fig. 3.5D
25		Diversity of CWD-colonizing plants			**X**			A	Fig. 3.5F
26	Micro-topographic features (MTF)	Frequency of pits				**X**		A	Fig. 3.6B
27		Frequency of matrix			**X**			A	Fig. 3.6C
28		Frequency of mounds			**X**			A	Fig. 3.6D
29		Diversity of MTF-colonizing plants			**X**			A	Fig. 3.6E-G
30	Physical soil properties	Soil sand content	**X**					A	Fig. 4.6F
31		Soil silt content	**X**					A	Fig. 4.6G
32		Soil clay content	**X**					A	Fig. 4.6H
33		Soil water content	**X**					**U**	Fig. 3.4C
34		Soil bulk density					**X**	A	Fig. 3.4A
35	Chemical & biological soil properties	Soil cation exchange capacity	**X**					A	Fig. 4.6E
36		Soil phosphorus	**X**					A	Fig. 4.6D
37		Soil pH			**X**			A	Fig. 4.7E
38		Soil magnesium content		**X**				A	Fig. 3.4F
39		Soil calcium content		**X**				A	Fig. 3.4E
40		Soil potassium content		**X**				**H**	Fig. 4.6D
41		Soil aluminum content					**X**	**H**	Fig. 4.7F
42		Soil organic matter					**X**	**U**	Fig. 3.4B
	Checklist Summary	Total number of features	14	7	12	3	6	**T best for 8 features**	
		Total percentage of features	33	17	29	7	14	**U best for 3 features**	
		Cumulative percentage of features	**33**	**50**	**79**	**86**	**100**	**H best for 6 features**	
								All equal for 25 features	

*The reference condition; i.e. the first line can be read: “planted stands developed equivalent canopy closure to reference forests within 30 years of planting, but only in Underthinned stands (U); details can be found in Fig. 3.2A of this Report”

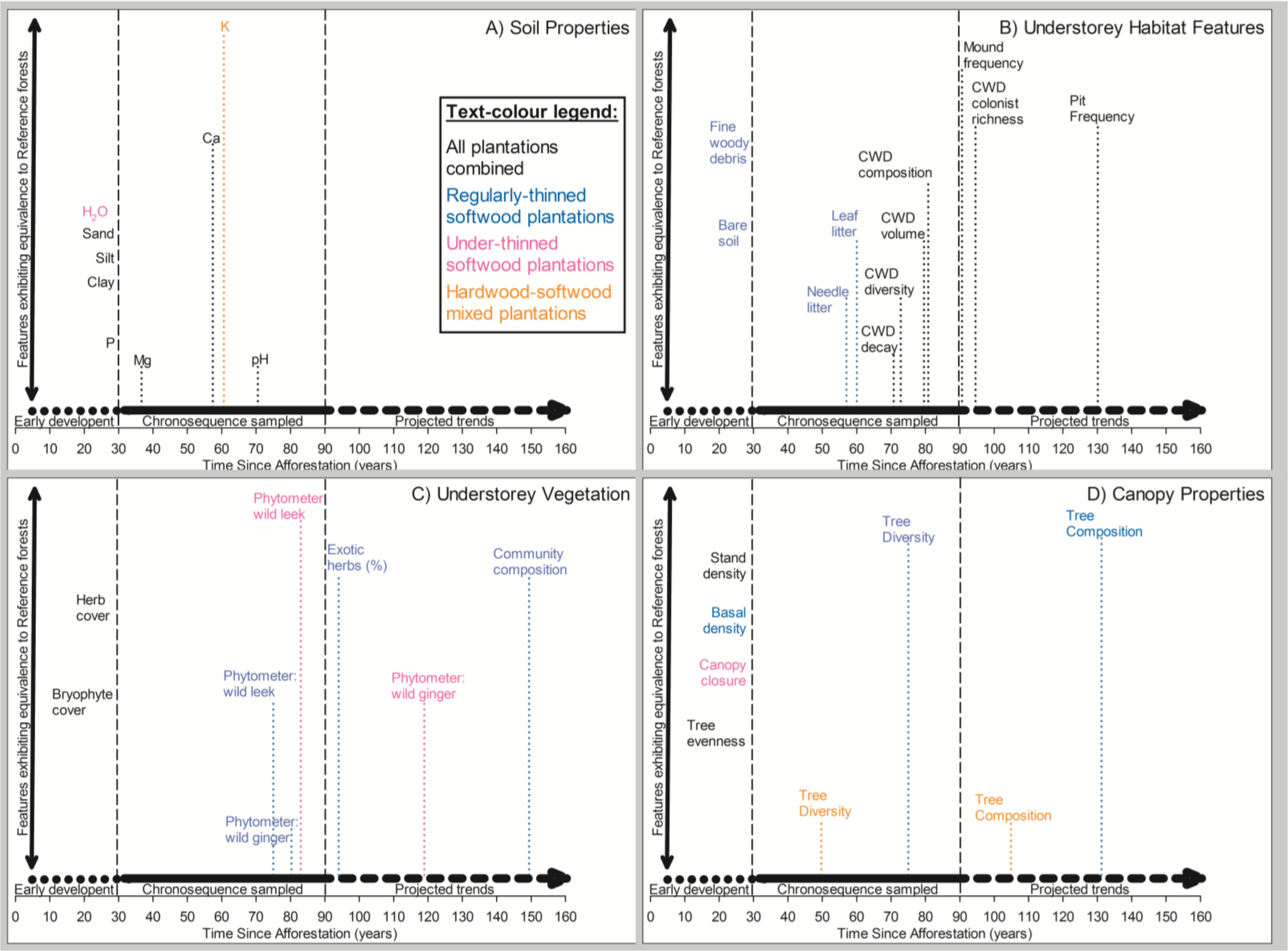


Figure 2.1 Time-lags separating stand-planting from convergence with mitigation targets spanning multiple layers of forest structure. Within each layer (panels A-D), features are presented according to when following stand planting average levels within planted stands are expected to match those of reference forests. Where time-lags differed significantly among stand types, labels are coloured to indicate this (see inset colour-key). For example, panel D shows that reference levels of tree species diversity are expected 50 years after planting in hardwood-softwood mixed stands, but after 75 years in thinned softwood-only plantations.

2.4 Significance and Applications of Results

2.4.1 Aggregate producers and other industries which must occasionally remove patches of existing forest in order to develop a resource are increasingly faced with the challenge of recreating equivalent forests at new, off-site locations. Such actions are required in cases where on-site rehabilitation can never adequately replace the removed forest (for example, if mining is to proceed below the water table), and the developer is bound by regulation to fully mitigate all negative ecological impacts of development. This makes planning and acquiring licences for new or expanded extraction operations difficult because significant knowledge gaps prevent managers from honestly claiming they have the understanding and capacity to comprehensively replicate all pertinent biological diversity, structures and functions of mature natural hardwood forests. Prior to this study, it was not clear how closely replacement forests resemble the originals for numerous ecosystem features relevant to biodiversity conservation, nor was it well understood how much time is needed or what management strategies can maximize the level of similarity and/or minimize time-lags required to achieve mitigation targets. This understandably makes effective mitigation planning difficult, and likewise makes it challenging to convince government regulators of resource industries that new planted forests can be as good, or even better, than those which must be removed.

2.4.2 The prime significance of the Afforested Environment Study is that it presents strong empirical evidence which helps fill these knowledge gaps, and demonstrates that superior-quality forests can indeed develop from conventional afforestation, although different features require different lengths of time to develop, and sometimes respond differently to different management strategies. This highlights the importance of understanding and agreeing upon specific mitigation goals at the earliest stages of afforestation planning, acknowledging that some aspect of similarity to target forests may need to be sacrificed if it means improving the likelihood of achieving other aspects deemed to be of higher priority. The AES also shows that while conventional methods of afforestation can do a good job of recovering many properties of target ecosystems, some high-priority features are difficult to replicate. Fortunately, our analysis revealed several specific steps for adapting conventional methods to potentially meet these elusive targets faster, more fully, or with greater cost efficiency. The direction that research must take to field-test the validity of such practices is also informed by the patterns discovered to date.

2.4.3 Planted forest showed a general overall pattern of eventually recovering 86% of the 42 individual forest features investigated, while 14% of these features are unlikely to ever be recovered under conventional afforestation practices. Results suggest nearly one-third of forest features converge with target values within the first 30 years of stand development; one-sixth converge between 30 and 60 years post-planting, and another third of features converged between 60 and 90 years post-planting. Development between 90 and 150 years post-planting is projected to be associated with convergence of a numerically small proportion of investigated features (7%), but the biological significance of these is immense as they relate to the species composition of the canopy-forming tree community (requiring 130 years) and the understorey plant community (requiring 150 years).

2.4.4 Features developing within the first 30 years correspond to many physical soil properties (e.g. soil texture), some duff-layer features (e.g. fine woody debris), and understorey and canopy features related to general vegetation cover but not the identities of species contributing to this cover.

2.4.5 Features converging between 30 and 60 years correspond to aspects of canopy structure including average trunk-diameter, the diversity of trunk-diameter classes, and tree species diversity present. The balance of needle and leaf litter in the duff layer also aligns with reference forests over this period.

2.4.6 Numerous critical habitat features converged with reference forests between 60 and 90 years post-planting, including the frequency of encountering small forest-floor mounds (which were found to provide important habitat for diverse plant species) and the volume, composition and diversity of fallen logs, stumps and snags exhibiting different degrees of decomposition. It was also during this stage that non-native understorey herbs dropped from representing 30% to 5% of all understorey species. Phytometers indicated multiple plant-relevant habitat conditions become indistinguishable from reference forests during this duration as well.

2.4.7 Features which are apparently not on a trajectory toward eventual recovery in planted forests include the number and evenness of understorey species (at both square-metre and whole-site scales) as well as soil organic matter, bulk density, and some micro-nutrient concentrations (e.g. magnesium, aluminum).

2.4.8 For 60% of features there were no discernable differences among the main strategies investigated, but for the remaining features the afforestation strategy was important. On the whole, stands planted with softwood species only and thinned regularly showed the fastest and most reliable convergence with target forests over time, out-performing under-thinned softwood-only stands as well as mixed hardwood-softwood stands. This pattern held for the balance of needle and leaf litter in the duff layer, the average girth of trees in the canopy layer, and the composition, diversity and proportion of native species in the understorey.

2.4.9 Phytometers responded strongly to afforestation strategy, with thinned softwood plantations developing habitat similarity to target forests more rapidly than under-thinned plantations while mixed hardwood-softwood plantations failed to develop similarity from the phytometer perspective. In contrast, with respect to the diversity and composition of the canopy layer, mixed hardwood-softwood plantations showed significantly more rapid emergence of similarity to target forests than the other strategies. The same is true for the overall number of species present in the understorey. Under-thinned softwood-only plantations showed a slight advantage over the other strategies with respect to approaching target levels of canopy closure and trunk-diameter diversity. Afforestation initiatives placing high priority on recovering canopy-layer features may thus favour a strategy that includes planting hardwood species. However, efforts to replicate understorey features and communities would be more successful if just softwoods were planted and progressively removed over the course of stand development.

2.4.10 The chief significance of these findings is that they squarely challenges two common assumptions about hardwood forest restoration: that planting hardwood species is needed to comprehensively replace hardwood forest ecosystems, and that after planting, nature should be allowed to “take its course” with minimal human intervention. In fact, hardwood forests can be best replaced by planting quick-growing softwood species which are allowed to mature – creating ideal habitat conditions for native understorey and next-generation canopy species in the process – but which are subsequently progressively removed. What remains are diverse, spontaneous vegetation communities well-adapted to the forest understorey but assembling faster than the long duration typically needed for planted hardwood stands to mature. While calls for alternatives to conventional afforestation have frequently been made, the AES showed that several goals of ecological forest restoration are met admirably using just such methods. Alternative approaches focussing on introducing hardwood species or reducing timber harvesting may actually delay or prevent achievement of mitigation goals.

2.4.11 The best afforestation practices for mitigating ecological impacts of forest removal will likely utilize aspects of conventional planting and thinning regimes, but also develop adaptations aimed at overcoming barriers to similarity discovered here. Forestry-based strategies designed to maximize production of marketable wood products may emphasize species which are often not major components of target forests, but the expert matching of species to environment to management which is a hallmark of good forestry may compensate for this. Management steps promoting optimum growth of tree height and girth, for example, appear to instill planted tree cohorts with strong functionality as a “nurse crop”, actively supporting spontaneous regeneration of high-quality natural hardwood forest in the understorey. This functionality may be reduced if planted species are not progressively removed or if slower-growing hardwood species are relied upon.

2.4.12 Part of the success of this “nurse crop effect” may be the rapid rate at which high semi-closed canopies emerge, projecting the right balance of shade patches on the forest floor to support establishment of diverse tree, shrub, and herb species emigrating from natural hardwood habitats in the region. As planted softwood species tend to grow tall faster than hardwoods but produce a more open canopy at full height, dense plantings of quick-growing softwoods adapted to open-field conditions and exposed to regular thinning may create very similar understorey habitat conditions to natural hardwood stands, but faster than if typical hardwood-forest species were to be planted. However, high levels of needle litter and low levels of broadleaf litter relative to the reference stands may constrain habitat suitability unless suitable surface-cover amendments are imposed

2.4.13 This is consistent with our observation that many aspects of similarity to target forests did not begin to emerge in planted forests until after the needle-to-leaf litter ratio inverted (i.e. cover by leaves became greater than cover by needles), around 60 years after stand planting. Broadleaf litter accumulated slowly as hardwood trees and herbs become established in the community, but this time-lag may be shortened by importing deposits of appropriate litter from other sources. In the mitigation context, the most logical source of such material would be the existing mature natural hardwood forests which are slated for removal. Collecting bulk quantities of duff-layer substrates from such sites and transferring 1 m^3 bundles to even 5-10 locations per hectare within young planted forests, allowing nature to disperse, could go far towards creating an appropriately patchy duff layer capable of supporting diverse species adapted to hardwood forest understories.

2.4.14 The softwood nurse-crop effect may work because a particular balance of light and shade, as well as soil moisture and nutrient conditions, is achieved in the understorey. Planting rows of hardwood species may fail to generate such appropriate habitat conditions because the slow growth of such species pushes the balance too far in favour of high light on the forest floor. While not necessarily a problem at the scale individual trees, entire rows of a species that grow substantially slower than adjacent rows produces lasting canopy openings with potentially broad impacts. Such conditions can promote dominance of grasses, including exotic invasive species, in the inter-row space, which can lend to the stand resembling savanna more than proper forest habitat. To avoid this, managers may wish to avoid planting hardwoods initially, or at least intermix them *within rows* of softwoods.

2.4.15 Several features typical of reference-forest soils were discovered in the soils underlying planted forests; some of these may have been present even prior to afforestation while others likely developed gradually (but some faster than others) as planted stands matured. Afforestation managers should endeavour to work effectively with the target features which are relatively easy to achieve (e.g. soil texture, cation exchange capacity) but also take appropriate steps to encourage those emerging at a slower pace (e.g. soil potassium, pH). More forceful interventions may be needed for those features which may not emerge at all (e.g. soil organic matter, bulk density) or are on a trajectory toward increasing *dissimilarity* to targets after initially over-shooting the mark (e.g. soil calcium, magnesium and aluminum). Amending the soil carefully through application of mild fertilizers and organic materials prior to stand-planting should help accelerate the pace at which soil similarity to targets. It would likely be even more effective, though, to combine initial amendments with at least some follow-up applications over the course of stand maturation, given that the features of concern do not reach potentially problematic levels until several decades after tree planting.

2.4.16 Altering soils in an established forest clearly requires different methods than used in open fields prior to planting; doing so in a completely uniform manner (as in ploughing a field) is probably infeasible. Fortunately, there are many reasons to believe that having a homogeneous soil profile throughout the forest is not desirable anyway, as the presence of gradients or variability in soil moisture and nutrient conditions tends to promote enhanced biological diversity at multiple scales. Thus a useful approach for amending soils in the understorey of existing woodlots may be similar to that recommended for duff-layer materials; deposit discrete bulk quantities of high-quality soil (e.g. 1 m^3 bundles) at multiple (non-edge) locations throughout forest floor, making some effort to intermix with surrounding soils but also allowing for natural forces to do the brunt of the labour. Added soils should be rich in organic matter and may be lightly fertilized (especially with potassium) and fortified with calcium and magnesium (crushed dolomitic limestone fines could help here), but not so much as to impede the slight soil acidification needed to bring plantations in line with target forests. Topsoil from commercial sources may be acceptable, especially if guaranteed to be “weed free”. However, in a mitigation context, the best source would likely be the existing natural hardwood forests destined for removal.

2.4.17 Planted stands contained variable sizes and numbers of coarse woody debris (CWD) objects, which translated to varying volumes of CWD produced per hectare of forest cover. Each CWD object encountered was classified as one of 15 possible “CWD species”, depending on its type (fallen log, stump, or standing dead tree) and its level of decay on a 5-point scale. We discovered that in planted stands, the total volume of CWD, the average decay status and the diversity and composition of CWD “species” present per site each gradually came to match expected values for reference forests 70-80 years after tree planting. The capacity for individual CWD objects to function as a substrate for the growth of diverse plant species also increased substantially over stand development, projected to converge with reference forests 95 years after tree planting. CWD is functionally important in forests, providing habitat for biodiversity and also playing important roles in nutrient, water, and fire cycles; its status is also an important indicator of overall forest maturity, as it depends on multiple factors that correlated with forest age and development.

2.4.18 While it is encouraging to discover that conventional afforestation methods can reliably replicate multiple CWD components provided that enough time is allowed to pass, managers may wish to achieve target similarity with respect to CWD features similarity sooner than the 70 years predicted here. This may be achieved by retaining appropriate quotas of fallen logs, high stumps and standing dead trees during early as well as later-phases of selective stand-thinning, although several decades would still likely be required to produce and then decompose sufficiently large volumes of CWD locally. The whole process may be jump-started, however, if appropriate selections of CWD are imported from other forest locations – especially target forests planned for removal. Care should be taken to relocate CWD spanning a range of size and decay states, but favouring larger and more decayed samples if available, as these are the slowest to develop naturally. If CWD objects are relocated directly, with minimal physical disruption and storage period, at least some of the species using CWD as habitat may continue to do so within the planted forests. As with organic litter and soil amendments, effort need not be spend ensuring a uniform distribution of CWD; sporadic small clusters (e.g. several CWD objects with total volume of 1-10 m^3) at dozens of random locations per hectare of forest floor should be sufficient to markedly increase similarity to reference forests.

2.4.19 The occurrence of micro-topographic variation in the form of small pits and mounds throughout the forest floor is also both an indicator of forest maturity and potentially a determining factor in the capacity for mature forests to support diverse and characteristic understorey species. We found that the number of plant species present on any given small patch of forest floor was generally related to the overall number of species on-site, but a greater proportion of site richness established on and around mounds than on level ground or in pits. However, as different species tended to colonize pits and mounds, having both pits and mounds in the environment should support greater diversity at the whole-stand scale. The frequency of encountering both pits and mounds is greater in reference forests than in plantations, but both pit and mound occurrence increased significantly as planted stands developed over time, with target-convergence projected to occur 90 years after stand-planting for mounds, and 130 years after planting for pits.

2.4.20 That these important components of mature natural forests do develop spontaneously in planted forests is encouraging, however, the projected 90-130 year time-lags may be too lengthy for mitigation planners. These may be shortened considerably if forest managers develop such features artificially, but this can require considerable labour if the natural process is to be mimicked accurately (i.e. uprooting large trees and leaving holes that become the base of the pit, while dislodged soil settles to form an adjacent mound). However, given that we discovered mounds but not pits were associated with high plant diversity, an effective shortcut may be to focus on creating mounds only. This could be done effectively if combined with importation of organic litter, topsoil and coarse woody debris; these materials could be mixed and used as the foundation for mound construction simply by refraining from spreading too thoroughly after depositing them (as recommended above), and “topping up” using small volumes of topsoil from the surrounding forest floor.

2.4.21 Spontaneous species immigration and establishment within the understorey of planted forests may eventually generate communities resembling those inhabiting target forests, but this process is expected to be very slow and result in relatively low species diversity no matter how much time passes. The single most important finding of the AES comes from our experimental relocation of *Allium* (wild leek) and *Asarum* (wild ginger) roots from their original habitats in the reference forests ("home") to new locations in the understoreys of planted forests. As transplant survival was very low in young and middle-aged plantations but relatively strong in old plantations – equally good, in fact, as within the home forest – the habitat conditions that matter most to target understorey vegetation must develop at a much faster pace than the rate at which target forest species immigrate to planted stand spontaneously. Although 150 years of natural succession may be required for understorey communities in planted forests to match those in reference forests, habitat conditions are actually ripe for colonization within about one-half of this timespan. Human-assisted immigration of target understorey species to developing natural forests may thus make major headway in closing the gap separating stand-planting from mitigation success.

2.4.22 Such an immigration-assistance can only work if ecologically-informed strategies are followed; introducing species too early will be wasteful, as conditions in young plantations are unlikely to support desired species unless substantial changes are made to the local environment through habitat creation. Perhaps most intriguingly, the planted forests that *least* resembled target hardwood forests (from the perspective of the relocated plants) were those where hardwood species were planted. Although a shorter time-span of hardwood-softwood plantations was available for study (32-64 years, compared to the 30-90 year span for softwood-only stands), neither phytometer species survived well in these plantations regardless of stand age. This may be due to increased competition from taller grass species, which tended to be abundant under the more open canopies of young and mid-aged hardwood-softwood plantations. Older mixed plantations appeared less defined by open-canopy rows but still failed to support phytometers, possibly because competitive grass species can exert long-term influence on community dynamics once they achieve an early foothold in an ecosystem. Regardless, this pattern suggests afforestation projects placing high priority on creation of high-quality habitat for target understorey communities should avoid planting hardwood trees, especially in rows and at the outset of afforestation. The potential effectiveness of under-planting hardwoods within maturing softwood plantations requires further study.

2.4.23 Thorough investigation of factors associated with high transplant success in the group of sites where hardwoods were *not* planted reveals that regular stand thinning may contribute importantly to reducing time-lags for habitat development. *Allium*, which is most sensitive to environmental conditions manifesting before canopy leaf-out, could not distinguish between reference forests and plantations 80 years of age or older, provided planted stands had been regularly thinned; if under-thinned, sites need to be at least 95 to provide equivalent habitat to reference forests. *Asarum*, representing herbs that grow all summer long by tolerating moderate or deep shade, could not distinguish between reference forests and softwood plantations established 80 or more years prior to the study, provided they had been thinned. Conversely, under-thinned sites would need to develop for 130 years before they could provide equivalent *Asarum* habitat to reference forests.

2.4.24 Interestingly, the types of softwood trees planted played a role in this effect. When thinned and under-thinned softwood plantations were additionally classified as either monocultures of a single species or mixtures of 2-4 softwood species, phytometer in thinned mixed plantations outperformed all other groups. However, phytometer survival in under-thinned monoculture plantations was actually greater than that in thinned ones. Multiple understorey habitat features – including the relatively high levels of understorey shade needed by *Asarum* – should therefore be promoted by afforestation managers through a combination planting multiple softwood species and thinning regularly. However, if monoculture plantings are to be used, the intensity of thinning should be reduced. The reason for the beneficial effect of *not* thinning in monoculture plantations may be that canopies close less-tightly when all trees have the same growth form, and thinning opens-up this already-sparse canopy too much. In contrast, when neighboring trees exhibit even slightly different growth forms, the density of inter-lacing branches in the canopy is relatively high and thinning produces a more favourable pattern of light reaching the forest floor.

2.4.25 Aside from the light environment, soil development over the duration of stand maturation likely has significant impact on a stand's capacity to support target understorey vegetation. As multiple soil variables correlate with both site age *and* increasing phytometer survival, it is difficult to determine which soil features are most important to target herbs. Fortunately, the experimental addition of different soils during phytometer relocation shed some light on this. For the most part, phytometers responded the same way to a given site regardless of whether they were planted as bare roots alone or bare roots in combination with soil from the home environment. The exception was in the group of plantations that had been planted with multiple softwood species and thinned regularly – i.e. the same group which showed the fastest emergence of *Asarum* habitat. Here, *Asarum* survival approached reference levels more rapidly where soil from the reference forests had been added. Factors beneficial to target vegetation are thus present in reference forest soils but reduced or absent in the soils of planted stands. When "home" soil is introduced to plantations alongside multi-species tree plantings and regularly thinning, suitable habitat for *Asarum*, and likely the numerous species it represents, emerged fastest.

2.4.26 While several physical and chemical components of the target forest soil could potentially be responsible for this beneficial effect of home-forest soil addition (i.e. organic matter, bulk density and pH are likely candidates), we were able to rule out living soil components as being responsible in this case. This is because the beneficial effects of soil addition were only observed for the group of phytometers receiving soil which had been steam-sterilized prior to planting; *Asarum* receiving the not-sterilized home soil responded the same as those receiving no soil at all. This reinforces the recommendation that topsoil transfer from target forests be used where possible to promote development of target habitat conditions in planted forests, but suggests further efforts to improve soil conditions should focus on overcoming physical and chemical rather than biological differences in soil composition.

2.4.27 Interestingly, the reverse is true in softwood plantation that were not regularly thinned. Regardless of the composition of trees planted (i.e. mixtures or monocultures), under-thinned plantations approached reference conditions significantly faster where not-sterilized soil had been added than where either sterilized soil or no soil had been added. Living components in the reference forest soil (e.g. symbiotic bacteria and fungi) beneficial to target vegetation are thus apparently lacking in under-thinned planted forests. Efforts to quicken the pace of habitat generation in such stands should thus focus on either preserving or enhancing biotic components in any topsoil amendments added. The reason why biotic soil conditions are more limiting to *Asarum* survival in under-thinned than in thinned plantations is currently unknown, but could be tied to the numerous other soil, microhabitat and vegetation features that tend to emerge most rapidly in the regularly thinned stands.

2.5 Recommended Best Practices

2.5.1 As a whole, these findings make clear that although conventionally afforested farmlands tend to develop along trajectories that eventually result in high similarity to natural hardwood forests, there is no single fool-proof method for maximizing the pace of such development. Rather, multiple environmental and management factors interact with each other to create a complex situation whereby particular management steps can accelerate target convergence for some goals but not others, or under some environmental conditions but not others.

2.5.2 A major take-home message for managers is that the range of goals for a given afforestation project must be decided upon early in the management planning process, and considered carefully in light of any internal or external constraints that may be placed upon the project. Different strategies may be required, for example, depending on whether rapid achievement of canopy versus understorey community composition is the more pressing goal, or whether available planting stock will include many or only a few species.

2.5.3 That being said, some shared trends among results for different features and management contexts suggest a few key strategies in combination with each other should be highly successful across a range of common project contexts. These recommended best afforestation practices are as follows.

2.5.4 Plant multiple tree species per stand, emphasizing softwoods; if hardwoods are to be used, they should be added sporadically within rows rather than as rows.

2.5.5 Amend the soil to reduce bulk density and increase concentrations of organic matter, calcium, magnesium and potassium; this may be facilitated prior to planting by scraping fields and ploughing-in appropriate sources of soil organic matter (e.g. plant-based compost or rich topsoil from donor forests) and possibly potassium, calcium and magnesium supplements including limestone fines. Amendments should be periodically repeated during stand maturation (e.g. every decade or so) but at a smaller scale, using replicated random locations and minimal machinery to minimizing disturbance to tree growth.

2.5.6 Manufacture sources of variability in forest-floor microtopography; this should be attempted using grading equipment to create numerous small depressions and mounds across the forest floor (i.e. undulations ranging from 10-30 cm above and below grade). Such management must balance maximizing surface heterogeneity against the need to maintain the field for relatively easy access by forestry equipment, and may need to be reapplied at smaller scales periodically throughout stand maturation with an emphasis on creating mounds more than pits.

2.5.7 Import coarse woody debris such as fallen logs, stumps or small standing dead trees shortly after tree planting, and again at intermediate stages of stand development; woody debris objects should be selected to span a range of volumes, types and degrees of decomposition, ideally extracted from afforestation reference sites planned for removal; fallen logs, stumps and snags should additionally be retained as they develop naturally in the maturing plantation.

2.5.8 Import appropriate sources of duff-layer organic litter, aiming for fallen leaves from herbs and broadleaved trees over conifer needles. Repeat this throughout stand development; existing mature forests slated for removal may be a good source for the necessary material.

Where availability of materials is low, that which is available may still have important beneficial effects when applied at small scales.

2.5.9 Apply row and selection thinning strategies (with retention of all tree components other than the main bole) to planted stands beginning 20-30 years after establishment and repeating every 10-15 years, Leave some forest patches minimally thinned to preserve a range of habitat conditions , especially within softwood monoculture plantations.

2.5.10 Facilitate immigration of understorey species from mature hardwood forests to afforested environment, but do so at an appropriate leverage point in stand development (i.e. a confluence of suitable developments in the soil, light environment, neighbouring species, and management context). If conventional planting and thinning methods are to be applied, an ideal opportunity for leveraging stand development using species introductions arises around 80 years after tree planting, although moderate success may be achieved as early as 50-60 years after planting. Interventions at earlier points are unlikely to succeed unless effective alternative afforestation methods are developed. Taking the recommended steps for promoting development of target habitat condition – i.e. importing soils, litter, and coarse woody debris from target forests slated for removal, or creating topographical heterogeneity – may push forward the ideal time-window for species introduction, but this needs to be tested experimentally.

2.5.11 A variety of approaches may be successful when introducing species at the chosen leverage point. Adding seeds from target species – selected based on representativeness, availability, likelihood of success, and importance to ecosystem processes or natural heritage – may be the most economical means of promoting rapid establishment of multiple species across large areas. However, adding established plants – especially if in a stage of dormancy, such as a bulb – would be more likely to ensure growth of particular species and species combinations at particular locations. Creating small "islands" featuring extensively altered habitat conditions and numerous introduced target species may be effective for establishing footholds for target communities in the afforested environment, which may then spread over time, but research is needed to determine whether or not this would be more effective over the long term than seeding large areas.

2.5.12 A potentially valuable alternative strategy is to avoid targeting particular species for direct planting and instead focus on maximizing the diversity and abundance of native plants that can be elicited following relocation of bulk quantities of topsoil (including the embedded seed/propagule bank) from target forests to new afforestation sites. Strategies for replicating target habitat condition will be needed in conjunction with this, but field testing is still needed to determine the relative effectiveness of different potential habitat-mimicry methods at management scales.

2.5.13 Importantly, although afforestation managers may be most interested in learning how to establish target-forest conditions in newly planted forests as rapidly as possible, there is no reason that mitigation steps should be constrained solely to planting new forests. Taking steps to improve the quality of existing planted forests is also a valid means of mitigating impacts of forest removal, especially if a strategic "push" can shift an older plantation from minimal to maximal ecological resemblance to target forests. Pursuing strategic improvements to both newer and older plantations simultaneously may be especially effective for robustly mitigating impacts of forest removal at the landscape scale.

Part 3: Scientific Report

3 Evaluating time-lags and leverage points in biodiversity mitigation via afforestation

Abstract: Mitigating industrial impacts on biodiversity by setting-aside reconstructed ecosystems can only work where long-term outcomes of interventions are predictable. This is rarely the case for slowly-maturing ecosystems including temperate deciduous forests. We tested an innovative combination of phytometer and chronosequence methods to rapidly forecast the success of off-site forest replacement on conventionally afforested farmland. We compared old-growth stands to 30-90 year-old conifer-dominated plantations, assessing environment, plant diversity and survival of characteristic understorey herbs relocated from target forests. Results project that under careful stand-planting, thinning and soil management, plantations develop target canopy, microhabitat and phytometer-support properties within 80-100 years – half the time required for understorey composition. Mature plantations may therefore function as leverage points, capable of fast-tracking forest replacement under assisted immigration. Fundamentally, crossing phytometer and chronosequence strategies can drastically improve predictability of offsets in any ecosystem.

Regulating mining and energy production for more sustainable management requires policies that enforce compensation for ecological impacts; most effectively, this is accomplished by setting aside restored ecosystems (*1-3*). This demands realistic projections of the timespan and management required for replacement ecosystems to function equivalently to the originals (*4-6*), which is vexing for slowly-maturing ecosystems like temperate deciduous forests (*7, 8*). Outcomes of forest-creation are difficult to predict because multiple trajectories are possible, the influence of management is unclear, and confirming replacement success may take decades (*9-12*). Conventional afforestation relies on homogenous planting and regular thinning of a low diversity of often nonindigenous conifer species. Do such departures from natural succession prohibit ecosystem structures consistent with mature natural forests – including canopy, understorey and microhabitat composition – from ever emerging? Or does planting then removing quick-growing trees actually promote eventual emergence of these target features through effects of habitat-filtering and competitive-release? Conflicting reports support both the former (*13-17*) and the latter (*18, 19*) scenarios, but answers remain elusive because of the long timescale of forest succession (*20*). Solutions may lie with the chronosequence approach, which compares sites similar in all regards other than time-passed since tree-planting (*21, 22*). Predicting community assembly this way is difficult though, because compositional differences may reflect stochastic dispersal patterns rather than habitat suitability (*23*). Intriguingly, this limitation may be overcome using phytometers: habitat-sensitive plant species typically transplanted to test sites and monitored as indicators of habitat similarity to the home ecosystem (*24*)). While phytometers have been successfully incorporated into studies of primary succession (*25, 26*), they have never been utilized in forecasting the extent and timeline of mitigative forest replacement.Our innovation was to cross phytometer and chronosequence approaches to determine empirically the timespan and conditions necessary for southern Ontario’s heritage hardwood biodiversity (*27*) to emerge within conventionally afforested farmlands . This technique rapidly evaluates the compensatory capacity of biodiversity offsets over long timescales and projects critical time periods (or leverage points) where assisting immigration of target species should be most successful (*4, 6*). The broad implications are that forest replacement can be more effective, resource companies can legitimately claim likely success, and policymakers and the public will be assured that correct interventions are being implemented to mitigate ecological damage. This 'triple win' can be a powerful approach for reconciling the often-clashing goals of resource extraction and ecological values (*28*).

We compared 5 mature natural hardwood stands (targets) to 36 softwood-dominated plantations established between 30 and 90 years ago (Fig. 3.1). We evaluated both spontaneously-developing forest structures and survivorship of two characteristic understorey herb species – *Allium tricoccum* (wild leek) and *Asarum canadense* (wild ginger) – that were experimentally relocated from target forests to plantations as phytometers. Study sites were drawn from a pool of 123 potential sites representing a land base of 40,000 km^2, selected at random but following criteria such that each of three tree composition strategies (MO=monoculture of a conifer species; MX=mixture of 2-4 conifer species; HW= mixture of 2-4 conifer and hardwood species) and two stand-thinning regimes (regularly vs. rarely/never thinned) spanned the age gradient (See Appendix B for details). These qualities were selected because theory suggests trees with different traits have different capacities to produce shade, soil and litter conditions favourable to target vegetation, while thinning provides diverse growth opportunities through enhanced environmental heterogeneity and competitive release (*29, 30*).By elucidating relationships among tree composition, thinning and successional trajectories, we were able to project and infer strategies for minimizing time-lags in forest replacement. Because *Allium* and *Asarum* were introduced experimentally – as bare roots only, or bare roots combined with either sterilized or unsterilized soil from target forests – comparing effects of soil manipulations across the age gradient helped quantify relationships between time-lags and biotic versus abiotic components of soil development.

Figure 3.1. Sampling locations in the Afforested Environments Study. Research forests consisted of reference mature natural hardwood stands (REF, n=5), and plantations established 30-90 years before sampling under different planted composition and thinning strategies (n=36). See inset site-label legend for details. Note that some locations are too close together to differentiate visually at the spatial scale shown.

3.3 Methods Overview (Scientific Report) – Fig. 3.1

3.4.1 Spontaneous canopy development

Stand designation as a Reference(R) forest or a Thinned (T), Under-thinned (U), or mixed Hardwood-Softwood (H) plantation explained significant variance in the degree of canopy closure ($F_{(4,37)}$=3803.70, $P < 0.0001$, R^2=0.16). However, when just plantations were analyzed, time since afforestation had no effect. The least-squared mean value for canopy closure in R stands fell into a 95% confidence interval (95%CI) of 86-94%, which tended (i.e. P=0.0821) to be slightly greater than that in T stands (83-88%). Canopy closure in R was not significantly different (i.e. P>0.30) from that in U (84-90%) or H (86-92%) plantations (Fig. 3.2A).

Similarly, tree density (i.e. the number of woody stems > 2 cm diameter per hectare) was highly variable among plantations and unrelated to plantation age (P>0.67). Stand type (H, T, U or R) explained significant variance when all sites were analyzed together ($F_{(4,36)}$=80.59, P<0.0001, R^2=0.08), but there were no significant differences (i.e. P>0.17) between R stands (809-1911 stems/ha) and T (1337-1973 stems/ha), U (1565-2308 stems/ha) or H (1332-2153 stems/ha) stands (Fig. 3.2B).

Stand basal density (i.e. the total area of forest floor per hectare occupied by the cross-sectional area of tree trunks) was also highly variable, and somewhat explained by stand type ($F_{(4,37)}$=77.84, P<0.0001, R^2=0.17). Basal density tended to be greater in R (63-104 m^2/ha) than in T (50-73 m^2/ha), U (39-67 m^2/ha), or H (40-70 m^2/ha) stands (i.e. P=0.12, 0.03, and 0.06, respectively) (Fig. 3.2C). Time since afforestation did not explain significant variance in basal density when plantation stands only were analyzed.

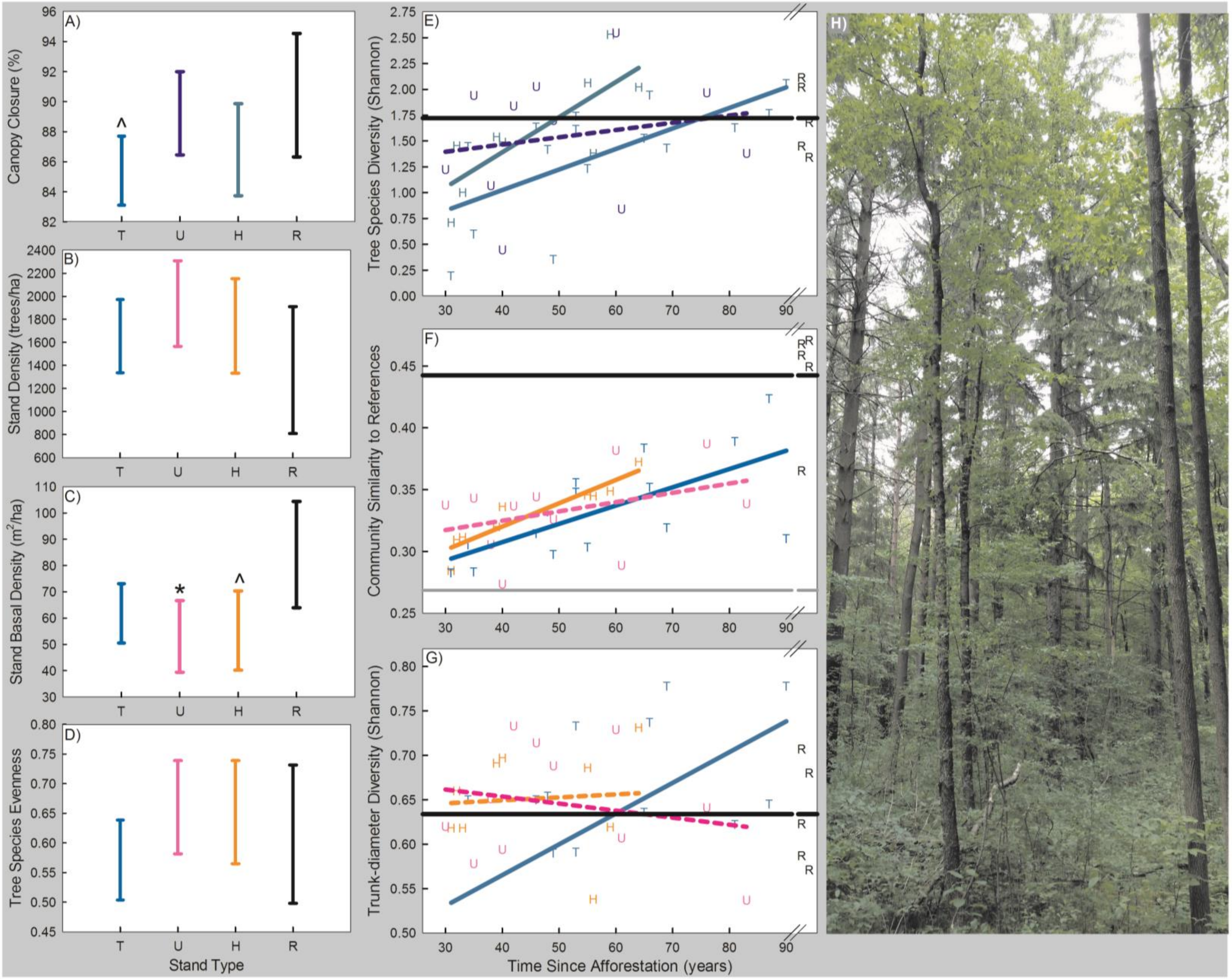
Tree species evenness showed a similar pattern to stand density: highly variable within stand types, with no significant differences between any plantation type and the reference forests (Fig. 3.2D). In contrast, the Shannon diversity of the tree community in plantation stands was best explained by a general linear model considering the effect of stand type and its interaction with time since afforestation (mode $F_{(6,29)}$=55.87, P<0.0001, R^2=0.36) (Fig. 3.2E). This analysis revealed significantly increasing diversity over time in T (t=2.83, P=0.0084) and H (t=2.55, P=0.0164) but not U (t=0.78, P=0.4445) plantations. Diversity increased more rapidly in H than in T plantations, reaching lsmean diversity for R stands (1.72; 95% CI=1.22-2.22) 50 years after tree-planting in H but 75 years after tree-planting in T.

A similar pattern was found for the degree of compositional similarity of plantation tree communities to R tree communities, as gauged using the Bray-Curtis similarity coefficient. Variance in the mean coefficient value was best explained by stand type and its interaction with time since afforestation (model $F_{(6,29)}$=770.08, P<0.0001, R^2=0.41), revealing that similarity increased significantly over time in T (t=3.55, P=0.0013) and H (t=2.39, P=0.0236) but not U (t=1.40, P=0.1710) stands (Fig. 3.2F). The mean value calculated for a community with no species in common with any of the five reference forests was 0.26 while that calculated for a community identical to any one of the reference stands was 0.41. Against this background, the modelled value for H plantations climbed from 0.30 to 0.36 across the 31-66 year age-gradient available (67% of the maximum similarity value) while that for H plantations climbed from 0.29 to 0.38 (80% of maximal similarity) across the 31-90 year age-gradient. Given the shorter time-span available, the rate of increasing similarity was greater in H than T, and we projected 105 and 131 years of development are respectively needed for each stand type in order to achieve complete similarity to reverence forests. An alternate, non-parametric and multivariate approach to assessing community similarity through Canonical Correspondence Analysis (CCA) found a broadly similar pattern of gradually increasing compositional similarity over time (see Fig. 4.4 in Appendix C).

Variance in the diversity of trunk-diameter size-classes in plantation stands was likewise best explained by a model including the interaction of stand type with plantation age (model $F_{(6,29)}=365.56$, $P<0.0001$, $R^2=0.25$), but in this case the value of the diversity index only increased with time in T plantations ($t=2.98$, $P=0.0057$) while in U and H plantations size diversity was unrelated to time since afforestation ($t=-0.53$, $P=0.5991$ for U; $t=0.15$, $P=0.8803$ for H) (Fig. 3.1G). Trunk-diameter diversity equivalent with R stands (0.55-0.71) was reached in T plantations after 60 years of stand development, but in U and H plantations diameter diversity was not significantly different from R regardless of plantation age. Visually, the profile and composition of trees in plantation stands 60 years or older were difficult to distinguish from reference forests (Fig. 3.1H).

Figure 3.2. Canopy-layer features of plantation stands and reference natural forests. For each canopy feature shown, black-shaded lines indicate average values observed within reference mature natural forests (R) while blue, pink, and orange-coloured lines indicate results from thinned softwood (T), under-thinned softwood (U) and mixed hardwood softwood (H) plantation stands, respectively. Panels A-D show features that did not depend on the amount of time passed between initial stand-planting and sampling, with capped vertical bars indicating the 95% confidence intervals (symmetrically) surrounding the least-squared mean value for each stand type. Symbols above bars indicate that the mean value for the corresponding stand type differed statistically from that for R (*=P<0.05; ^= 0.05<P<0.10). Panels E-G show features that did vary significantly with time since afforestation in at least one type of plantation (solid regression lines; dashed lines indicate no significant relationship). For each such feature, points along the x-axis where significant regression lines intersect the reference line correspond to expected time-lags between stand planting and equivalence to reference forests, within the indicated plantation type. Panel H shows typical canopy structure for a mature T plantation.

3.4 Results (Scientific Report) – Fig. 3.2



3.4.2 Spontaneous understorey development

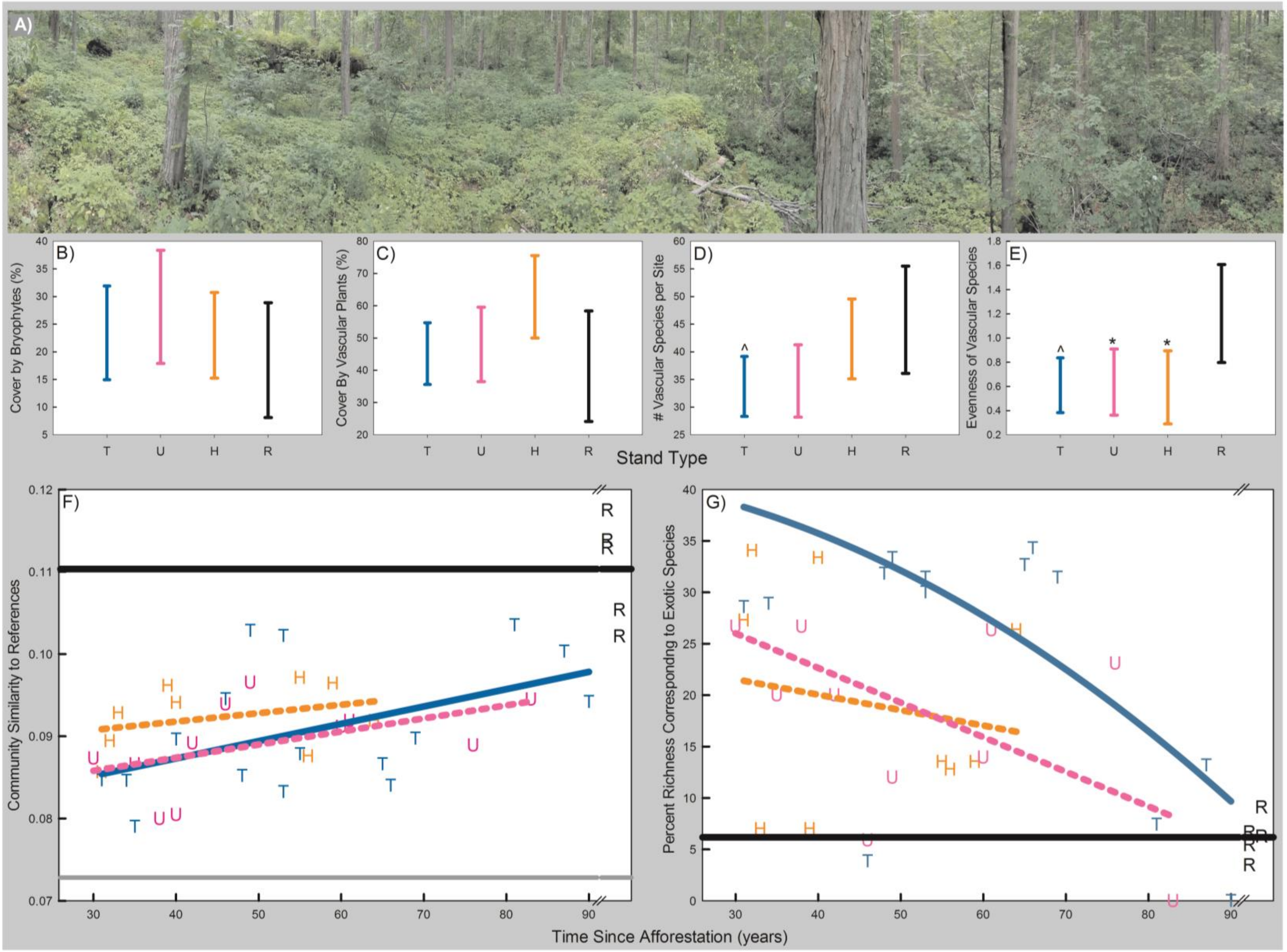
Several aspects of vegetation at the ground layer were highly variable within R stands (i.e. Fig. 3.3A), as well as among stand types, contributing to lack of significant differences between reference and plantation stands for some variables. Although stand type but not time since afforestation explained significant variance in surface cover by both bryophytes (model $F_{(4,37)}=18.34$, $P < 0.0001$, $R^2=0.22$) and other non-tree vegetation (model $F_{(4,37)}=71.30$, $P<0.0001$, $R^2=0.15$), there were no significant differences between R stands and any type of plantation type for either variable ($P>0.10$) (Figs. 3.3A-B). Variance in the average number of vascular plant species per square-metre sampling plot (species density) was also significantly explained by stand type but not time since afforestation (model $F_{(4, 37)} = 94.52$, $P<0.0001$, $R^2=0.19$), and species density in R (7-11 species/m^2) tended to be greater than in T (5-8 species/m^2) or U (4-7 species/m^2) but not necessarily H (7-9 species/m^2) plantations (P-values for the comparisons were 0.0733, 0.0331, and 0.5931, respectively)(Fig. 3.3CD). The mean number of understorey species at the whole stand level (site richness) showed a similar pattern (model $F_{(4,37)}=126.65$, $P<0.0001$, $R^2=0.17$) except that greater richness in R (36-55 species/stand) only tended towards a significant difference from T (28-39 species/stand, P=0.0763) and was not significantly different from U (29-41 species/stand, P=0.1337) or H (35-50 species/stand, P=0.8521) (Fig. 4.5 in Appendix C). The evenness with which these species were represented at each site was also explained by stand type but not time (model $F_{(4,37)}=25.91$, $P<0.0001$, R2=0.17), and evenness was greater in R (0.8-1.6) than T (0.4-0.8, P=0.0320), U (0.4-0.9, P=0.0521) or H (0.3-0.9, P-0.0441) plantations (Fig. 3.3E).

Compositional similarity of plantation understorey communities to reference forests as estimated using the Jaccard similarity coefficient was significantly explained by stand type interacting with time since afforestation (model $F_{(6,30)}=1429.49$, $P<0.0001$, $R^2=0.26$), with similarity increasing over time in T (t=2.56, P=0.0156) but not U (t=1.46, P=0.1553) or H (t=0.64, P=0.5270) stands (Fig. 3.3F). The mean coefficient value calculated for a community with no species in common with any of the five reference forests was 0.073 while that calculated for a community identical to any one of the reference stands was 0.110. Against this background, the modelled value for T plantations climbed from 0.085 in a 31 year-old to 0.098 in a 90 year old stand, reaching a point 68% along the similarity gradient. We projected that 150 years of post-planting stand development would be required for thinned softwood plantations to reach maximal similarity on the scale. The other plantation types did not shift significantly over time from their average similarity values (0.085-0.093 for U, 0.088-0.097 for H), which were significantly less than that for R (0.104-0.116; P<0.0001 for both comparisons). An alternate, non-parametric and multivariate approach to assessing community similarity through Redundancy Analysis (RDA) found a broadly similar pattern of gradually increasing compositional similarity over time (see Fig. 4.6 in Appendix C).

The proportion of site richness attributable to non-native plant species (percent exotic richness) was best explained by stand type interacting with the square of time since afforestation (model $F_{(6,30)}=25.83$, $P<0.0001$, $R^2=0.34$). Exotic richness was overall variable but significantly lower in R stands (4-9%) than plantations (19-28% across all types; P=0.0076). In H and U plantations, exotic richness was unrelated to time since afforestation (P>0.13). However, in T plantations exotic richness exhibited a steady curvilinear decrease over time (t=-2.93, P=0.0064), dropping from 38% to 10% over the 31-90 year timespan (Fig. 3.3G).

Figure 3.3. Understorey-layer features of plantation stands and reference natural forests. For each understorey feature shown, black-shaded lines indicate average values observed within reference mature natural forests (R) while blue, pink, and orange-coloured lines indicate results from thinned softwood (T), under-thinned softwood (U) and mixed hardwood softwood (H) plantation stands, respectively. Panel A shows typical understorey structure in an R forest (i.e. the target of mitigative afforestation). Panels B-E show features that did not depend on the amount of time passed between initial stand-planting and sampling, with capped vertical bars indicating the 95% confidence intervals (symmetrically) surrounding the least-squared mean value for each stand type. Symbols above bars indicate that the mean value for the corresponding stand type differed statistically from that for R (*=$P<0.05$; ^= $0.05<P<0.10$). Panels F-G show features that did vary significantly with time since afforestation in at least one type of plantation (solid regression lines; dashed lines indicate no significant relationship). For each such feature, points along the x-axis where significant regression lines intersect the reference line (or would intersect, if extrapolated) correspond to expected time-lags between stand planting and equivalence to reference forests, within the indicated plantation type.

3.4 Results (Scientific Report) – Fig 3.3



3.4.3 Spontaneous soil and microhabitat development

Of the 17 forest variables measured related to soil properties and surface cover by various non-living substrates, variance in 10 was significantly explained by stand type but not time since afforestation while the remaining 7 could be explained by time since afforestation alone or through interaction with stand type. Of the 10 variables unresponsive to time, 7 showed no significant differences when each plantation type was compared to R: surface cover by bare soil and fine woody debris, soil phosphorus, cation exchange capacity, and relative distributions of sand, silt and clay-sized particles in the mineral fraction of the soil (see Fig. 4.7A-F in Appendix C). In contrast, soil bulk density (model $F_{(4,37)}$=707.95, P<0.0001, R^2=0.20) was significantly lower in R (0.93-1.20 g/mL) than in T (1.20-1.35 g/mL, P=0.0240) or H (1.20-1.40, P=0.0168), and tended towards lower values than in U (1.14-1.32, P=0.1050) (Fig. 3.4A). Both soil organic matter (model $F_{(4,37)}$=59.46, P<0.0001, R^2=0.15,Fig. 3.4B) and moisture content (model $F_{(4,37)}$=94.50, P<0.0001, R2=0.19, Fig 3.4C) were significantly greater in R (3.1-5.4% OM, 17-27% H_2O) than in T (2.0-3.3% OM, P=0.0381; 11-17% H_2O, P=0.0136), and tended to be greater in R than in U (2.2-3.8% OM, P=0.1691; 13-20% H_2O, P=0.1360) or H (2.2-3.9%OM, P=0.1796; 13-20% H_2O, P=0.1635).

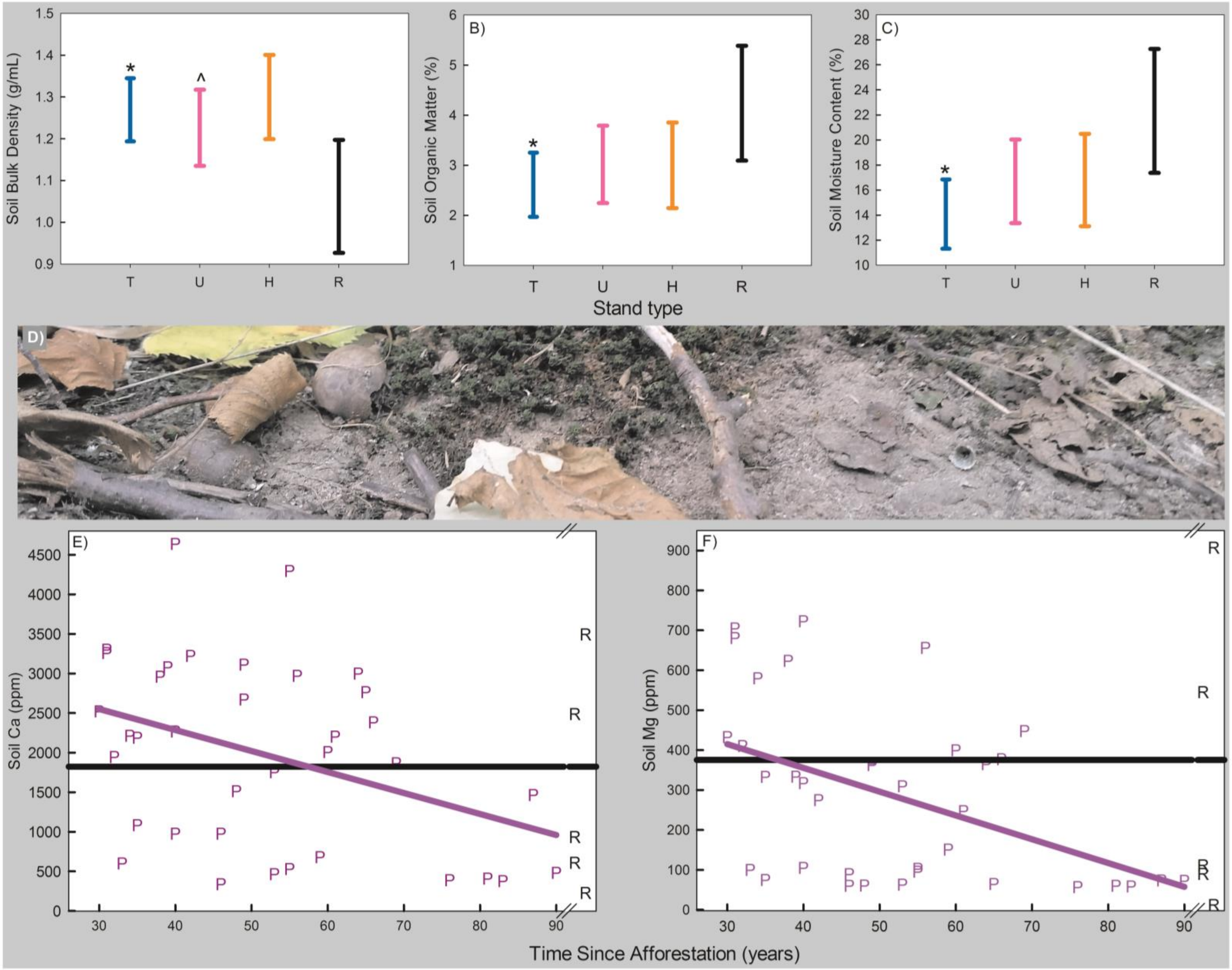
Of the seven variables that did change significantly over time since afforestation, soil calcium and magnesium concentrations did so independent of stand type on a trajectory of increasing similarity to R stands early on, followed by equivalence and then eventually decreasing similarity to target values (i.e. a difference in the opposite direction to that existing at the onset of afforestation). Calcium decreased substantially in plantations over time (model $F_{(1,34)}$=5.84, P=0.0212, R^2=0.15), with predicted values dropping from 2500 (+/- standard error 300) ppm in 30 year-old stands to 900 (+/- 500) ppm in 90 year-old stands (Fig. 3.4E). In contrast, in R stands the concentration was 1800 (+/- 500) ppm, a value achieved in plantations 57 years after tree planting. Similarly, soil magnesium in plantations decreased over time (model $F_{(1,34)}$=9.41, P=0.0042, R^2=0.22), from 400 (+/- 50) ppm in 30 year-old stands to 60 (+/- 80) ppm in 90 year-old stands , while the average value in R stands (375 +/- 100 ppm) was predicted to be reached 37 years after afforestation (Fig. 3.4F).

Soil potassium decreased over time since afforestation (model $F_{(6,30)}$=61.22, P<0.0001, R^2=0.62) in H (t=-3.47, P=0.0016) but not T (t=-1.32, P=0.1978) or U (t=-0.76, P=0.4507) plantations, reaching equivalence with R (70+/-10 ppm) after 61 years (Fig. 4.8B in Appendix C). The average concentration in T (50+/-6 ppm) and U (55+/-7 ppm) plantations were not significantly different from that in R (P=0.2622 and P=0.4768, for the respective comparisons). Surface cover by both needle (model $F_{(6,30)}$=46.76 , P<0.0001 , R^2=0.49) broadleaf (model $F_{(6,30)}$=18.20, P<0.0001, R^2=0.35) litter changed significantly over time in T (t=-3.94, P=0.0004 for needles; t=3.90, P=0.0005 for leaf litter) but not U or H stands (P>0.35 for all comparisons) (Fig. 4.8C-D in Appendix C). Cover by needle litter decreased from 77 (+/- 8)% to 21 (+/-9)% as T plantations aged from 31 to 90 years post-planting; over this same period, cover by leaf litter increased from 2 (+/-7)% to 56 (+/-9)%. R forests exhibited mean needle cover of 42 (+/-10)% and broadleaf cover of 28 (+/-10)%, levels reached by T plantations after 57 and 60 years, respectively. A similar pattern of overshooting target values was discovered for soil pH (model $F_{(6,30)}$=631.87, P<0.0001, R^2=0.32) and aluminum concentration (model $F_{(6,30)}$=61.44, P<0.0001,

$R^2=0.31$), but for U stands only (t=-2.99, P=0.0056 for pH; t=3.43, P=0.0018 for Al) and not T or H plantations (P>0.15 for all tests). The mean pH of R soils was 5.9 (+/-0.3) while that of U plantations dropped from 7.3 (+/-0.3) to 5.5 (+/-0.4) as they aged from 30-83 years (target pH was reached after 71 years). The mean aluminum concentration in R soils was 740 (+/-150) ppm while that in U plantations rose from 530 (+/-150) ppm to 1600 (+/-200) ppm as they aged from 31-83 years, reaching equivalence with the target value after 40 years. For each of the five variables where one plantation type changed over time while the others did not, the time-insensitive stand types exhibited no significant differences from R stands (P>0.16 for all comparisons).

Figure 3.4. Soil features of plantation stands and reference natural forests. For each soil feature shown, black-shaded lines indicate average values observed within reference mature natural forests (R) while blue, pink, and orange-coloured lines indicate results from thinned softwood (T), under-thinned softwood (U) and mixed hardwood softwood (H) plantation stands, respectively. Violet lines in the absence of other colours represent results for all plantation stands taken as a single group (P). Panel A-C show features that did not depend on the amount of time passed between initial stand-planting and sampling, with capped vertical bars indicating the 95% confidence intervals (symmetrically) surrounding the least-squared mean value for each stand type. Symbols above bars indicate that the mean value for the corresponding stand type differed statistically from that for R (*=P<0.05). Panel D shows the typical appearance of surface-layer soil and fine-scale habitat features in an R forest. Panels E-F show features that did vary significantly with time since afforestation, independent of plantation type.

3.4 Results (Scientific Report) – Fig. 3.4



3.4.4 Spontaneous development of coarse woody debris

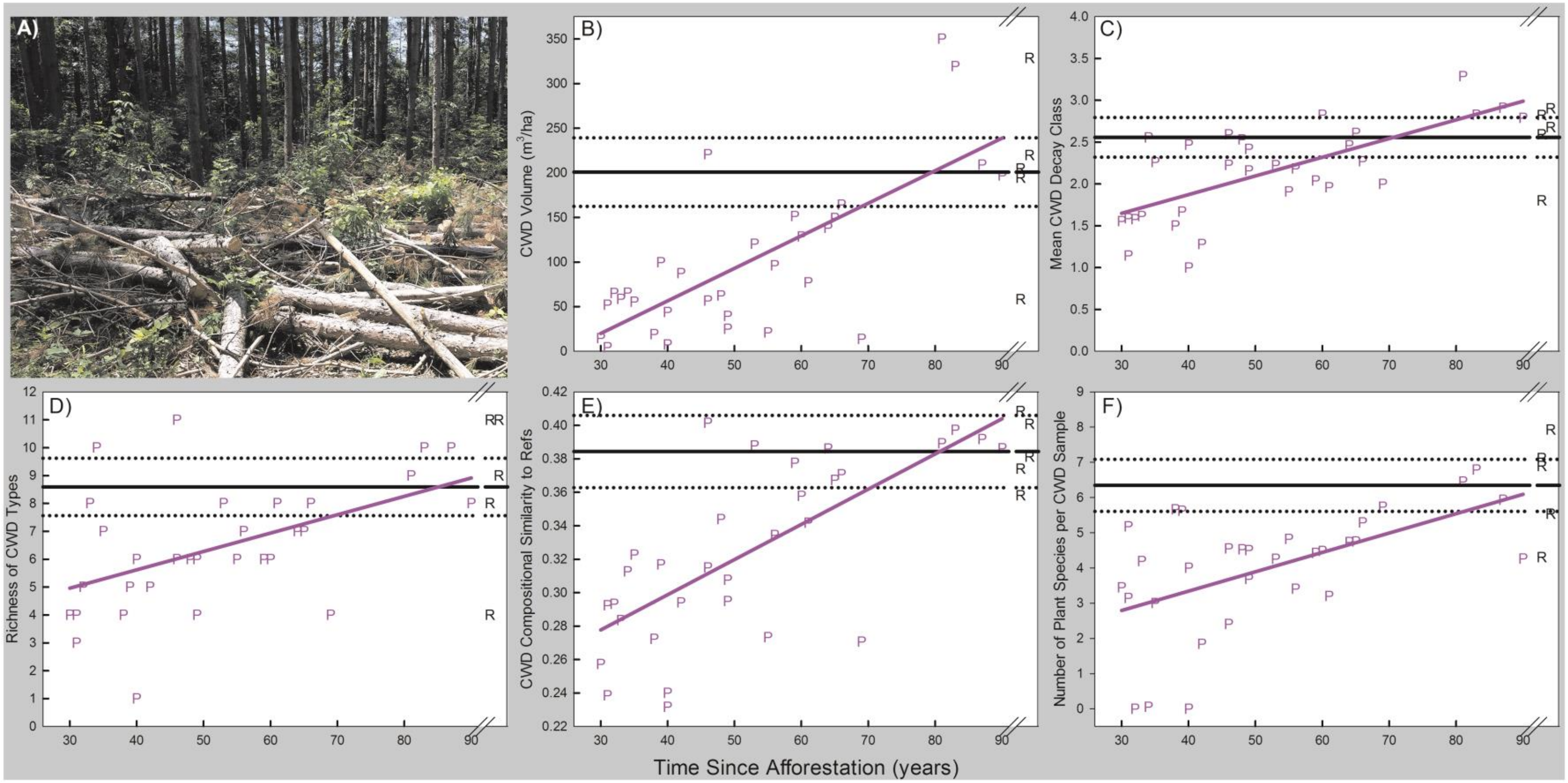
All of the major aspects of coarse woody debris (CWD; Fig. 3.5A) investigated showed strong patterns of change over time but no differences among the plantation stand types. The generalized linear model of CWD volume as a function of time since afforestation revealed a significant effect of time (Wald X^2 [1, N=31]= 34.82, P<0.0001) and produced predicted values that were strongly correlated with the observed values (Pearson's r=0.73). As the square of the correlation between observed and predicted values is a robust indicator of goodness-of-fit analogous to R^2 for general linear models (*31*) , it is reported here as R^{2*}; i.e. R^{2*}=0.53 for the model of CWD volume). The mean total volume of CWD in R stands was 200 (+/-40) m^3/ha; in plantations, the model-predicted values for CWD volume increased linearly from 20 (+/-20) m^3/ha in a 30 year-old stand to 240 (+/-40) m^3/ha in a 90 year-old stand, intersecting the target volume 80 years after tree-planting (Fig. 3.5B). The mean decay class of CWD objects encountered also increased significantly over time since afforestation (X^2 [1, N=31]= 29.13, P<0.0001, R^{2*}=0.48), ascending from 1.7 (+/- 0.1) to 3.0 (+/-0.2) over the age gradient and intersecting the average value for R forests (2.6+/-0.2) after 71 years (Fig. 3.5C).

When CWD objects were treated as different "species" based on their type (log, stump or snag) and decay state (class 1-5), the mean number of "species" present per stand increased with time since afforestation (X^2 [1, N=31]= 10.37, P=0.0013, R^{2*}=0.25), climbing from 5 (+/-0.6) to 9.0 (+/-0.9) over the age gradient and intersecting the average value for R forests (9+/-1) after 85 years (Fig. 3.5D). Analysis of other metrics of diversity (i.e. Shannon diversity of CWD "species") revealed similar patterns (not shown). Compositional similarity to R forests with respect to CWD "species" increased significantly over time (X^2 [1, N=31]= 29.74, P<0.0001, R^{2*}=0.49), increasing from 0.28 (+/- 0.01) to 0.40 (+/-0.02) over the plantation age gradient and reaching equivalence with R (0.38+/-0.02) after 81 years (Fig. 3.5D).

Finally, the mean number of plant species using each CWD object as habitat increased with time since afforestation (X^2 [1, N=31]= 13.61, P=0.0002, R^{2*}=0.31), growing from 2.8 (+/-0.4) to 6.1 (+/-0.6) species/object over the 30-90 year development span (Fig. 3.5E). Equivalence with R (6.4+/-0.7 species/object) was projected to emerge 95 years after tree planting. Similar patterns were discovered when other metrics of diversity of CWD-colonizing vegetation were considered, including the number of different life-form groups represented and the evenness with which these represented by different species (not shown).

Figure 3.5. Coarse woody debris (CWD) features of plantation stands and reference natural forests. For each CWD feature shown, black-shaded lines and parallel dotted lines indicate the least-squared mean and 95% confidence interval for reference mature natural forests (R). Violet lines indicate results from plantation stands (all types taken as a single group, P). Panel A shows CWD at a recently thinned, mid-aged T plantation. Panels B-F show features that varied significantly with time since afforestation. For each such feature, points along the x-axis where significant regression lines intersect the reference line correspond to expected time-lags between stand planting and equivalence to reference forests.

3.4 Results (Scientific Report) – Fig. 3.5



3.4.5 Development of micro-topography features (MTF)

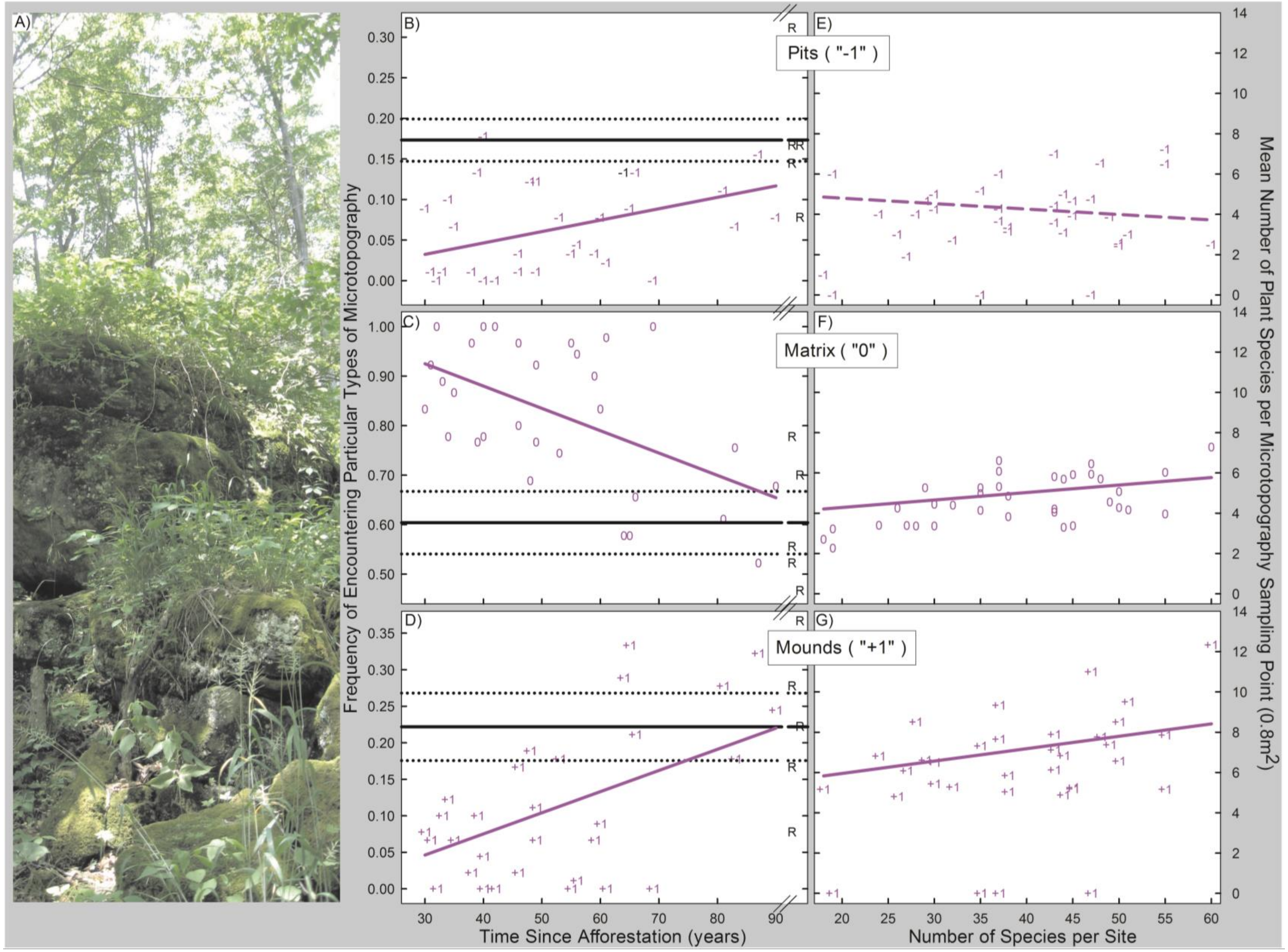
Variance in the frequency of encountering pits, mounds and level "matrix" micro-topography features (MTF) (Fig. 3.6A) was explained best by general linear models that included time since afforestation but not plantation type, plus (in two cases) a covariable related to soil texture. The frequency of encountering pits in R forests was 0.17 (+/-0.03) while in plantations it depended (model $F_{(2,28)}$=5.07, P=0.0132, R^2=0.27) on time since afforestation (F=7.16, P=0.0123) and soil clay content (F=6.21, P=0.0189). Given mean clay content (17.5%), pit frequency was predicted to climb from 0.03 (+/-0.01) (i.e. 3% of points sampled) in a 30 year-old planted stand to 0.12 (+/-0.02) in a 90 year-old planted stand, and reach the frequency for R stands 130 years after tree planting (Fig. 3.6B). Pit frequency increased with soil clay content; given median plantation age of 60 years, pit frequency at the lowest observed level of clay content was predicted to be 5 (+/-1)% while that at the highest observed level of clay content was 14 (+/-3)%. Pit frequency equivalent to R stands was predicted to emerge at this highest level of clay content (38.5%) 85 years after tree planting.

The mean frequency of encountering matrix microtopography in R forests was 0.60 (+/-0.06). In plantations, matrix frequency was best explained by time since afforestation alone (model $F_{(1,29)}$=12.52, P=0.0014, R^2=0.30), decreasing from 0.94 (+/-0.04) in the youngest to 0.65 (+/-0.05) in the oldest stands and projected to reach equivalence with R forests 101 years after planting (Fig. 3.6C). The frequency of encountering mounds in R stands was 0.22 (+/-0.04). In plantations, mound frequency was best explained (model $F_{(2,28)}$=11.63, P=0.0002, R^2=0.45) by time since afforestation (F=11.09, P=0.0024) and soil silt content (F=4.97, P=0.0339). Under mean soil silt content (32%), mound frequency increased from 0.05 (+/-0.02) to 0.22 (+/-0.04) over the 30-90 year age-gradient, reaching equivalence with R stands after 91 years (Fig. 3.6D). Mound frequency increased with decreasing soil silt content; at the median plantation age of 60 years, mound frequency increased from 0.08 (+/-0.03) at the highest observed level of silt (61%) to 0.18(+/-0.02) at the lowest observed level of silt (6%). Given such low silt content, mound frequency consistent with R stands could be achieved 72 years after tree planting.

The capacity for different microtopography features to support different numbers of inhabiting plant species was investigated in the broader context of multiple factors potentially controlling the number of plant species occupying a particular point in space, especially the number of species in the local species pool (estimated here by Site Richness determined from the understorey species composition survey). The Generalized Linear Model that best explained variance in the number of species per 0.8 m^2 microtopography sampling plot revealed significant effects of microtopography, but also significant effects of stand type, site richness, and environmental covariates including tree density, cover by bare soil, and the sandiness of the soil (see Table 4.3 in Appendix C for full statistical details). The effect of microtopography was best expressed through interaction with site richness; as the number of species per site increased, in general so too did the number of species per 0.8 m^2. However, the strength and even the direction of the response differed depending on whether the small patch of forest floor in question was a pit, matrix or mound feature. At mean levels of the other explanatory factors, species density in pits was predicted to be consistently low (4+/-1 species/0.8 m^2) regardless of whether site richness was the minimum (18 species) or maximum (60 species) observed level (.e. X^2=0.32, P=0.5701 for the effect of site richness within pits; Fig.3.4D). In contrast, under the same conditions within matrix sampling points, species density was higher and increased significantly with site richness, from 4.2 (+/-0.2) species//0.8 m^2 at the 18-species site to 5.8 (+/-0.2) species//0.8 m^2 at the 60-species site (X^2=20.04, P<0.0001; Fig. 3.4E). At mound sampling points the relationship was even stronger, with species density increasing from 5.8 (+/-0.6) to 8.4 (+/-0.7) species/0.8 m^2 across the range of variation in site richness (X^2=4.33, P=0.0374; Fig. 3.4E). Thus, when effects of site richness on small-scale species density were adequately accounted for, forest floor mounds thus clearly supported a greater diversity of plant species than did matrix patches, and both of these supported greater diversity than pits. With respect to the other explanatory factors of species density, R and H stands were similarly associated with approximately 1 more species per quadrate than T and U stands (compiled into the Softwood-only group “S” because U did not differ from T during exploratory analysis) (Table 4.3). Species density increased significantly with increasing cover by bare soil and soil sandiness, but decreased with increasing tree density (Table 4.3). Each of these effects was relatively weak however, predicting species density shifts of approximately 1 species/0.8 m^2 across the observed range of each covariate. The R^{2*}value for the model including all significant factors except for those related to microtopography was 0.17; when microtopography and its interaction with site richness was added to the model the value increased to 0.27.

Figure 3.6. Micro-topographic features (MTF) of plantation stands and reference natural forests. Panel A shows a pronounced example of MTF observed at a reference mature natural hardwood forest stand (R). More typically, small depressions 0.3-1.0 m below-grade were observed (pits, represented by the symbol “-1”), as well as hummocks 0.3-1.0 m above-grade (mounds, “+1”), relative to more-or-less level-ground “matrix” patches of forest floor (“0”). Panels B-D show the respective frequencies of encountering pit, matrix and mound MTFs in plantation stands over the 30-90 year age-gradient; solid coloured lines indicate statistically significant relationships with time since afforestation. Black solid lines and bracketing dotted lines indicate the least-squared mean and 95% confidence interval for the feature’s frequency in R stands. Panels E-G illustrate the average number of plant species respectively using pit, matrix and mound MTFs as habitat, in each case shown as a function of the number of plant species in the site species pool (i.e. site richness as observed during the understorey survey of all stands). The dashed regression line in E) indicates the number of species colonizing pits did not increase significantly as site richness increased, while the solid lines in F) and G) indicate that species density in matrix and mound sampling plots increased significantly with site richness, and at a greater rate in mound than in matrix habitat. Mounds thus supported greater plant diversity than did matrix patches, which in turn supported greater diversity than pits, even when accounting for the covarying influence of site richness on plot-level species density. Note that these results reflect the patterns observed in plantation and reference stands taken as a single group.

3.4 Results (Scientific Report) – Fig. 3.6



3.4.6 Response of Phytometers – *Allium tricoccum*

Individual *Allium tricoccum* (wild leek) plants (Fig. 3.7A) were observed in the herb-relocation plots in the first and third year following transplantation, but not in the second year, likely because sampling began too late in the spring to adequately capture this shade-avoiding ephemeral spring species. For this reason, analyses considered Year 1 and Year 3 data only. An initial general linearized model exploring potential influences of study year and stand identity as R, H, M (i.e. monoculture softwood plantations) or X (mixed softwood plantations) revealed no significant differences between M and X stands or between Year 1 and Year 3 patterns (Table 4.4 in Appendix C), so M and X were consolidated into stand type "S" (softwood-only) and study year was dropped as an explanatory factor in subsequent analyses (though accounting for potential auto-correlation due to repeatedly measuring the same plots over time was retained). As the actual response variable analyzed by this model was the log of the odds of observing living wild leek individual in a herb-relocation plot (expressed as a ratio), back-transformation of means and 95% confidence intervals was required to translate results to probability or percent likelihood of occurrence, while the scale of the log-transformation was retained for the clearest possible illustrations of the patterns of results. Across all stand types there were 2460 trials and 679 *Allium* occurrence events, corresponding to a 28% likelihood of *Allium* occurrence. From this initial analysis of all stands, we determined that the log-odds of observing wild leek in relocation plots within R forests was -0.3 (+/-0.1), corresponding to a 44% (95% CI: 37-50%) likelihood of occurrence.

When just the data from plantation stands were analysed, the log-odds of wild leek occurrence was significantly related to stand identity as an S or H plantations, the amount of time passed since afforestation, and the interaction between these factors (Table 4.5 in Appendix C). This dataset included 2160 trials over the two sampling years (Year 1 and Year 3) and yielded 548 *Allium* occurrence events (25%). Back-transformation of the log-odds results showed that in H plantations there was a 19% (16-23%) likelihood *Allium* occurrence in relocation plots following root transplantation, regardless of time since stand planting (X^2=1.91, P=0.1666). In S plantations, however, likelihood of *Allium* occurrence increased significantly with time since afforestation (X^2=82.23, P<0.0001), from a predicted 15% (12-18%) in a 30 year-old plantation to 50% (46-57%) in a 90 year-old plantation (Fig. 3.7B). *Allium* occurrence equivalent to that in R forests was predicted to emerge in S plantations 82 years after initial tree-planting. The R^{2*} for the analytical model was 0.05, indicating considerable variance in *Allium* occurrence could not be explained by stand composition type or time since afforestation.

A separate analysis focussed on *Allium* occurrence in the last sampling year within thinned (T) versus under-thinned (U) softwood-only stands (but still informed by data from both sampling years and H plantations) indicated that *Allium* survival increased with time since afforestation under both thinning treatments, but followed a different pattern for each (Table 4.6). In U stands, the predicted likelihood of *Allium* occurrence increased significantly from 4% (2-8%) in a 30 year-old plantation to 44% (32-58%) in an 83 year-old one (X^2=29.70, P<0.0001), the latter a value equivalent to that observed in R stands. In T plantations, *Allium* occurrence increased significantly from 23% (17-29%) to 53% (43-62%) across the 31-90 year age gradient

(X^2=20.14, P<0.0001), intersecting with the target R value after 75 years (Fig. 3.7C). In contrast, *Allium* occurrence in H plantations in the third sampling year significantly decreased with time since afforestation (X^2=9.23, P=0.0023), from 26% (18-35%) to 7% (3-14%) over the 31-64 year age-gradient available for H. The R^{2*} for this model was 0.07, indicating more variation in transplant survival was explained by consideration of softwood thinning rather than planted composition classes. The log-odds of *Allium* occurrence were not significantly different from a value of zero in R stands or the oldest T or U plantations (P > 0.10), indicating "even odds" or a 50:50 chance of phytometer occurrence in such plantations three years after transplantation (i.e. $\log_e$ (50/50)=$\log_e$(1)=0). Although both and T and U plantations reached phytometer equivalence with R in a similar length of time, U stands exhibited significantly lower *Allium* occurrence in young plantations (intercept contrast: X^2=20.41, P<0.0001) and a sharper rate of increasing occurrence as plantations aged (slope contrast: X^2=8.60, P=0.0023) compared to T stands.

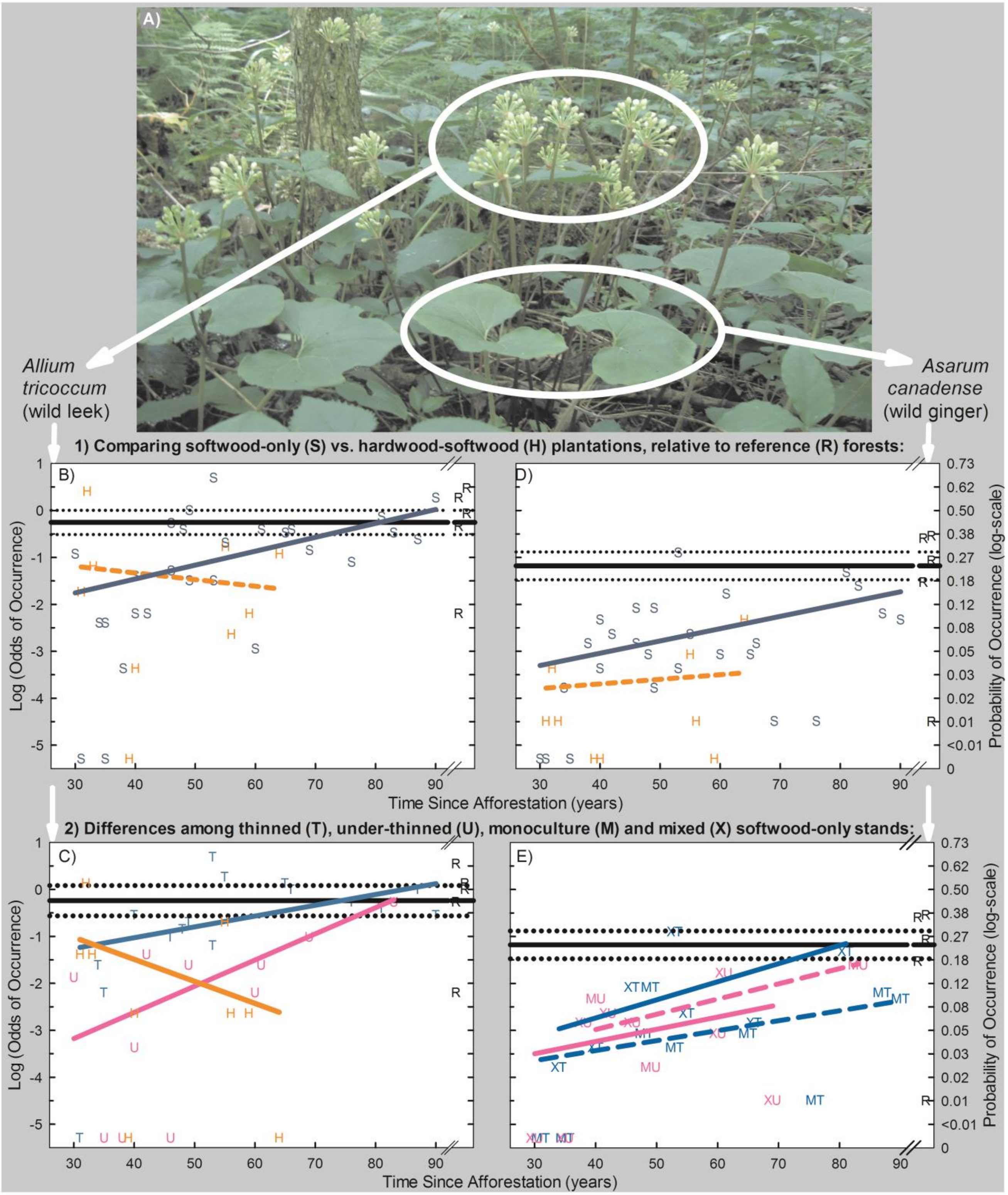
3.4.7 Response of Phytometers – *Asarum canadense*

Unlike *Allium*, the shade-tolerant phytometer *Asarum canadense* (wild ginger; Fig. 3.7A)) was observed in herb relocation plots in all three sampling years but at relatively low frequency, with only 292 *Asarum* occurrence events across the 3690 trials over the three years. *Asarum* occurrences decreased with each sampling year, but as this was true for references forests (R) as well as plantation stands (P), subsequent analysis of phytometer responses considered average responses across sampling years for all stand types. The initial analysis of *Asarum* occurrence as a function of sampling year and stand identity as R or P revealed a significant interaction between these factors (Table 4.7), with mean (95%CI) likelihood of occurrence in R dropping from 36% (29-44%) in the first year to 20% (14-27%) in the second and 14% (9-21%) in the third sampling year. In P stands, *Asarum* likelihood dropped from 14% (12-16%) to 2% (1-3%) to 1% (0.8-2.0 %) across sampling years. R^{2*} for this model was 0.10. The time-averaged likelihood of occurrence in R stands was 24% (19-30%).

Analysis of *Asarum* in P stands only showed that a small but significant portion of the variation in phytometer occurrence was explained by time since afforestation interacting with planted composition (Table 4.8; model R^{2*}= 0.02). In stands planted with a mixture of hardwood and softwood species (H), *Asarum* occurrence was very low and unrelated to time since afforestation (X^2=1.56, P=0.2122), with an average likelihood of 2% (2-4%). In contrast, *Asarum* occurrence in softwood-only plantations (S) increased significantly with time since afforestation (X^2=42.32, P < 0.0001) from a predicted 4% (3-5%) in a 31 year-old stand to 15% (12-19%) in a 90 year-old stand (Fig. 3.7D). Slightly more variation was accounted for when consideration was given to both the level of thinning experienced (T vs. U) and whether softwoods planted at each stand were a monoculture (M) or a mixture of a few species (X) (Table 4.9; model R2*=0.03). This model revealed that although all groups exhibited an increase in *Asarum* occurrence with planation age, levels of occurrence and rates of change over time differed and thinning had a positive effect in X stands but a negative one in M stands. In other words, the projected time lag until equivalence with R forests was shorter for under-thinned monoculture (MU) stands than thinned monoculture stands (MT), but longer for under-thinned mixed (XU) stands than thinned mixed stands (XT) (Fig. 3.7E). MT stands exhibited the slowest rate of convergence, increasing from 3% (2-4%) in a 31 year-old stand to 9% (6-13%) in a 90 year-old stand (X^2=4.31, P < 0.0001), with equivalence to R forests projected to develop over 146 years. MU stands developed faster, with *Asarum* occurrence increasing significantly over the 40-83 year-old age-gradient available for this group (X^2=6.00, P < 0.0001), from 5% (3-7%) to 17% (12-24%) and with R convergence expected after 96 years. In contrast, XU plantations were relatively slow to develop equivalent *Asarum* occurrence to R, increasing significantly (X^2=3.93, P < 0.0001) from 3% (2-4%) to 8% (5-12%) over the 30-69 year age-gradient and convergence not expected until 119 years have passed. XT plantations developed 33% faster than this, increasing significantly (X^2=7.25, P < 0.0001) from 5% (4-7%) to 24% (18-32%) between 34 and 81 years post-planting, such that R equivalence was achieved after only 80 years. Combining regular thinning with mixed softwood planting (and excluding hardwoods) thus produced habitat conditions ideal for the *Asarum* phytometer in a similar amount of time needed to produce optimal *Allium* habitat as well as multiple target canopy, understorey, and microhabitat features, suggesting current best afforestation practices can achieve robust ecological equivalence with mature natural hardwood forests within about 80 years.

Figure 3.7. Increasing similarity of planation stands to mature natural hardwood forests over time, as indicated by relocated herb species (phytometers). Panel A) shows the two phytometer species used – *Allium tricoccum* (wild leek) and *Asarum canadense* (wild ginger) – in their natural habitat, the understorey of a mature natural hardwood forest (reference, R). Panels B-C) shows the likelihood (including the transformed log-odds scale and the back-transformed percent probability scale) of observing *Allium* within herb-relocation plots in plantation stands spanning a 30-90 year gradient in time since afforestation. Panels D-E) show results for *Asarum*. For both phytometers, the first level of analysis involved comparison of stands initially planted with softwood trees only (S, blue lines) versus a mixture of hardwood and softwood trees (H, orange lines), with respect to rates of change in phytometer occurrence over the plantation age-gradient (panels B and D). Solid regression lines indicate a statistically significant relationship with time since afforestation; dashed lines indicate no significant relationship (i.e. in H); black lines and bracketing dotted lines indicate least-squared means and associated 95% confidence intervals for phytometer occurrence within relocation plots in R stands. Phytometer occurrence was based on observations in each sampling year. The second level of analysis for *Allium* (panel C) was to consider survival in the final sampling year and investigate potential differences between thinned (T, blue line) and under-thinned (U, pink line) softwood plantations, as well as H stands (orange line). For *Asarum,* the deeper analysis (panel D) investigated potential differences among four types of softwood-only plantations, looking at all sampling years: thinned monoculture plantings (MT, dashed blue line), under-thinned monoculture plantings (MU, dashed pink line), thinned mixed-softwood plantings (XT, solid blue line) and under-thinned mixed-softwood plantings (XU, solid pink line).

3.4 Results (Scientific Report) – Fig. 3.7]



3.4.8 Response of Phytometers to Soil Addition Treatments

Addition of sterilized or non-sterilized soil from the home reference forests alongside phytometer root material had relatively minor and inconsistent impacts on phytometer occurrence in the relocation plots; for example, analysis of *Asarum* occurrence as a function of stand type (H=hardwood-softwood plantation; M=monoculture softwood plantation; X=mixed-softwood plantation; R=mature natural reference forest), soil addition treatment (i.e. C=control, or no soil added; N = not-sterilized R soil added; S = sterilized R soil added), and the interaction of these yielded no significant effects of either soil addition or the interaction term (Table 4.10; *Allium* showed a similar pattern but the data is not shown). We addressed the question of whether differences between plantation and reference forest soils influenced the rate at which suitable phytometer habitat developed in different plantation types by investigating the rates of this development within various combinations of plantation types and soil addition treatments, focussing the analysis on predictions for the middle sampling year for simplicity (see Table 4.11 for *Asarum*; *Allium* showed no interesting differences between soil addition treatments and the data is not shown).

Significant interactions between the effect of time since afforestation, soil addition, planted composition and thinning were resolved to reveal significant effects of soil addition treatment on *Asarum* response to plantation age in softwood-only plantations, but with different patterns in thinned mixed-softwood plantations and under-thinned softwood plantations (monoculture and mixtures combined). In the thinned mixed-softwood plantations, likelihood of *Asarum* occurrence was 5% (3-8%) in the youngest stand (34 years old) regardless of whether or not soil was added; after 81 years of stand development, this likelihood increased to 21% (9-41%) in the treatment which had received sterilized home soil, but only increased to 14% (5-37%) in the treatment receiving non-sterilized soil and that receiving no additional soil. This difference in the strength of the effect of time since afforestation was statistically significant (X2=10.24, P=0.0014) but overall not great in magnitude (Fig. 3.8A).

In the under-thinned softwood plantations, *Asarum* only exhibited a significant increase in occurrence with time since afforestation in the relocation plots where non-sterilized soil had been added (X^2=5.67, P=0.0172); this effect was not significant where no soil had been added (X^2=0.92, P=0.3365) or where sterilized soil had been added (X^2=0.01, P=0.9281) (Fig. 3.8B). Thus both thinned and under-thinned plantations exhibited faster convergence with reference conditions (with respect to *Asarum* habitat) where soil differences from R were experimentally overcome, however, in the thinned plantations the pertinent differences related to non-living components of the soil while in the under-thinned plantations the key differences were living soil features.

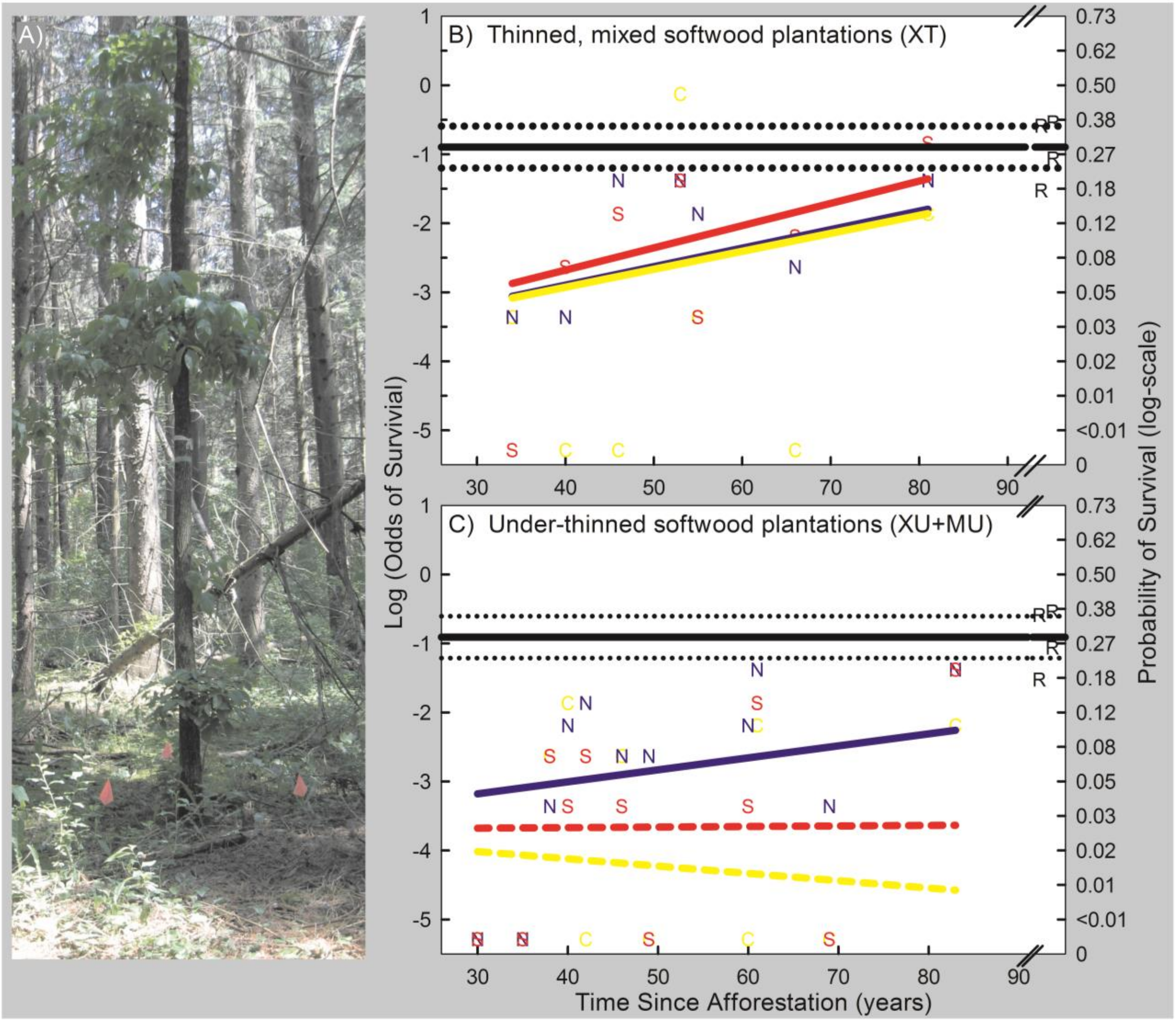


Figure 3.8. Effects of the soil-addition treatment on *Asarum* response to time since afforestation. Panel A) shows a typical set of herb-relocation plots nested within the three levels of the soil addition treatment; each flag marks a location of phytometer out-planting, but one corresponds to addition of the phytometers only (control, C), one corresponds to addition of phytometers plus not-sterilized soil (N) collected from the nearest natural hardwood reference forest (R), and the third corresponds to addition of phytometers plus sterilized soil (S) from the R stand. Panel B) shows that in thinned, mixed-softwood plantations, *Asarum* occurrence increased significantly with time since afforestation in C (yellow line), S (red line) and N (indigo line) soil-addition plots, but in S plots the rate of increase was significantly greater and the expected time-lag until equivalence with R stands (black line) was shorter. Panel C shows a similar set of relationships for under-thinned softwood plantations (both monocultures and mixed-softwoods combined), except that in this case only phytometers in N plots exhibited a significant increase in *Asarum* occurrence over the plantation age gradient (and thus was the only level of the soil-addition treatment expected to eventually yield plantation forests which are indistinguishable from R stands, from the perspective of relocated *Asarum* individuals).

3.5 Discussion

3.5.1 General trends

Patterns of similarity between mature natural forests and plantations show that conventional afforestation can reverse some biodiversity impacts of deforestation, but it may take two centuries before understorey communities are replaced. This is consistent with similar studies in a range of biomes (*13, 14, 16*). Industries responsible for deforestation would thus be prudent to devote resources to mitigative afforestation capable of addressing habitat losses an equivalent timescale. More optimistically though, our results show that aspects of canopy, understorey and microhabitat structure in plantation forests can be expected to converge with target forests as soon as 50-150 years after tree-planting, depending on the feature considered, traits of the trees planted, the regularity of stand-thinning, and constraints on soil development. Similar degrees and timescales of convergence have been observed in other investigations of old plantations, albeit rarely (*21, 32*). However, our findings are complemented by novel phytometer responses that indicate microhabitats supportive of target understorey vegetation can be expected to emerge in approximately half the time needed for community assembly overall. Very mature plantations (e.g. 80-100 years old) may thus be primed for accelerated succession towards targets in response to managed relocation of understorey flora (*33*), reflecting a powerful intervention leverage point.

3.5.2 Insight from phytometers

Superior performance of spring-ephemeral *Allium* relative to shade-tolerant *Asarum* phytometers suggests light-related target conditions take the longest to emerge. However, poor development of target biota in plantation soils delayed emergence of suitable *Allium* habitat while abiotic soil dissimilarities deferred emergence of *Asarum's* preferred microenvironment. Developing methods of closing these gaps, combined with assisted migration to primed plantations, may shorten mitigation time-lags and reduce biodiversity impacts of deforestation as well as costs of offset creation. Our findings agree with previous studies that aging plantation soils converge with target conditions for some properties (*32*) but not all, i.e. they become increasingly saturated with aluminum and depleted of potassium, magnesium and calcium (*34*). As these may inhibit plant growth, manipulations capable of addressing micronutrient deficiencies may be needed. Strategies based on altering soil pH – which is linked to bioavailability of these elements – may not be an option, however, as pH at plantation soils tends to converge with target forests naturally.

3.5.3 Best practices moving forward

Planting hardwoods alongside conifers may accelerate convergence of canopy-layer biodiversity features, but conifer monocultures bring the advantage of highly predictable structural change including gradual replacement of exotic understorey vegetation by natives. Regular stand-thinning may promote expression of these changes through increased heterogeneity and reduced competition, but mechanistic investigation is required. In contrast, planting mixtures of conifer species resulted in the most rapid convergence of plantations with target forests, based on phytometer survival, but only under soil modifications and stand thinning. Accounting for this multitude of potential trade-offs between different management practices and goals will be a major challenge to devising optimal afforestation strategies for mitigating biodiversity impacts of resource extraction. Maintaining a diversity of afforestation strategies may have the greatest chance of preserving the broadest diversity of forest structures and functions. The true importance of our study is the revelation of the unique power of crossed chronosequence and phytometer approaches for investigating over the short-term much longer-term dynamics of complex suites of biologically-relevant ecosystem features.

4 Part 4: Supplementary Information (Appendices)Appendix A: Detailed Background of the Afforested Environments Study

4.1.1 Introduction and Background to the Afforested Environments Study

Mineral aggregate production provides essential resources to multiple industries, in Ontario and worldwide. However, the long-term sustainability of aggregate production depends upon operator capacity to meet environmental standards set by government regulators and public expectations. These include avoiding, minimizing, or reversing negative environmental impacts wherever possible, and mitigating unavoidable impacts by purchasing sites of equivalent size and ecological quality to impacted areas and committing them to long-term conservation (*35*). Unfortunately, achieving this goal of no net habitat loss can be difficult when lands appropriate for mitigation are rare or unavailable for purchase. This is the case for many of southern Ontario's aggregate reserves, which underlie mature mixed deciduous-coniferous forests (*36*). Many of these forests may be considered "ancient" in that they were either never cleared for agriculture ("primary forests"), or were cleared but allowed to return to forest ("secondary forests") at least two centuries ago (*17, 37*). Ancient forests have high natural heritage value due to the specialized habitat they provide to diverse native understory plant and animal communities (*38*), and high natural capital value due to the important ecosystem goods and services produced (e.g. carbon sequestration, climate moderation, nutrient cycling, production of food and raw materials (*39*)).

One possible solution to the problem of low-availability of natural ecosystems for mitigation use is to establish new forests ("afforestation") on lands that presently have low ecological value, such as former agricultural fields (*32, 40, 41*). Given Ontario's extensive pre-settlement forest cover, it is likely that farmlands planted with forest tree species will eventually become "fully functioning" forests, with respect to ecosystem processes and provisioning of habitat for biodiversity (*42*). Nevertheless, the amount of time required, and the ecological processes that set this rate, are presently unknown, as are specific impacts of different afforestation methodologies such as conventional low-diversity plantations versus applied high-diversity ecosystem restoration, or planting trees with widespread, even spatial distribution versus planting trees in clumps or "pods" (*40, 43*). As modern extraction operations can have lifespans extending over many decades, it is conceivable that planting forest tree species in former agricultural fields immediately prior to initiating aggregate extraction nearby will result in fully functional forests before extraction is complete. Here we propose a research project that will determine how long it takes to create a functioning forest on agricultural land, and gain insight into what ecological and management factors set this pace. This knowledge will help maximize the efficiency with which managers develop new forests on former farmlands to help mitigate unavoidable impacts of aggregate production.

4.1.1.1 ***Compensatory mitigation of ecological impacts***

It is well established that human activities such as natural resource extraction and urban development can have strong negative impacts on critical ecosystem services (*44*). Furthermore, it is increasingly clear that the diversity of organisms comprising ecological communities can fundamentally affect ecosystem goods and services produced (*45*), and that restoration of biodiversity can enhance ecosystem functioning (*46*). Growing awareness of negative ecological impacts from industries has prompted legislators, policy makers, and corporations to pay increased attention to the idea that unavoidable harmful impacts at one location should be compensated for at other locations by setting-aside target ecosystems for long-term preservation (*2, 47, 48*). Such compensation may involve setting-aside natural areas that are ecologically equivalent to impacted sites, or it may involve rehabilitating marginal-value lands in order to create new ecosystems capable of fulfilling this role. While this idea of "compensatory mitigation" must be approached with caution given the difficulty of characterizing and adequately replicating complex ecological systems (*49*), the idea is extremely attractive to industries and governments because it suggests that efficient systems may be created for banking positive effects of ecosystem offsets ("credits") and negative impacts of development (*50*). Such "conservation banking" has great potential to help reconcile human needs with maintenance of globally important ecological structures and functions.

Creation of offsets for the preservation of biodiversity, habitat types, or ecosystem services may help industry managers efficiently achieve biodiversity goals, comply with existing regulations, save money, and generate profit (*47*). The effectiveness of initiatives aimed at protecting threatened species depends on the total area of lands set aside for protection, but also on the ecological quality and spatial contiguity of these lands. Under a system of conservation banking, negative impacts to several small, moderate-quality sites (where species populations may be threatened or unstable anyway) may be mitigated by setting aside one or a few large, contiguous, high-quality sites. Though the total area may be equivalent, the conservation potential of the few large sites may far greater than that of the many small sites, and thus biodiversity offsets can work strongly in the favour of conservation interests. Furthermore, as lands available for set-asides are often further from developed areas than close-to-market extraction/development sites, set-asides of high ecological quality may have a lower market cost per unit area than impact sites. Funds accruing from the sale and economic development of mined lands may thus be sufficient to purchase and set-aside much larger, and ecologically more valuable, lands for conservation (*2, 51*). Using ecosystem offsets in the mitigation of ecological impacts can thus clearly be advantageous from the perspective of conservation ecologists.

Internationally, ecological mitigation through offset creation has been formalized in several pieces of legislation and policy, and further development of regulatory frameworks for utilizing such offsets within Canada is expected (*52*). In the USA, the Clean Water Act (973) and the Endangered Species Act (1973) both provide for cases where banking of mitigation credits is acceptable or required, with high-quality wetland area serving as currency in the former case, and specific habitats and populations of red-listed species serving as currency in the latter case (*47*). Similarly, in Brazil, the Forest Code of 1965 (Law 4771) requires that land owners engaged in forestry reserve particular quantities of land for conservation, but this land may either be on the forested site or on other properties purchased for the express purpose of conservation. In this same country, a National System of Conservation Units exists, under which industries generating negative environmental impacts much purchase "conservation credits" from a federal agency responsible for developing reserve networks targeting particular species, habitats, and ecosystems of concern. In Switzerland, the Federal Law for the Protection of Nature and Landscape insists that unavoidable ecological damage to species-specific habitats must be offset by "replacement" or "reconstitution" of the impacted habitats in new locations.

In Canada, the Fisheries Act (1985) mandates that fish habitat must not be destroyed by development, but in cases where such destruction is unavoidable, new or improved habitats must be created and dedicated to long-term preservation (*47*). While the recently legislated Species-At-Risk Act (SARA) does not currently allow for conservation offsets, “everyone chats quietly about whether eventually this may change” (M. Weber, personal communication 7/20/2010). At the provincial level, the Alberta Land Stewardship Act explicitly enables conservation offsets and other market-based tools for biodiversity protection, and other provinces may soon follow suit.

Although Ontario’s Aggregate Resource Act (1997) does not explicitly permit compensatory mitigation as an alternative to rehabilitating extraction sites, efforts by extraction companies to mitigate ecological damage through offset creation can contribute importantly to meeting public and government expectations of corporate environmental responsibility. Such responsibility is becoming increasingly important in determining whether new site licences are granted to aggregate producers. This benefit of voluntary set-asides or ecological-enhancement efforts may indeed be one of the most compelling reasons for company-driven mitigation of unavoidable environmental impacts. Given the widespread availability of degraded or marginal-value farmlands in southern Ontario (particularly relative to mature forest), creating set-asides based on ecologically-rehabilitated ex-farmlands may significantly improve both the environmental trustworthiness of aggregate producers, and the degree to which maintenance of critical population and habitat sizes is achieved.

4.1.1.2 ***Creating functional forests on abandoned agricultural lands***

Much of the legislation, policy, and corporate practice related to biodiversity offsets allows for creation of set-asides from either previously existing high-quality natural areas, or degraded lands raised to high ecological quality through applied ecosystem rehabilitation efforts. This latter option is often preferably from an economic standpoint, as marginal-value lands such as ex-arable farmlands can be more readily available, and for lesser cost, than high-quality natural areas. Of course, significant economic investment is necessary to raise the ecological value of ex-farmlands to that of natural forests, prairies, meadows or wetlands, but in many cases the pay-off of mitigating large-scale extractive activities is far greater than the cost of purchasing and rehabilitating marginal lands (*51*).

Problems arise, however, when managers cannot satisfactorily demonstrate that new or restored ecosystems are actually comparable to target natural ecosystems with respect to biodiversity supported and ecosystem services provided. In southern Ontario, where predominant land cover has shifted from forest to farmland over the past several hundred years, creating new forests on agricultural lands is a major avenue for mitigating industrial developments. However, although such afforestation has been practiced for centuries, there remains a great deal of uncertainty as to what spans of time, and what suites of ecological conditions, are needed in order for planted forests to resemble ancient forests in an ecologically meaningful way (*43*). Ecosystem properties of particular concern include production of a closed canopy of appropriate hardwood and softwood species; appropriate rates of ecosystem processes such as carbon sequestration, water cycling (including flood prevention), nutrient cycling, soil development, and litter production; and appropriate understory floral and faunal communities, including native herbs, birds, and mammals (*39, 53,* *54*). At larger spatial scales, planted forests should connect existing forests in the landscape (*55*), as well as important forest-associated ecosystems such as savannas (grasslands with moderate tree cover). Such spatial linkages can contribute essentially to the ecological integrity of the wider region, and over time should influence the extent to which greenhouse gasses are sequestered (*56*) and water and nutrients are recycled (*57, 58*). Although theoretical studies have investigated how afforestation may contribute to such ecological services (*41, 59*), there have been too few empirical studies to determine with confidence the length of time required before such services are produced, the degree to which they are produced at all, and the influence of different methods for initiating forest development .

Assessing how well afforested farmlands measure-up to reference forests is challenging, but some hope lies in the fact that the relevant ecosystem properties are rarely independent of one another (*60*). As such, knowledge about one property can allow for inferences about other properties. This is particularly true for understory plant communities, which depend strongly on particular light and soil conditions associated with particular levels of canopy cover and age of forest development (*32, 37*). Use of plant species as indicators of environmental conditions is a powerful tool in ecology, particularly where survival, growth, or reproduction of indicator species is strongly correlated with particular habitat conditions (*13, 61-64*). Understory plant communities in ancient forests fit this description closely, exhibiting physiological adaptations to light and soil conditions that are largely unique to forest interiors. Performance of understory indicator species may estimate forest functioning by determining the success with which indicator species have established under natural conditions. However, a much stronger test would explicitly introduce indicator species to test and control areas and determine relative survival rates in each.

4.1.1.3 *The Ecology of Forest Development*

Forest "succession", or natural development at a location not presently covered by forest, has been long-studied in ecology (*65*)). Historically, ecologists have viewed succession as driven primarily by variation in light requirements (or shade tolerance) among tree species, interacting with changes in available light intensity as the forest canopy gradually closes. Tree species adapted to establish under intense-light conditions (and often dry and low-fertility soils) are "pioneers" capable of colonizing recently created or disturbed environments. In primary succession (forest development from a state where no species propagules and few soil resources are available), initial cover and soil-formation by light-adapted herbaceous species is necessary before pioneer tree establishment is possible. As such, expanses of grasslands or meadows with low to moderate tree cover (savannas) may precede extensive cover by forest.

Four general stages of forest development have been described as stand initiation, stem exclusion, understory reinitiation, and old growth (*42, 43*). *Stand initiation* is the initial colonization and growth of pioneer tree species, while *stem exclusion* is the shading-out of all but the most shade-tolerant ground layer species, which occurs as the pioneer trees compete for light and grow rapidly to close the canopy. However, pioneer tree species are generally themselves not adapted to shade, and as such new seedlings fail to establish under the canopy produced by their parents. Instead, shade-adapted tree species colonize the understory (possibly imported by animals attracted to the food and shelter provided by pioneer trees), and eventually replace the canopy layer created by relatively short-lived pioneer species. This process may repeat itself several times, as the canopy layer becomes increasingly dense and species exhibiting increasingly greater shade tolerance at the establishment stage have an advantage. *Understory reinitiation* happens when small and moderate-sized breaks in the closed canopy form as a result of trees growing taller (less branch overlap), and due to eventual death of the pioneer trees in response to drought, wind, or natural end-of-lifespan. As large overstory trees eventually die naturally, canopy breakages are larger and more irregular in occurrence, tree fall creates pit-and-mound topography on the forest floor, and habitat conditions eventually become suitable for a greater diversity of species, including moderately shade-tolerant trees that may form patches of sub-canopy. In nature, this "*old growth*" or "ancient" stage of forest succession may take from 80-500 years to develop.

Initial soil conditions, as well as changes in soil conditions resulting from the shifting forest composition, are also important in determining the course or "trajectory" of development, though generally the role of the soil environment is less critical than that of the light environment (*42*). Factors such as the pH and mineral composition of the soil, the amount of organic material, and nutrient concentrations could all help select for or against the establishment of particular tree species, often in combination with the regional precipitation regime. Acidic, dry, and nutrient-poor soils may support establishment of more conifer species, for example, while moist calcareous and nutrient-rich soils would likely support more hardwood species. Shifts in such soil conditions over time can result from tree growth itself (for example, acid-tolerant conifers can increase soil acidity). Differences among species nutrient requirements can foster competition that may interact with stress and resource conditions to influence forest composition and function. While great variability in local soil, light, and water conditions could lead to great diversity in the composition and dynamics of developing forests, early views in ecology held that the outcome of succession is highly predictable once initial conditions are known (i.e. a "deterministic" view).

Modern views on forest succession emphasize the potential for "stochastic" factors to modify the basic process outlined above (i.e. events having a strong component of randomness or unpredictability) (*66*). Variation in plant response to changing light, soil, and water status certainly play fundamental roles in driving succession, but stochastic forces can sometimes alter or overwhelm these deterministic ones. Important stochastic processes include various disturbances produced by weather events (e.g. floods, wind storms, droughts), environmental impacts of human activities (e.g. patterns of fragmentation, pollution, changes in land use), and the inherent unpredictability of species dispersal and immigration (*67*). On the other hand, contemporary ecology also explicitly considers potential for deterministic processes, such as particular species interactions that cause ecosystem properties to be influenced by both the identities and biological diversity of herb and tree colonists (*37, 43*). Synthesis of the importance of stochastic and deterministic factors in forest ecology is ongoing (*68*), but to date ecologists are unable to strongly predict the outcome of succession under a broad range of environmental conditions and former land uses (e.g. resource extraction, urbanization, agriculture, ecological rehabilitation) (*32*).

4.1.1.4 ***Afforestation of agricultural land***

Forests have been establishing on former agricultural lands for centuries in North America (*32, 37*), and for millennia in Europe (*69*). Such forests are sometimes the outcome of active tree planting, but more often, they establish spontaneously following farm abandonment. Despite this long history, many questions remain open regarding how to establish new forests capable of fulfilling the same ecological roles as ancient ones. What factors limit the rate of forest establishment, and determine the structure of the canopy, sub-canopy, and ground layers? If the only requirements are arrival of appropriate species and sufficient time for forest succession to play-out, then active forest creation should focus on accelerating or "jump-starting" processes of species immigration and growth. However, if major environmental changes associated with farming (e.g. soil erosion, nutrient-stripping, fertilization, homogenization) were so severe that historical forest communities can never be supported, restoration should focus on either recreating historical environments or on directing succession towards new target ecosystems that are better suited to post-agricultural environments (e.g. tallgrass prairie, savanna) (*70*). Research from France on lands farmed by the Romans and abandoned nearly 2000 years ago demonstrates although the sites are presently forested, aspects of forest structure were directly influenced by the intensity of past agricultural usage (*69*). This suggests farming can have ecological impacts that are irreversible on a historical time scale. On the other hand, research from Sweden on farms previously planted with tree species reveals that after 70-80 years, understory plant communities in plantations became indistinguishable from those occupying ancient forests, although this was only true where plantations were located near ancient forests; isolated plantations had distinct low-diversity understory communities regardless of how much time passed (*21*). Afforestation therefore seems capable of promoting recovery of important forest functions in a reasonably short period, provided species immigration from ancient forests is possible. Indeed, enhancing the connectivity among remnant ancient forests may be one of the chief ecological benefits of post-agricultural tree plantations (*55*).

For similar reasons, though at a smaller spatial scale, an emerging practice in restoration-oriented afforestation is to plant trees in a clumped pattern within fields. Clumps of trees (commonly referred to as "pods") are usually planted using a a small number of tree species that have similar growth rates. Species typical of late-successional forests are planted in the centre of the pod while species typical of early successional forests are planted towards the edge. Trees are planted more closely together than on conventional plantations, and the space between pods ("inter-pod space") is usually seeded with native herbs and grasses that repel invasion by problem weed species. The combination of these factors stimulates strong competition for light among tree species, and consequently, straight, tall growth of late-successional species and expansion of pioneer species into the inter-pod space. Pods may range from 5-30 m in diameter (depending on field conditions and resources available), and are usually spaced at least 6 m apart. An advantage of the pod approach is that although each pod is comprised of only a few tree species, different pods can have different species, and particular species can be targeted to particular field patches based on patch conditions and species preferences (*43*). Furthermore, a wide diversity of tree species will eventually colonize the inter-pod space. In pod plantings the chief desire is to establish a wide diversity of species and ecosystem services that will be sustainable over the long term. Natural processes of colonization and succession are utilized, but accelerated by creating wooded “islets” within a wider, non-forested ecosystem (*40*). Such islets function as refuge habitat for important native understory species, and bases from which these species can invade the forest habitat gradually emerging in the inter-pod space.

In contrast to this ecosystem restoration-oriented approach to afforestation, conventional afforestation involves establishing plantations – often featuring a single tree species – where timber production is a guiding interest. Plantations are created by planting trees on former farmland and spacing them evenly and widely, to promote efficient tending. Species are selected based on their capacity to produce particular marketable wood products in a manner that is as quick and and cost-effective as possible. Little consideration is usually given to ecological roles of tree species, or whether they are native to the region..Traditionally in Ontario, managers planted conifers such as Red Pine and White Pine for fast timber production, with hardwoods such as Ash and Aspen used in the southern-most regions of the province. Such planting often occurred where added tree species were native but far from dominant in the region. However, despite differences between conventional plantation and applied forest restoration, plantations can perform important ecological services in addition to economic ones, including habitat-creation for understory herbs, birds, and mammals (*13, 71*). Quick-growing trees planted for economic reasons actually provide strong ecological benefits in the form of a rapidly-emerging shaded understory environment favourable for regeneration of both native understory herbs and late-successional tree species (*43*).

Plantations are often designed with the intention of “thinning”, or harvesting some trees as soon as they are marketable (e.g. after 30 years). Thinning functions to open-up growth space and increase light availability, causing remaining trees to expand their basal areas and extend lateral branching, and creating a greater diversity of understory light conditions capable of supporting a greater diversity of understory species. However, conventional plantations generally result in canopy-forming trees that are spaced considerably further, and that have larger basal areas, than in natural forests (*32*).

While plantations involve far fewer tree species than restoration initiatives, there is growing interest in establishing plantations with 2-5 tree species that “combine well” and lead to greater wood production than in monocultures (*72*). Such species may utilize different resources and therefore compete less, or may even reduce environmental stress and facilitate growth. In addition to enhanced wood production, multi-species plantations may have benefits for biodiversity, such as suppressing exotic invaders and providing heterogeneous habitat for native ground cover.

4.1.1.5 ***Understory vegetation as indicators of forest functioning***

Ecologists have long recognized that certain plant species can serve as useful indicators of ecosystem properties including plant productivity, soil fertility, soil moisture, light intensity, and the species diversity. Good indicators are species that have evolved to thrive under very narrow ranges of environmental conditions, and thus occur naturally only in certain distinct habitat patches. Plant species that have adapted tolerance to deep shade, for example, are weak competitors or become physiologically damaged in higher-light environments, and thus serve well as indicators of mid- to late-successional forests (*60*). A chief benefit of using plants rather than direct measurement to characterize the environment is that good indicator species are well adapted to multiple environmental factors that correlate in nature and have aspects that are difficult or expensive to measure directly. Inferences can be made about a complex suite of environmental variables influencing forest floors (e.g. soil nutrient ratios, soil microorganisms, light intensity, presence of non-native competitors) based on occurrence or performance of characteristic shade-adapted forest understory herbs (*61, 73, 74*). Rare species do not make great indicators because absence of such species from a site may be due to unsuitable environmental conditions, or it may be due to lack of immigration to the site. While this issue can be confounding for any species, at least non-threatened species are likely to be available for seeding or transplantation to test sites in order to assess environmental conditions. When an indicator species is used experimentally to test environmental conditions – e.g. transplanted to a location and monitored for growth, survival, and reproduction – it is referred to as a "phytometer". Very common species make for poor indicators or phytometers because they are generally adapted to a wide range of environmental conditions and thus do not indicate distinct suites of conditions.

Adding species directly in the context of a controlled transplantation experiment has additional potential to identify ecological and management factors limiting the similarity of test sites to target ecosystems. For example, adding Trillium plants directly to test afforested locations without any additional treatments and monitoring survival over time can indicate how similar each location is to the Trillium's "home" habitat, from the Trillium's point of view. However, if the transplants do not survive, it would be difficult to determine whether light conditions were inadequate, whether soil conditions were inadequate, or perhaps whether competition with existing exotic invasive understory species (e.g. Garlic Mustard) precluded Trillium success. Failure of the Trillium could also simply be due to the stress of transplantation, regardless of other factors. Implementation of a transplantation control treatment (e.g. movement of Trillium individuals to new locations only a few meters away from their old ones), or creation of a competition control treatment (e.g. removal of existing understory vegetation prior to Trillium addition) could address these sources of uncertainty in powerful ways.

While not every understory species is a suitable indicator of ecosystem properties such as ancient forest canopy, light and soil conditions, previous research has characterized and scored most of the understory flora of temperate forests of eastern North America based on precisely these features. Thus, analysis of the literature yields a short list of candidate species with strong potential to indicate facets of ancient-forest understory conditions that are of keen interest to afforestation managers. A recently published Shade Tolerance Index for understory plants presents many species with high but not extreme shade-tolerance, suitable for indicating closed-canopy conditions of older-growth forests (*75*). A subset of these species strongly indicates aspects of ecosystem recovery in degraded southern Ontario forests (*64*). Of these species, a few meet criteria of being relatively common to ancient forests throughout Ontario but intolerant of other habitats, and of being attractive, recognizable components of Ontario's natural woodland heritage. While final species selection in transplantation experiments is ultimately subject to availability of source populations, four understory herb species likely to be available, amenable to transplantation in general, and meeting the above-described criteria are: Painted Trillium (*Trillium undulatum*), Hairy Solomon's Seal (*Polygonatum pubescens*), Canada Mayflower (*Maianthemum canadense*), and Downy Yellow Violet (*Viola pubescens*). Each of these species is a "spring ephemeral", or herb that flowers and fruits in spring but exists primarily below ground for most of the year. Survival and reproduction of such plants following transplantation would strongly indicate understory environmental conditions characteristic of ancient Ontario forests. Such transplantation may have the additional benefit of reclaiming lost habit for these important natural-heritage species.

4.1.1.6 ***Research goals, hypotheses, and predictions***

The primary goal of the proposed work is to improve our understanding of the ecology of former farmlands planted with tree species. Specifically, we hope to discover the extent to which such locations resemble ancient forests with respect to ecosystem properties that are critical but possibly difficult to replicate in planted forests: light, soil, and competition conditions capable of supporting characteristic native understory species. We will assess the degree to which afforested sites resemble ancient forests by transplanting understory herb species known to indicate ancient forest conditions, from locations of natural occurrence to farms planted with trees at various points over the past 120 years. Comparison of indicator species (phytometer) performance among the different-aged sites will enable determination of the time span over which succession from planted saplings to fully functional forest occurs. Additionally, performance differences among sites exposed to different management practices (e.g. monoculture plantations vs. multi-species forest-restoration projects) will help indicate what steps managers should take achieve ancient-forest conditions most rapidly. The specific management practices to be investigated cannot be determined until a large list of potential sites is created (Phase 1 of the proposed work, and currently underway), but contrasting plantations against restorations, single-species tree plantings vs. multiple-species plantings, or coniferous vs. deciduous tree plantings are all strong possibilities.

Understanding factors limiting phytometer survival is crucial to discovering methods for accelerating forest development, and thus we will assess what ecological mechanisms related to site age limit the survival of phytometers. To do so, we will experimentally alter transplantation conditions in four different ways at each site. We hypothesize that the light environment is the primary regulator of phytometer survival, and thus a major filter on the equivalence of recently planted forests to ancient ones. However, competition with invasive non-native herb species, inadequate soil fertility, and lack of appropriate soil microorganisms are also likely to play important roles in regulating forest development on tree-planted farms. Factors such as the numbers and identities of different tree species planted may also affect the rate of forest development. We predict that ecosystem properties of ancient forests, as estimated by transplanted indicator herbs, will emerge most rapidly at planted sites where canopies have closed but feature some breaks, where competition with other herb species has been suppressed, and where physical, chemical, and biological conditions typical of ancient forest soils have been achieved. We additionally predict that sites planted with mixed native deciduous species will resemble target forest more rapidly than conifer plantation sites. Based on theory and previous studies in other regions, we expect phytometers to reach optimal survival performance (in the absence of soil amendments and competitor-removal) 60-70 years following tree planting. As this timeline is similar to the lifespan of many aggregate extraction operations, empirical confirmation of this prediction would suggest that planting tree species on agricultural lands just prior to developing new aggregate extraction operations would effectively mitigate negative ecological impacts of removing existing forests to access aggregate deposits.

4.2 Appendix B: Detailed Methods Used in the Afforested Environments Study

4.2.1 Study sites

We created a database of locations in southwestern Ontario where large-scale tree-plantings had occurred, as well as mature natural reference forests, focussing on afforested lands previously used for agriculture because this is the land-use most commonly selected for conversion to new forests. We obtained records from government agencies responsible for managing forests on public lands, confining our geographical scope to locations within ≈130 km of the University of Waterloo to ensure a high number of replicate sites could be sampled repeatedly within a growing season. To be considered, records for planted stands needed to include the year of planting, a map of the area planted, the identities and proportions of different tree species planted, and general indication of the nature and frequency of subsequent stand-thinning management. In total, 122 suitable records were provided by regional municipalities (Dufferin County, Halton Region, Norfolk County, Simcoe County and York Region), provincial Conservation Authorities (Ausable-Bayfield, Grand River, Halton, Long Point Region, Toronto Region, and Upper Thames River), provincial Ministries (Natural Resources, Transportation), and private conservation organizations (rare Properties, the Nature Conservancy Canada).

These sites were drawn from the thousands of plantation stands within the study area by requesting from each agency what they considered to be good examples of stands falling into each of several different age classes (i.e. “Young”, “Middle”, “Old” and “Oldest”, respectively planted 30-35, 40-55, 60-70, or 75-90 year ago), general composition groups (i.e. “M”, “X”, and “H”, respectively planted with softwood monocultures, softwood mixtures, and mixed softwoods and hardwoods) and thinning categories (i.e. “Thinned” and “Under-thinned”, respectively experiencing and not experiencing selection and/or row thinning every 7-15 years following the first 30 years of stand development; e.g. under-thinned includes stands that were never thinned as well as those with abnormally long gaps between successive harvests). “Good” examples were defined as easily-categorized stands where sampling and experimental manipulation were likely to be permitted as well as logistically feasible. Excellent examples of mature natural hardwood forests (including but not limited to stands generally considered to exhibit “old-growth” or “older-growth” features) were also requested, for potential use as reference forest ecosystems.

Selections represent the types of stands that were most commonly found in the landscape; sites planted with softwoods only, for example, are more abundant and span a larger age gradient than sites planted with mixtures of softwoods and hardwoods, and thinned sites were more abundant than under-thinned sites. Only a few softwood species accounts for the majority of planted trees across all sites (i.e. red pine, white pine, and white or Norway spruce) while stands receiving hardwood species were usually planted with white ash, black walnut or poplars, though exceptions include black cherry, maple, hickory and basswood at a few sites. Stands planted with only hardwood species were sought, but these tended to be too young and too few to fit well with the goal of identifying changes over time and across thinning practices.

A final suite of 36 planted and 5 natural stands were selected from the 122 potential sites by randomly choosing several replicate sites within each combination of Age, Composition, and

Thinning classes. More replicates were selected from class combinations representing the most commonly practiced afforestation strategies (e.g. regularly-thinned monoculture softwood plantations), as these spanned the widest age gradients and could provide the background for expected forest development following conventional methods. Sufficient replicates were selected from the less common groups (e.g. mixed softwood-hardwood plantings, under-thinned plantations) to test whether patterns of change over time differed from this background. To meet these goals as well as disperse replicates of the same treatment combinations across the geographic expanse of the study region, some selected sites were rejected and replaced by randomly choosing new sites from small groups of criteria-appropriate options. Selected stands representing relatively uncommon species compositions were similarly rejected and replaced to avoid ambiguity attributing results to main factors of interest versus the presence of unusual species. The list of 122 potential sites as well as the 41 selected sites (highlighted rows) are presented in Table 3.1; a map showing the locations of the final study sites is presented in Fig. 3.1.

4.2 Detailed Methods (Appendix B) – Table 4.1

Table 4.1: List of Potential and Final Study Sites

Stand type	Thinned Regularly?	Site Age (years)	Age Class	Tree Species Planted	Site Name	Site Owner/Manager	Latitude	Longitude	Included (ID Code)
Softwood plantation	Yes	16	Young	White Pine	NCC	Nature Conservancy Canada	42°37'51.00"N	80°32'38.77"W	
		27	Young	White Pine	LongpointYoungPine	Long Point Region Conservation Authority	42°39'14.58"N	80°28'34.05"W	
		29	Young	Red Pine	Clark Tract	York region	44° 1'47.79"N	79°18'39.54"W	
		31	Young	White Pine	Luther Marsh	Grand River Conservation Authority	43°58'10.08"N	80°28'1.64"W	31MT
		34	Young	Red Pine + White Pine	Hilton Falls	Conservation Halton	43°30'38.55"N	79°58'18.64"W	34XT
		35	Young	Red Pine	Simmons tract	Dufferin County	44°14'23.73"N	80° 4'15.72"W	35MT
		35	Young	Red Pine	Coughlin tract	Simcoe Region	44°29'34.58"N	79°49'13.70"W	
		35	Young	Red Pine	Coughlin tract	Simcoe Region	44°29'53.05"N	79°49'34.25"W	
		40	Middle	Red Pine + White Pine	Kolodzieg2	Toronto Region Conservation Authority	43°57'23.55"N	79°57'32.22"W	40XT
		40	Middle	White Spruce	Kolodzieg1	Toronto Region Conservation Authority	43°57'28.88"N	79°57'36.07"W	
		41	Middle	Red Pine	Randwick Tract	Dufferin County	44°14'52.74"N	80° 4'20.01"W	
		44	Middle	White Pine	Wolverton Tract	Ontario Ministry of Natural Resources	43°16'10.97"N	80°31'28.15"W	
		44	Middle	White Spruce	WildwoodSpruce	Upper Thames River Conservation Authority	43°14'50.54"N	80°58'56.89"W	
		46	Middle	White Pine	Conely Tract	Halton Region	43°32'3.27"N	80° 5'19.05"W	
		46	Middle	Red Pine + White Pine	Swick king	Long Point Region Conservation Authority	42°44'59.77"N	80°25'3.86"W	46XT
		46	Middle	White Pine	Ballycroy2	Toronto Region Conservation Authority	43°58'28.65"N	79°54'18.41"W	
		48	Middle	Red Pine	Clark Tract	York region	44° 1'42.40"N	79°19'5.78"W	48MT
		49	Middle	Red Pine + White Pine + White Spruce	Esquesing	Conservation Halton	43°32'19.06"N	79°56'56.24"W	
		49	Middle	White Pine	WildwoodPine62	Upper Thames River Conservation Authority	43°14'37.51"N	80°59'56.64"W	49MT
		51	Middle	Red Pine + White Pine	Coughlin tract	Simcoe Region	44°29'42.73"N	79°49'54.90"W	
		51	Middle	White Pine	WildwoodPine60	Upper Thames River Conservation Authority	43°15'24.92"N	81° 0'19.99"W	
		52	Middle	Red Pine + White Pine	Coughlin tract	Simcoe Region	44°29'33.98"N	79°48'32.36"W	
		52	Middle	White Pine	Hay Swamp 3	Ausable-Bayfield Conservation Authority	43°22'27.02"N	81°32'17.48"W	
		53	Middle	White Pine	Bellwood Lake	Grand River Conservation Authority	43°47'47.37"N	80°18'37.05"W	
		53	Middle	White Pine + Red Pine	Bellwood Lake	Grand River Conservation Authority	43°47'48.96"N	80°18'32.31"W	
		53	Middle	White Pine + White Spruce	Hay Swamp 1	Ausable-Bayfield Conservation Authority	43°21'31.46"N	81°33'3.46"W	53XT
		53	Middle	White Pine + White Spruce	Luther Marsh	Grand River Conservation Authority	43°58'44.86"N	80°26'46.92"W	
		53	Middle	Red Pine	Randwick Tract	Dufferin County	44°14'58.45"N	80° 3'59.99"W	53MT
		53	Middle	Red Pine	Coughlin tract	Simcoe Region	44°29'43.64"N	79°49'55.54"W	

	54	Middle	Red Pine + White Pine	Ballycroy1	Toronto Region Conservation Authority	43°58'39.83"N	79°53'28.46"W	
	54	Middle	Red Pine + White Pine + White Spruce	Luther Marsh	Grand River Conservation Authority	43°55'21.31"N	80°26'28.52"W	
	54	Middle	White Pine	Luther Marsh	Grand River Conservation Authority	43°55'10.29"N	80°26'24.26"W	
	55	Middle	Red Pine + White Pine	Luther Marsh	Grand River Conservation Authority	43°55'56.00"N	80°24'3.24"W	
	55	Middle	Red Pine + White Pine	Palgrave-Kelly	Toronto Region Conservation Authority	43°57'27.86"N	79°51'54.59"W	55XT
	55	Middle	Red Pine + White Pine + Norway Spruce	Finny tract	Halton Region	43°33'40.82"N	80° 0'58.79"W	
	55	Middle	Red Pine + White Spruce	Finny tract	Halton Region	43°33'45.63"N	80° 1'1.73"W	
	56	Middle	Red Pine + Scotch Pine	Swick King	Long Point Region Conservation Authority	42°41'50.03"N	80°24'34.66"W	
	59	Old	Red Pine + White Pine	Hickling Tract	Simcoe Region	44°25'58.15"N	79°44'44.17"W	
	61	Old	Red Pine	Randwick Tract	Dufferin County	44°14'59.49"N	80° 3'30.06"W	
	65	Old	Red Pine	Dainty	York region	44° 1'44.35"N	79°19'17.34"W	65MT
	66	Old	White Pine + White Spruce	Bellwood lake	Grand River Conservation Authority	43°44'23.11"N	80°20'35.53"W	66XT
	68	Old	Red Pine	Randwick Tract	Dufferin County	44°15'4.87"N	80° 4'23.95"W	
	71	Old	Red Pine	Hendrie tract	Simcoe Region	44°28'59.93"N	79°47'38.01"W	
	72	Old	Red Pine	Main Tract	Dufferin County	44°12'45.59"N	80° 2'27.35"W	
	72	Old	Red Pine	Main Tract	Dufferin County	44°12'45.85"N	80° 2'20.64"W	
	76	Oldest	Red Pine	Randwick Tract	Dufferin County	44°15'3.23"N	80° 3'29.08"W	76MT
	81	Oldest	Red Pine + White Pine	Tottenham Tract	Simcoe Region	44°28'38.15"N	79°48'21.65"W	81XT
	81	Oldest	Red Pine	Crawford 2	Conservation Halton	43°27'22.33"N	79°57'14.62"W	
	86	Oldest	Red Pine	Hollidge Tract	York region	44° 3'59.93"N	79°17'47.78"W	
	87	Oldest	Red Pine + European Larch	Hollidge Tract	York region	44° 3'56.65"N	79°18'1.80"W	
	87	Oldest	Red Pine	Eldred-King	York region	44° 3'33.68"N	79°18'35.68"W	
	87	Oldest	Red Pine	Hollidge Tract	York region	44° 3'57.14"N	79°17'50.60"W	
	87	Oldest	Red Pine	Hollidge Tract	York region	44° 3'55.55"N	79°18'3.59"W	87MT
	87	Oldest	Red Pine	Hollidge Tract	York region	44° 4'6.39"N	79°17'25.53"W	
	90	Oldest	Red Pine	LeFerberve tract	Norfolk County	42°40'35.02"N	80°30'3.80"W	90MT
	99	Oldest	Red Pine	Eldred-King	York region	44° 3'39.10"N	79°18'32.02"W	
	103	Oldest	White Pine	St. Williams	Ontario Heritage Trust	42°41'36.67"N	80°26'35.96"W	
No	22	Young	White Pine + White Spruce	Crawford Lake South	Conservation Halton	43°28'4.89"N	79°56'30.03"W	
	23	Young	White Pine	Bellwood Lake	Grand River Conservation Authority	43°46'50.28"N	80°19'8.20"W	
	23	Young	White Pine	Bellwood Lake	Grand River Conservation Authority	43°46'39.87"N	80°19'18.18"W	
	26	Young	White Pine	Leeming tract	Dufferin County	44°14'48.37"N	80° 3'25.43"W	
	26	Young	White Pine	Main Tract	Dufferin County	44°12'44.65"N	80° 2'40.67"W	
	26	Young	White Pine	DufferinYoung?	Dufferin County	44°14'4.16"N	80° 7'59.75"W	

4.2 Detailed Methods (Appendix B) – Table 4.1

		29	Young	White Pine + White Spruce	Crawford Lake North	Conservation Halton	43°28'33.98"N	79°57'5.31"W	
		30	Young	White Pine + White Spruce	Bellwood Lake	Grand River Conservation Authority	43°44'49.60"N	80°19'7.82"W	
		30	Young	White Pine + White Spruce	Bellwood Lake	Grand River Conservation Authority	43°44'49.55"N	80°19'27.74"W	30XU
		30	Young	White Pine	Bellwood Lake	Grand River Conservation Authority	43°44'40.38"N	80°19'21.18"W	
		35	Young	White Pine + White Spruce	Robertson tract	Halton Region	43°31'19.32"N	79°59'55.59"W	35XU
		38	Middle	White Pine + White Spruce	Fleming Tract	Ontario Ministry of Natural Resources	43°43'54.06"N	80°45'10.29"W	38XU
		38	Middle	Red Pine	Robertson tract	Halton Region	43°31'16.58"N	79°59'50.41"W	
		40	Old	Red Pine	Morriston	Ontario Ministry of Transportation	43°27'2.60"N	80° 7'34.85"W	
		40	Old	Red Pine	Morriston	Ontario Ministry of Transportation	43°26'57.41"N	80° 7'38.83"W	40MU
		42	Middle	White Pine + European Larch	Horner tract	Ontario Ministry of Natural Resources	43°12'17.77"N	80°37'51.25"W	42XU
		44	Middle	White Pine	Wolverton Tract	Ontario Ministry of Natural Resources	43°16'16.25"N	80°31'22.97"W	
		46	Middle	Red Pine + White Pine	Swick king	Long Point Region Conservation Authority	42°45'1.05"N	80°25'4.32"W	46XU
		49	Middle	White Pine	Hay Swamp 3B	Ausable-Bayfield Conservation Authority	43°21'26.61"N	81°32'12.65"W	49MU
		60	Old	Red Pine + White Spruce + Jack Pine	Snyder Tract	Halton Region	43°28'2.55"N	79°59'43.62"W	60XU
		61	Old	Red Pine + Norway Spruce	Patterson tract	Ontario Ministry of Natural Resources	43°17'15.73"N	80° 8'18.57"W	61XU
		69	Old	Norway Spruce + White/Scotch/Jack Pine	Kirkwall	Ontario Ministry of Natural Resources	43°22'17.09"N	80° 9'25.05"W	69XU
		83	Oldest	White Pine	Tottenham Tract	Simcoe Region	44°28'30.34"N	79°47'48.04"W	83MU
Softwood-Hardwood Plantation	Yes	19	Young	White Pine + Black Walnut	WildwoodWalnutPine	Upper Thames River Conservation Authority	43°15'5.95"N	81° 4'5.21"W	
		26	Young	White Pine + White Ash + Scotch Pine	Binbrook	Ontario Ministry of Natural Resources	43° 7'28.10"N	79°44'17.85"W	
		31	Young	White Pine + Black Walnut	Parkhill Campground k4	Ausable-Bayfield Conservation Authority	43.160411	-81.667859	31HT
		32	Young	White Pine + Black Walnut	Thomson tract P7	rare properties	43°22'22.49"N	80°22'22.24"W	32HT
		39	Middle	White Pine + Basswood	Coulson_m10_2	Halton Region	43.573824	-79.845194	39HT
		40	Middle	White Pine + White Ash	Coulson_m10_1	Halton Region	43.574351	-79.844701	40HT
		51	Middle	Scotch Pine + Jack Pine + White Cedar + Ash	Finny tract	Halton Region	43°33'43.93"N	80° 0'54.34"W	
		55	Middle	White Spruce /Pine + Ash + Cherry	Finny tract	Halton Region	43°33'44.96"N	80° 1'6.04"W	
		59	Old	Red/White Pine + Ash + White Spruce + Cherry	Foulds Tract	Grand River Conservation Authority	43°15'17.58"N	80°20'21.33"W	59HT
		64	Old	Red Pine + White Pine + White Ash	Bellwood Lake	Grand River Conservation Authority	43°46'59.68"N	80°19'58.27"W	64HT
		65	Old	White Pine + White Spruce + White Ash	Bellwood Lake	Grand River Conservation Authority	43°46'49.20"N	80°20'6.51"W	
	No	30	Young	White Pine + Norway/White Spruce + Ash	Bellwood Lake	Grand River Conservation Authority	43°44'32.60"N	80°19'20.63"W	
		30	Young	White Pine + White Ash	Bellwood Lake	Grand River Conservation Authority	43°44'31.36"N	80°19'21.01"W	
		33	Young	White Pine + Black Walnut	OHT	Ontario Heritage Trust	42°38'6.53"N	80°32'39.60"W	33HU
		36	Middle	White Pine + Burr Oak + Walnut +Maple + Ash	Nith river tract	Ontario Ministry of Natural Resources	43°17'25.58"N	80°30'7.71"W	
		40	Middle	White Pine + White Elm	Coulson tract	Halton Region	43°34'31.82"N	79°51'0.71"W	
		40	Middle	White Pine + White Spruce + Basswood + Tulip	Coulson tract	Halton Region	43°34'41.68"N	79°50'43.00"W	

4.2 Detailed Methods (Appendix B) – Table 4.1

		50	Middle	White Pine + White Elm	Bellwood Lake	Grand River Conservation Authority	43°44'10.77"N	80°18'54.26"W	
		55	Old	White Pine + White Ash	McKendrick Tract	Grand River Conservation Authority	43°15'13.79"N	80°19'59.61"W	55HU
		56	Old	White Pine + White Spruce + White Ash	Savage tract	Ontario Ministry of Natural Resources	43°20'58.19"N	80° 7'30.75"W	56HU
		60	Old	White Pine/Spruce + Ash + Tulip Poplar + Jack Pine	Snyder Tract	Halton Region	43°27'50.54"N	79°59'53.89"W	
Hardwood plantation	Yes	14	Young	Red/Burr Oak + Ash + Russian Olive + Locust + Tulip	Conrad (NCC)	Nature Conservancy Canada	42°37'37.53"N	80°32'32.07"W	
		15	Young	Mixed Hardwoods (composition uncertain)	WildwoodYoungHW	Upper Thames River Conservation Authority	43°22'0.22"N	81°11'41.62"W	
		16	Young	Mixed Hardwoods (composition uncertain)	Sydneyback	Long Point Region Conservation Authority	42°49'18.77"N	80°28'52.51"W	
		36	Middle	Butternut + Hickory + Cottonwood + Oak + Maple	Parkhill Campground j8	Ausable-Bayfield Conservation Authority	43.166334	-81.668923	
	No	18	Young	Silver Maple	LongpointMaple	Long Point Region Conservation Authority	42°39'13.67"N	80°28'49.09"W	
		18	Young	Red Oak	LongpointOak	Long Point Region Conservation Authority	42°39'14.78"N	80°28'49.51"W	
		26	Young	White Ash	Cayuga	Ontario Ministry of Natural Resources	42°54'26.20"N	79°44'12.07"W	
		36	Middle	Ash + Silver Maple + Alt-leaved Dogwood	Nith river tract	Ontario Ministry of Natural Resources	43°17'23.28"N	80°30'1.25"W	
		36	Middle	Black Walnut + White Ash + Hard Maple	Nith river tract	Ontario Ministry of Natural Resources	43°17'24.39"N	80°30'9.36"W	
		43	Middle	Black Walnut	Coulson tract	Halton Region	43°34'39.77"N	79°50'54.03"W	
		43	Middle	Black Walnut	Coulson tract	Halton Region	43°34'33.28"N	79°50'37.54"W	
Reference Hardwood Forest	(Some)	REF	REF	Reference Hardwood	Parkhill	Ausable-Bayfield Conservation Authority	43° 9'40.49"N	81°38'55.31"W	RPAR
		REF	REF	Reference Hardwood	Crawford 2	Conservation Halton	43°27'20.21"N	79°57'19.26"W	RCRA
		REF	REF	Reference Hardwood	Redstone Bush	Grand River Conservation Authority	43°59'26.01"N	80°26'18.33"W	
		REF	REF	Reference Hardwood	Hughes tract	Long Point Region Conservation Authority	42°53'58.83"N	80°42'43.13"W	RHUG
		REF	REF	Reference Hardwood	Hyde tract	Ontario Ministry of Natural Resources	43°20'1.84"N	80° 8'36.36"W	RHYD
		REF	REF	Reference Hardwood	Cliffs and Alvars	rare properties	43°22'57.99"N	80°21'2.13"W	
		REF	REF	Reference Hardwood	Wildwood	Upper Thames River Conservation Authority	43°14'19.26"N	81° 1'54.43"W	
		REF	REF	Reference Hardwood	Hollidge Tract	York region	44° 4'6.45"N	79°17'9.83"W	
		REF	REF	Reference Hardwood	Hendrie tract	Simcoe Region	44°29'53.39"N	79°49'33.25"W	RHEN
		REF	REF	Reference Hardwood	Glen Haffy	Toronto Region Conservation Authority	43°56'30.92"N	79°56'36.20"W	

4.2.2 Field sampling

4.2.2.1 *Canopy vegetation*

The structure of the arboreal forest layer (i.e. the tree community, excluding saplings) was assessed identically within each stand using the Point-Centred Quarter Method (*76-78*). This required sampling at 20 randomly-located points per stand (selected using random numbers of paces along randomly-oriented transects). At each point, the transect was bisected by an imaginary perpendicular line and the plane of the forest floor was divided into four quadrants. Within each quadrant, the distance to the nearest tree trunk > 2cm in diameter (1.5 m above ground) was measured, as was trunk diameter and tree species identity. This enabled calculation of average distances between neighboring trees and projection of the density of the stand (i.e. number of individual trees per hectare), as well as the number of species in the local tree community (richness), the relative abundances of the different species in the tree community (evenness, or equitability), and the Shannon diversity of species (i.e. an index summarizing the combination of richness and evenness information). The Shannon index generally ranges between 0 and 3.5, with higher values corresponding to communities featuring either a greater number of species, more equivalent proportions of different species, or both.

A Shannon calculation was also used to describe the diversity of different trunk-diameter size-classed represented per stand (“trunk-diameter diversity). This was done by considering the number of different decametre-scale diameter size-classes (i.e. < 10 cm, 11-20 cm, 21-30 cm…etc.) represented per stand and the number of individual trees falling into each class. Stands featuring multiple classes of trunk-girth thus received a higher index value than those where most trees are virtually the same diameter, and stands where trees are evenly distributed across size-classes had a higher index value than those where only unusually wide or narrow trunks were present. This index should correlate with natural stand development over time in both planted and natural forests as some trees grow to full maturity while others die or are removed and become replaced by new saplings.

The overall community composition of each stand was described as a matrix of each species present and its corresponding Importance Value (a metric summarizing three aspects of the species’ abundance: the average frequency with which it is encountered, density where it does grow, and the cross-sectional area of space occupied by living stems).The relative similarity of the tree community at each planted stand to the communities inhabiting reference stands was estimated by calculating the mean Bray-Curtis similarity coefficients for simulated communities based on different combinations of the reference and plantation composition matrices. Given a group of sample composition matrices, Bray-Curtis similarity estimates the level of compositional overlap between each possible pair of matrices (in this case, accounting for both the presence and Importance of shared species, relative to species which are unique to one sample or the other). The mean Bray-Curtis value is the sum of values across all pair-wise comparisons divided by the number of comparisons made, and thus estimates the average level of sample-to-sample compositional similarity given a group of sample compositions.

We sought to estimate relative similarity of each plantation community to the reference state by creating a Relative Similarity scale that has at its lowest value the mean Bray-Curtis similarity for a group of six samples: the five reference stands and one fictional stand of average species richness but having no species in common with the reference stands. At the high end of the scale is the mean Bray-Curtis similarity for the group of five reference stands plus one fictional stand with an identical composition of species as to one of the reference stands (a duplicate matrix of each reference stand was substituted into the calculation in turn and the average was taken). The high end of this scale thus corresponds to “test” communities that are approximately as similar to the group of reference-forest communities as the reference communities are to one another, while the low end corresponds to “test” communities that have completely dissimilar composition to the group of reference communities. The composition matrix for each plantation stand was then substituted into the similarity calculation in turn, with the mean Bray-Curtis similarity value for the group of five reference communities plus the plantation community functioning as a convenient estimator of the relative similarity of that plantation stand to the reference state (provided the mean value for groups featuring only reference stands are shown as the “bar” to which plantation values are held to).

Point-quarter data was not collected from one site (40XT) due to unexpected ongoing forestry activities at the time of sampling.

In addition to canopy-layer data from the Point-Centred Quarter sampling, the degree of canopy closure was assessed at five random locations per stand using a spherical densitometer. This is a small convex mirror featuring a superimposed grid; when held level at chest height and 45-60 cm away from the body, it reflects the mosaic of closed tree canopy and open sky above the forest floor. Canopy closure was measured at each point by facing each cardinal direction in turn and counting the number of grid squares in the central portion of the view field reflecting open sky, with the mean of these four readings used to calculate percent canopy closure (e.g. if 4 of the 16 central squares are dominated by reflected sky, canopy closure is approximately (12/16 x 100%=) 75%).

4.2.2.2 *Understorey vegetation*

The structure of the understorey plant community was assessed using standard quadrate methods (*79*). Temporary 1 m^2 plots (2.0 m x 0.5 m) were established at 21 random locations within each stand and the identity of each vascular plant species found within each plot was recorded. This included ferns, graminoids (grasses, sedges, rushes) wildflowers, shrubs, vines and saplings, but not non-vascular species such as mosses and lichens (which are dealt with elsewhere; see methods for coarse woody debris and microtopography). Plants were counted as present in the plot if any portion of their above-ground biomass occupied any space above the plot. Abundance was estimated for each species at each site as the frequency with which it was encountered (i.e. the number of plots were presence was recorded, divided by 21), while overall cover by living vegetation (ignoring species identities) was measured at 5 separate (2.0 m x 0.5 m) plots. Each plot in this second group was located randomly within the stand and subdivided into 8 (0.25 m x 0.25 m) cells. Within each cell, cover by living vegetation was visually classified on a scale ranging from 0 (no vegetation present) to 4 (76-100% of cell surface area covered by vegetation; classes 1-3 used for cover ranging from 1-25%, 26-50%, and 51-75%, respectively). Cover by bryophytes (i.e. mosses and liverworts) was calculated in the same manner using the same plots. Values for each variable were summed across subplots and divided by Mean percent cover by vascular and non-vascular vegetation was thus calculated for each plot, and for each stand by averaging the 5 plot values.

Aspects of the diversity and composition of the understorey vegetation community in each stand were calculated similarly to the canopy-layer metrics, but with some key differences. At the scale of the 1 m^2 plots, the average number of species present (richness) was the main diversity measure of interest (recall that within-plot abundances were not recorded and thus evenness and Shannon diversity cannot be calculated). At the whole-site scale, relative frequencies of encountering each species was used to calculate evenness and Shannon diversity as well as site-level richness. While the total number of different species encountered across all sampling plots provides a good estimate of the complete list of all species present on the site (also known as the site species pool), a second, more accurate estimate of species pool was obtained by adding species observed on-site outside of the sampling plots, either during a 30 minute walkabout carried out by two researchers, or during subsequent site visits while carrying out other components of the study (each stand was visited more than a dozen times between 2011 and 2014). As both estimates of site richness produced similar patterns when included in multiple analyses, here we primarily present data using the second, more encompassing version of site species pool richness (hereafter simply referred to as “site richness”).

In this same vein, two different approaches for calculating community composition were explored, one considering only data from the 21 sampling plots per site and one considering data from the 21 plots plus a 22nd simulated plot comprised of all species recorded on-site not represented within the 21 plots. Again, as there were no major differences in overall patterns of results, data from the latter, more inclusive approach is presented here. The understorey of each stand was thus represented as a group of 22 binary data sets, denoting whether each species in the site pool was absent (“0”) or present (“1”) in each plot. A scale for assessing similarity of each planted stand to the group of 5 reference stands was developed by first calculating the mean level of compositional similarity between two sets in a group of 132 binary species composition sets: 110 sets corresponding to the 22 sample plots at each of the 5 reference sites, plus 22 sets featuring no species in common with any of the reference sets. This was done by calculating similarity between each possible pair of sets the group was calculated using the Jaccard method (which is ideal for binary data, whereas Bray-Curtis similarity requires counts or proportions), accounting equally for the number of species shared and the number of species unique to each set in the pair. The sum of the coefficient values across all possible comparisons in this group, divided by the number of comparisons made thus provides a mean Jaccard coefficient reference value for a single stand that is as dissimilar as possible from the reference state, in a context that accounts for natural levels of variability in species composition both within and between reference stands. Likewise, a reference value for a single stand that is reasonably similar to the reference state –in this same context of moderate variability both within and among individual sites corresponding to the reference state – was obtained by calculating mean Jaccard similarity for a group of 122 binary composition sets including the 110 reference sets plus a duplicate of one of the reference sets (actually calculated separately for duplicates of each reference set in turn, and then averaged across trials), Planted stands were then assessed in similar manner, with each individual stand assigned the value of the mean Jaccard coefficient for the group of 122 sets including all reference sets plus the 22 sets corresponding to the plots sampled in that stand. Comparing plantations based on these value is thus a comparison of relative compositional similarity to the reference state; although values of the simulated minimum and maximum similarity stands were not required for such analyses, they are useful as visual references providing the context of such comparisons and are thus included in visual representations of the results of analyses.

4.2.2.3 *Microhabitat and soil properties*

Fine-scale features of the forest floor potentially important to plant growth ("microhabitat") were evaluated using the same methods and plots used for assessment of vegetation cover, except surface cover by various substrates other than living vegetation were measured. As with vegetation, surface cover on a scale from 0-4 was assessed in each (0.25 m x 0.25 m) subplot (with 8 subplots per plot, and 5 plots per stand), looking specifically at cover by rocks, bare soil, organic litter attributable to fallen leaves, litter attributable to fallen needles, and fine woody debris. Cover by each feature was assessed independently, with values were summed across subplots then divided by 64 to determine the proportion of total plot area covered by that feature. Proportions were multiplied by 100% to express as a percentage, and average percent cover per site was calculated for each feature by taking the mean of the 5 plot values per site. Similar methods were previously used with high success in characterizing microhabitat features in heterogeneous rock-outcrop ecosystems (*80*)

Soil was collected from each stand and sent to SGS Agri-Food Laboratories (Guelph) for analysis. One full soil core (2 cm diameter x 20 cm deep) was taken from each of the five plots per stand used for assessing microhabitat and vegetation cover. Soil samples were collected within a three week period in autumn 2012, and submitted for analysis immediately. In the lab, three soil cores per site were selected at random and tested for moisture content and bulk density (by weighing, drying to constant mass, and reweighing to calculate mass of water lost, and by dividing the dry weight of the soil sample by the known fresh volume of the sample in the field to calculate bulk density). The three dried samples per site were combined then divided into two samples for assessing particle size distribution (percent sand, silt and clay) and texture class. The two cores per site that were not used in these analyses were combined and analyzed for multiple nutrient properties using Mehlich 3 extraction methods (*81*). Reported soil variables include acidity (pH), available phosphorous (P ppm), potassium (K ppm), calcium (Ca ppm), magnesium (Mg ppm), and aluminum (Al ppm) concentrations, as well as cation exchange capacity (CEC MEQ/100g) and organic matter (OM %). Standards and methods advocated by the Association of Official Organic Chemists (AOAC) International (*82*) were used for all analyses.

4.2.2.4 *Coarse woody debris and microtopography*

Stands were surveyed in summer 2013 for coarse woody debris (CWD) features including downed logs, stumps and standing dead trees ("snags"), as well as variation in microtopography including the frequency of encountering small pits and mounds on the forest floor. Only 31 of the 36 plantation stands were successfully sampled due to ongoing forestry activities at 5 sites. A surveying protocol capable of rapidly evaluating both sets of variables was developed by adapting transect methods for sampling CWD (*83*) and pit-mound microtopography (*84*) previously found to be previously under similar conditions to the present study. Three Y-shaped transects composed of three 10 m segments were placed at random non-edge, non-overlapping locations within each site and walked once to sample CWD, and then a second time to assess microtopography. During the initial walk, each CWD object coming into contact with the transect (and having stem diameter > 7cm at its widest point) was sampled. The object was classified as a log, stump or snag, stem diameter was measured at each end, the length of the main stem was measured, and the average state of decay of the object was assessed using a five-point scale. The key used to classify each objects decay state, as published in (*83*), was reprinted here for reference (Fig. 3.2). CWD objects exhibiting virtually no signs of decay fall into class 1 while those with virtually no structural integrity left due to the extent of wood decomposition fall into class 5.

4.2 Detailed Methods (Appendix B) – Fig. 4.1

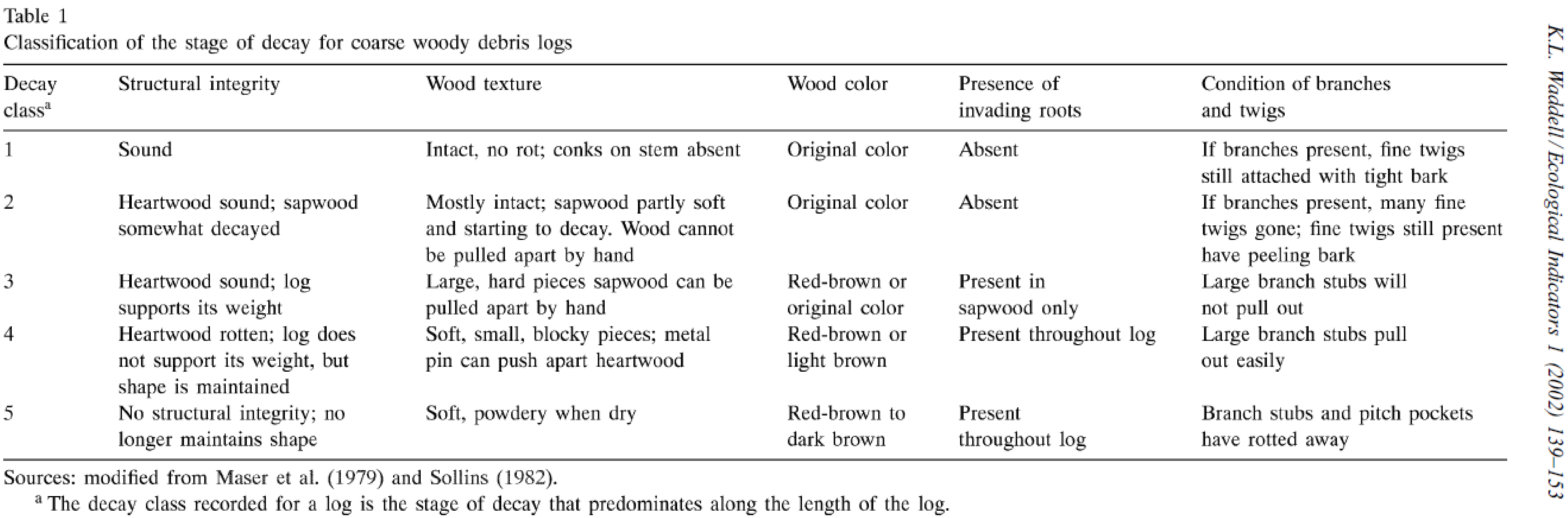


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Table 1
Classification of the stage of decay for coarse woody debris logs

Decay class[a]	Structural integrity	Wood texture	Wood color	Presence of invading roots	Condition of branches and twigs
1	Sound	Intact, no rot; conks on stem absent	Original color	Absent	If branches present, fine twigs still attached with tight bark
2	Heartwood sound; sapwood somewhat decayed	Mostly intact; sapwood partly soft and starting to decay. Wood cannot be pulled apart by hand	Original color	Absent	If branches present, many fine twigs gone; fine twigs still present have peeling bark
3	Heartwood sound; log supports its weight	Large, hard pieces sapwood can be pulled apart by hand	Red-brown or original color	Present in sapwood only	Large branch stubs will not pull out
4	Heartwood rotten; log does not support its weight, but shape is maintained	Soft, small, blocky pieces; metal pin can push apart heartwood	Red-brown or light brown	Present throughout log	Large branch stubs pull out easily
5	No structural integrity; no longer maintains shape	Soft, powdery when dry	Red-brown to dark brown	Present throughout log	Branch stubs and pitch pockets have rotted away

Sources: modified from Maser et al. (1979) and Sollins (1982).

[a] The decay class recorded for a log is the stage of decay that predominates along the length of the log.

Figure 4.1. An image of the classification key for coarse woody debris decay status, originally published in Waddell (2002).

Each sampled CWD object was also evaluated for number of vegetative species apparently using the object as habitat, either rooted in the decaying wood or using it to physically support plant biomass. The number of different species within each of several possible life-form groups (including trees, shrubs, grasses, herbs, ferns, mosses, liverworts and several classes of lichens, as defined in the Ontario Plant List (*85*)) associated with each CWD object was recorded. The identities of these species were not recorded, however, due to time constraints and the large number of sites that needed to be surveyed. We acknowledge that this necessary sacrifice limited our ability to infer patterns of species diversity using CWD as habitat at scales larger than the individual CWD object (i.e. if one object supports 4 species and another supports 8, the collected data does not allow determination of the number of species occupying the combination of the two objects), however, we were able to calculate for each stand the average number of species colonizing each CWD object, as well as the average number of life-form groups represented and the Shannon diversity of life-form groups (where abundance of each group per object is estimated as the number of different species present on the object belonging to the group). Each of these estimators of the diversity of organisms using CWD objects as habitat performed similarly in multiple exploratory analyses, so the most straightforward measure is mainly focussed on here, the total number of plant species present.

Using the calculations recommended in (*83*), the information about CWD length, diameters, and frequency of encounter was manipulated to determine for each stand the total volume of CWD on a cubic-metre per hectare basis, as well as volume of each object and the frequency and clustering of objects. Consideration of decay-status information enabled calculation of the mean level of decay for CWD objects within each stand. Additional calculations incorporating each object’s type and decay state status enabled analysis of CWD at each site as a “community” composed of up to 15 different “species” based on the combination of 5 decay and 3 type classes (i.e. “decay 1 logs” is one species while “decay 5 stumps” is another). Each “species” was considered to have an “abundance” at each site equivalent to the proportion of total volume attributable to CWD objects falling into the pertinent combination of CWD type and decay classes. This framework enabled calculated of the richness, evenness and Shannon diversity of CWD “species” per stand, as well as the degree of compositional similarity between planted and reference stands, with respect to these “species” (using the Bray-Curtis approach taken for living tree communities).

Furthermore, for every CWD object encountered this way, the number of vegetative species using the object as habitat was assessed. This involved assessing both how many different life forms were growing directly on the object (including trees, shrubs, grasses, herbs, ferns, mosses, liverworts, lichens, fungus and algae), and how many different-appearing organisms in each group were present. For example, a log supporting both blue and orange lichen spots was evaluated as supporting two lichen “species”, even if the specific identities of these could not be determined in the field. The purpose of this evaluation was to help determine whether the functionality of CWD as habitat for biodiversity could be as strong in planted forests as in natural ones, as well as the amount of time needed for planted forests to develop such functionality.

4.2.2.5 *Microtopography*

Variation in microtopography was assessed by walking transects and stopping every 1 m to evaluate whether the forest floor at that point on the transect is more-or-less level or at-grade with the majority of the surrounding forest floor (within 20 cm, above or below grade; classified as “matrix” microtopography), or whether it is substantially lower elevation (> 20 cm below-grade, classified as “pit” microtopography), or substantially higher elevation (> 20 cm above-grade, classified as “mound” microtopography). This information provided for calculation of the frequency of occurrence of pit, matrix and mound features at each stand (i.e. the total number of occurrences of each feature across all transects, divided by 90, the total number of sampling points per site). Finally, using the same approach employed in the CWD survey, the total number of species using each microtopography feature was assessed. However, as microtopography features were less defined in space than the CWD objects, diversity sampling was conducted by centring a 1-m diameter circular plot at each stopping point along each transect and counting all of the species present within this boundary (again, recording life-from but not species identity of each).

4.2.3 Phytometer experiment

4.2.3.1 *Phytometer overview*

We carried out a plant relocation experiment to assess the capacity for planted forests to provide high-quality habitat suitable for understorey vegetation typical of mature natural hardwood forests. Habitat quality can be a complex property dependent on many interactive environmental factors, such that direct measurements can be a difficult and expensive due to the need for knowledge about which factors are important and how they interact even before measurements are made. The indirect approach of using phytometers (“plant-indicators”) can be much more efficient, as this requires relatively little expense but can provide clear, comprehensive and intuitive indication of habitat quality speaking directly to the suite of factors that are most important to the organisms of interest. Beforehand knowledge is still required however, to ensure that the plant species selected to serve as phytometers are ecologically appropriate. A good phytometer must adequately represent the broader community or ecosystem features of interest. They must be sensitive, requiring a relatively narrow range of environmental conditions typical of the habitat of interest. In a landscape featuring multiple locations of a target habitat, a good phytometer will occur frequently across locations, it will grow to high abundance when it does occur in the target habitat, but it will show very low colonization or establishment success in habitats other than the target ecosystem. Additionally, a good phytometer must not be too sensitive; for example, it would not be effective if it exclusively required conditions consistent with only a narrow subset of wider range of environmental conditions characterizing the target habitat. Likewise, it would not be effective if it responded more negatively to the act of relocation itself (e.g. to root disturbance) than to specific environmental differences between reference and test locations. To a large extent, these latter challenges can be overcome by including a layer or experimental control involving relocation of phytometers within their home locations, as well as to test locations, such that a baseline expectation for mortality due to transplantation shock unrelated to habitat differences can be established. Additionally, the longer that phytometers are followed over time, the more likely that impacts of fundamental rather than spurious habitat differences will be reflected by phytometers.

4.2.3.2 *Phytometer selection*

We selected phytometers with an appropriate level of sensitivity to understorey habitat conditions within mature hardwood forests, and representative of the plant communities typically flourishing in is habitat, though a careful process of literature review, site visitations and consultation with experts. A list of potential species was drawn from research specifically looking at plants capable of indicating the success of hardwood forest recovery in southern Ontario (*64*), as well as a comprehensive study of southern Ontario's best examples of old-growth forest (*27*). Perennial herbaceous species were focussed on because they held the best hope for both rapid responses in the short term and potential for successful and informative long-term monitoring. Species with a dormant life phase and discreet energy-storing overwintering organs (e.g. bulbs, rhizomes, tubers) were additionally desirable as manipulation of such organs during the dormant phase would minimize the likelihood of negative impacts of transplantation shock. As consideration of such factors steadily narrowed the list of potential phytometers, the top candidates were presented to each forest ecologist or manager involved in the research (e.g. those assisting with locating appropriate study sites) and opinions on the suitability of each, as well as alternative options, were sought. Finally, as potential study plantations and reference forests became available and were visited, phytometer candidates which occurred frequently and abundantly in the latter locations but not at all in the former, rose to prominence on the list. Similarly, candidates that grew abundantly in reference forests but were also found along trails, ditches and fields adjacent to the target forest fell in position on the list.

The outcome of this complex and iterative process was a selection of two herb species that were generally agreed upon as excellent phytometer candidates, and sensitive to different aspects of mature understorey habitats. Wild ginger (*Asarum canadense)* is a shade-tolerant, low-growing herb species with paired kidney-shaped flowers that emerge in mid-spring and persist through late autumn. Its fleshy underground shoots form clumped rhizomes just below the soil surface. The species spreads slowly through clonal growth and seed production, which are dispersed by ants and thus migrate relatively short distances. It is a member of the Aristolochiaceae ("Birthwort") family and has a broad history of culinary and medicinal usage in North America by aboriginal peoples. Wild ginger's evolved capacity for high shade tolerance makes it a likely indicator of similarity of the light environment in particular to its ancestral environment. However, its frequent success in mesic and limestone-rich forests as well as its obligate relationships with mycorrhizal fungi suggest it is a significant indicator of soil quality as well. In contrast, the second selected phytometer species is primarily an indicator of soil and microhabitat quality other than the light environment. Wild leek (*Allium tricoccum*) is an ephemeral spring vegetable, emerging as a single long, pale green leaf from its overwintering bulb in early spring and completing its yearly above-ground growth phase by dying back (if not consumed by herbivores) before the surrounding dense tree cover leafs-out. By early summer there is virtually no trace of the species above-ground, until a non-photosynthetic flowering stalk emerges in mid to late summer and the iconic campanulate white flowers open. Like wild ginger, the seeds produced are dispersed by ants and thus migrate relatively short distances, while individual bulbs divide clonally slowly over time. Wild leek is a monocot and member of the Liliacaea (“Lily”) family, with historical as well as modern culinary use. The species’ adaptation of avoiding rather than tolerating shade suggests it is a good indicator of forest habitat quality with respect to features other than the light environment. It has evolved in a forest environment and does not fare well in open fields or meadows despite its capacity to tolerate light, likely due to poor competitive ability. However, it is only active in the forest when variation in light quality is not likely to play a role in plant survival and reproduction, and thus is likely limited by other factors, such as soil richness and moisture, duff-layer properties, and the activities of competitors for nutrients and space.

4.2.3.3 *Collection of soil and plant material for phytometer experiment*

Natural populations of wild ginger and wild leek were located and flagged in spring and summer 2011 in the 10 potential reference stands where permission to collect plant material had been granted (all but the *rare* Properties site, Table 3.1). In autumn, several large (>10 m^2) patches supporting relatively dense populations of either or both target species were selected at each site and used as sources for individual plants for relocation. This was done by using a spade and trowel to extract 5-10 turves (0.3 m x 0.3 m x 0.3 m) of forest floor from random locations within each patch, which were gently cut and pulled apart by hand to extract individual wild leek bulbs and/or wild ginger rhizomes. Each turve yielded approximately 10-20 individual bulbs and/or rhizomes that were deemed suitable for transplantation, after returning to the soil material which may be been damaged during extraction or which measured less than 3 cm along the greatest dimension. Interconnected rhizomes were separated by cutting with secateurs (leaving 5 cm of the connecting root attached to each rhizome) while bulbs were gently pulled apart by hand. Each individual bulb or rhizome was cleaned during or shortly after collection by carefully pulling and brushing off attached soil by hand, however a fine layer of soil was usually left intact if it did not fall off easily, in order to minimize damage to the fine roots protruding from the bulbs and rhizomes. All cleaned bulbs and rhizomes from the same site were packaged together with commercially purchased sterile peat moss in large paper bags and stored in darkness at room temperature for 1-3 weeks until they could be replanted in new forest locations. In total, 100-200 bulbs and 100-200 rhizomes were collected from each of the 10 reference stands.

Soil for use in the phytometer experiment was collected in bulk from each of the reference stands where wild leek and ginger root material was collected. Turves of soil were extracted in the same manner as for the plant material, but all visible non-soil matter (e.g. plants, roots, rocks, sticks) was removed by hand prior to transferring turves to 30L Rubbermaid bins. Turves were homogenized using a spade following transfer, and multiple turves were added and intermixed until each bin was full. Turves were extracted from randomly selected locations which were within or adjacent to the larger patches supporting high densities of target species, but which themselves had a minimal cover of above-ground vegetation. Depressions in the forest floor created by turve extraction were filled and covered as much as possible using soil, leaf litter and woody debris from the surrounding area. Approximately 100-200 L of soil was collected from each reference stand in summer 2011 and stored covered and in darkness at room temperature until the fall out-planting.

Approximately one-half of the bulk soil bins collected from each stand was subjected to steam-sterilization using an autoclave machine located in the plant growth facilities of the Department of Integrative Biology at the University of Guelph. The contents of each 30 L soil bin were distributed evenly to two large autoclave bags, each of which was placed open in an autoclave bin with soil spread as flat and shallow as possible to promote maximum infiltration of steam. Each bag of soil was sterilized on a standard “wet” cycle, with steam reaching 121 C steam for at least 15 minutes, followed by slow venting.

Each plantation stand was assigned a reference stand to serve as the source of soil and plant material for the phytometer experiment. Designations were based on geographical proximity, with each reference stand serving as the source for the nearest 3-5 plantation stands. No plantation was greater than 50 km from its corresponding reference stand, and the majority of plantations were within 10 km of the reference. Out-planting materials were prepared for each plantation stand by placing one wild leek bulb and one wild ginger rhizome together in each of 30 1 L zipper-lock freezer bags, 10 of these bags additionally received 1 L of reference-stand soil which had been sterilized while 10 of the remaining bags received 1 L of reference-stand soil which had not been sterilized. Similar preparation was required for the five reference stands selected for inclusion in the phytometer experiment, in which soil and plant material was to be relocated to new patches proximal (0.5-1.5 km) but separate from those where it had been collected.

4.2.3.4 *Experimental design*

Phytometers were out-planted to the 5 reference stands and 36 plantation stands in October 2011. At each stand, 10 randomly-selected non-edge trees were selected to serve as “bases” for locating phytometer-receiver plots. Three such plots were established 2 m from the trunk of each base tree, spaced equidistantly at compass positions of 0, 120 and 240 degrees. If the forest floor at a designated plot was deemed unsuitable to receive phytometers (e.g. flooded or extremely rocky terrain, or occupied by a tree) the plot was moved to the closest suitable location that was still in line with the designated compass bearing and 1-4 m from the base tree. If no location within this range was suitable, a new randomly-located base tree was chosen. As base trees were marked in several ways, this system enabled rediscovery of phytometer plots even if the in-plot flags were to be removed (which occurred on several occasions). Phytometers were planted in the designated plots by inserting a trowel 20 cm into the forest floor and loosening all soil within a circular area 20 cm in diameter. Resident vegetation or surface features occupying this area were removed along with 1 L of soil from the centre of the plot. In plots not designated to receive soil from the reference stand, half of the just-removed soil was returned to the plot, one wild ginger rhizome and one wild leek bulb were laid on top (10 cm apart and in root-down orientation), and these were covered by the remainder of the just-removed soil. In plots that were to receive reference-stand soil, the soil packed in the bag with the plant material was used instead of the soil just removed from the plot, but added in this same manner below and above the bulb and rhizome (i.e. producing a planting depth of 5 cm). Aside from removed vegetation, surface cover materials which had been removed from the plot (e.g. leaf litter and fine woody debris) were returned after planting, though no larger rocks or coarse woody debris objects were placed directly on top of the planted vegetation. At each base tree, one plot did not receive reference-stand soil, one plot received autoclaved reference-stand soil, and the third plot received unsterilized reference-stand soil; these three treatments were randomly allocated to the three plot locations.

Phytometer plots were visited multiple times throughout 2012, 2013 and 2014. At each occasion, the presence of either or both phytometer species as living above-ground plants was noted. At some junctures we additionally recorded the length of the longest leaf and any evidence that flowering had occurred, but this data was not included in the final analyses due to the possibility that such features speak more to the condition of the bulb or rhizome at the time of collection than to the status post-transplantation environment. To simplify analyses of transplant survival, we focussed on presence-absence data collected once annually for each phytometer species separately. Data collected in early spring each year was used to assess wild leek while that collected in early summer was used to assess wild ginger.

4.2.4 Analysis

4.2.4.1 *Canopy, understorey, duff layer and soil variables*

We independently analyzed each of the 42 forest features independently, using a similar approach for each. Data including the spontaneously developing understorey, canopy, duff layer and soil variables met assumptions of General Linear Models (GLM) – including normality and homogeneity of variance – and were thus analyzed using Proc GLM in SAS (v. 9.13, The SAS Institute, Cary, NC). Each of these response variables was initially modelled simply as a function of stand type (i.e. stand membership within H, T, S, or R groups), with the least-squared mean (lsmean) values and 95% confidence intervals for the R group taken as the target level and range for that feature in analysis of plantation similarity to natural forests. The significance of the lsmean difference between R and each of the other groups was tested using Dunnett's adjustment for multiple comparisons to a single control.

The second layer of analysis for these variables was to ignore the data from the R stands and focus on the plantation stands only, modelling each response variable as a function of stand type (H, T, or S), the amount of time passed since stand planting, and the interaction between these. For response variables found to be unrelated to time, the initial model was returned to and the similarity of each plantation type to the reference forests was calculated based on the lsmeans comparisons. For responses found to be related to stand type and time since afforestation, the intercept and slope for the relationship with time was calculated and plotted for each stand type separately. These relationships were in turn interpolated or extrapolated to determine at which point along the x-axis (time since afforestation) the y-value for the regression line intersected the lsmean value obtained for R from the first stage of the analysis. The slopes and intercepts for these relationships were compared among the plantation types using contrast statements, for example to determine if the rate of increasing similarity to R was greater for one plantation type or another. For response variable that were related to time since afforestation but not stand type, intercepts and slopes (and associated inter/extrapolations) for the relationship with afforestation time were calculated for the group of all plantation types combined ("P", plantations).

4.2.4.2 *Coarse woody debris and micro-topographic features*

During sampling of coarse woody debris (CWD) and micro-topographic feature (MTF) variation (i.e. pits and mounds), extenuating circumstances including forestry activities at research sites prevented full sampling of four of the plantation stands, reducing the total sample size across the age gradient to 31 stands. Likely related to this, data for every CWD and MTF response variable behaved similarly in exhibiting a less normality and homogeneity in variance –

particularly among the stand type groups – than the other types of response variables. To help account for this, relevant data was analyzed using Generalized (rather than General) Linear Models (Proc Genmod in SAS), with identity as the link function, so as to perform analysis analogous to analysis of variance (analysis of deviance) but without such a strict requirement for normally and homogeneously distributed data. Perhaps also related to the reduced sample size for this analysis, but the effect of plantation type (alone or through interaction with time since afforestation) was not significant for any variable, and thus final models tended to show each specific response variable as a simple function of time since afforestation for all plantations as a single group (again, with the same inter/extrapolations as the other analyses to calculate time-lags until equivalence with R). To enhance comparability among variables and models, a "goodness-of-fit" or "variance explained" metric analogous to the R^2 statistic from GLMs was adopted: the square of the Pearson correlation between the observed response variable data for a given feature and the data predicted for that feature under the particular model being tested/compared (*31*) (designated here as R^{2*}).

While the MTF variables related to the frequency of pits, mounds and matrix-level forest-floor patches was analyzed in the same way as the CWD variables – i.e. as functions of stand type and time since afforestation – we recognized that multiple factors aside from MTFs may influence the number of plant species using a given patch as habitat (most notably, the number of species in the site species pool), such that efforts to detect the influence of MTFs might fail if the influence of relevant covariates were not accounted for. To this end, analysis of the number of species using different MTFs as habitat considered both plantation and reference stands together and evaluated the number of species per MTF sampling plot as a function of stand type (H, S, or R), site-level species richness, the type of MTF characterizing the plot [pit ("-1"), matrix ("0") or mound ("+1")], and the interaction between site richness and MTF type. Contrast statements were used to test whether the rate at which plot richness potentially increased with site richness differed among pit, matrix, and mound plots, as would be expected if some MTFs are especially supportive of plant biodiversity. This analysis also considered potential effects of other measured site-level variables as covariates helping to predict plot-level diversity, and retained tree density, surface cover by bare soil, and soil sand content as significant factors in the final model (i.e. that with the best explanatory power, as judged by the model's R^{2*} value). A repeated-measures statement was included to account for potential auto-correlation related to taking multiple measures within each site on a single sampling date.

4.2.4.3 *Phytometer responses*

The analysis of phytometer survival over three growing seasons following relocation to new growth environments (herb-relocation plots) was also conducted using Generalized Linear Models (Proc Genmod), but with the natural logarithm (Ln) taken as the link function because the phytometer data followed a binary distribution (i.e. presence or absence of the out-planted individual at various points in time and space). This analysis essentially transforms the collection of presence/absence data points for a given stand (or a level of treatment within each stand) to the "log of the odds" (or log-odds) of selecting an individual sampling point at random and finding that the phytometer was indeed present. When considering the potential effects of the soil addition treatment (3 levels x 10 plots each, per stand), for example, if the no-soil-added control level (C) in a given stand at a given point in time exhibited 4 plots with the phytometer present and 6 plots without, the probability of observing an individual at a plot may be 0.4 (or 40%), but the odds are 4:6, which is converted to a fraction (4/6=0.6666) before the log is calculated ($\text{Log}_e(0.6666) = -0.41$). The analysis then considers how variance in this value is related to factors potentially responsible for phytometer survival or failure. Note that 50% survival corresponds to the logarithm of 1, which is 0; as survival improves from this point, the log-odds approaches infinity, while it approaches negative infinity as survival decreases. However, as the log-odds value for a stand with no observed phytometers would be undefined, a transformation was performed to retain all studied stands in the analysis, whereby the few stands exhibiting 0% phytometer presence were taken as exhibiting 0.5% survival for the purposes of the analysis (providing a "floor" log-odds value of -5.29 in plots on this scale). To maintain easy interpretability of graphs showing phytometer performance, the log-odds data points and regression lines are shown but a second y-axis showing how these relate to the more intuitive axis of percent-probability of phytometer presence was included.

Allium tricoccum

For the *Allium* phytometer, the log-odds of species presence within herb-relocation plots was first explored as a function of stand type with a focus on stand-planting rather than thinning differences (i.e. types H, M (monoculture), X (mixture), and R), as well as the date of sampling (the spring of the first, second, or third year after the fall 2011 out-planting), and the potential interaction between these. However, as year 1 and year 3 data were very similar but year 2 data showed nearly a complete absence of *Allium*, we concluded sampling in year 2 must have occurred too late to detect the species, and thus year 2 data was eliminated from the analysis. Analysis of just year 1 and 3 data revealed that overall patterns of results were nearly identical between years, though tending to drop in year 3 in H stands, and M and X stands were indistinguishable in their performances. Thus, in order to promote simplicity and retain statistical power for detection of potentially more determinants of phytometer success, *Allium* was reanalyzed as a function of simplified stand type (H, S (the combination of monoculture and mixed Softwood-only stands) or R), with sampling date ignored as an explanatory variable but considered as a potential source of autocorrelation, along with repeated measures in space, through use of the Repeated statement within Proc Genmod (i.e. the subject of repeated measures was declared to be herb-relocation plots within soil-addition levels within stands, with first-order auto-correlation expected). The lsmean (and 95%CI) log-odds of *Allium* presence in R stands was taken as the reference value for how well relocated phytometers can be expected to perform in habitats known to be suitable for the species (i.e. phytometers failing to survive in R likely do so due to transplantation shock rather than unsuitable habitat conditions). As with the other analyses, the next stage required ignoring R data and modelling *Allium* presence in plantation stands only, assessing how phytometer presence was related to stand type (H or S), time since stand-planting, and the interaction between these. Slopes and intercepts of relationships between phytometer presence and time since afforestation were compared and further analyzed to project the x-axis (time) position where phytometer presence in each plantation type intersects the lsmean value determined for R stands. A final level of analysis for *Allium* focussed on patterns observed in the third year only and considered potential differences among regularly-thinned (T) and under-thinned (U) plantation stands, as well as mixed hardwood-softwood (H) stands, with respect to changes in phytometer performance over time since afforestation. Initial exploratory analysis of potential effects of soil-addition treatments found no important influence on *Allium*'s behaviour as a phytometer, and is not discussed further here.

Asarum canadense

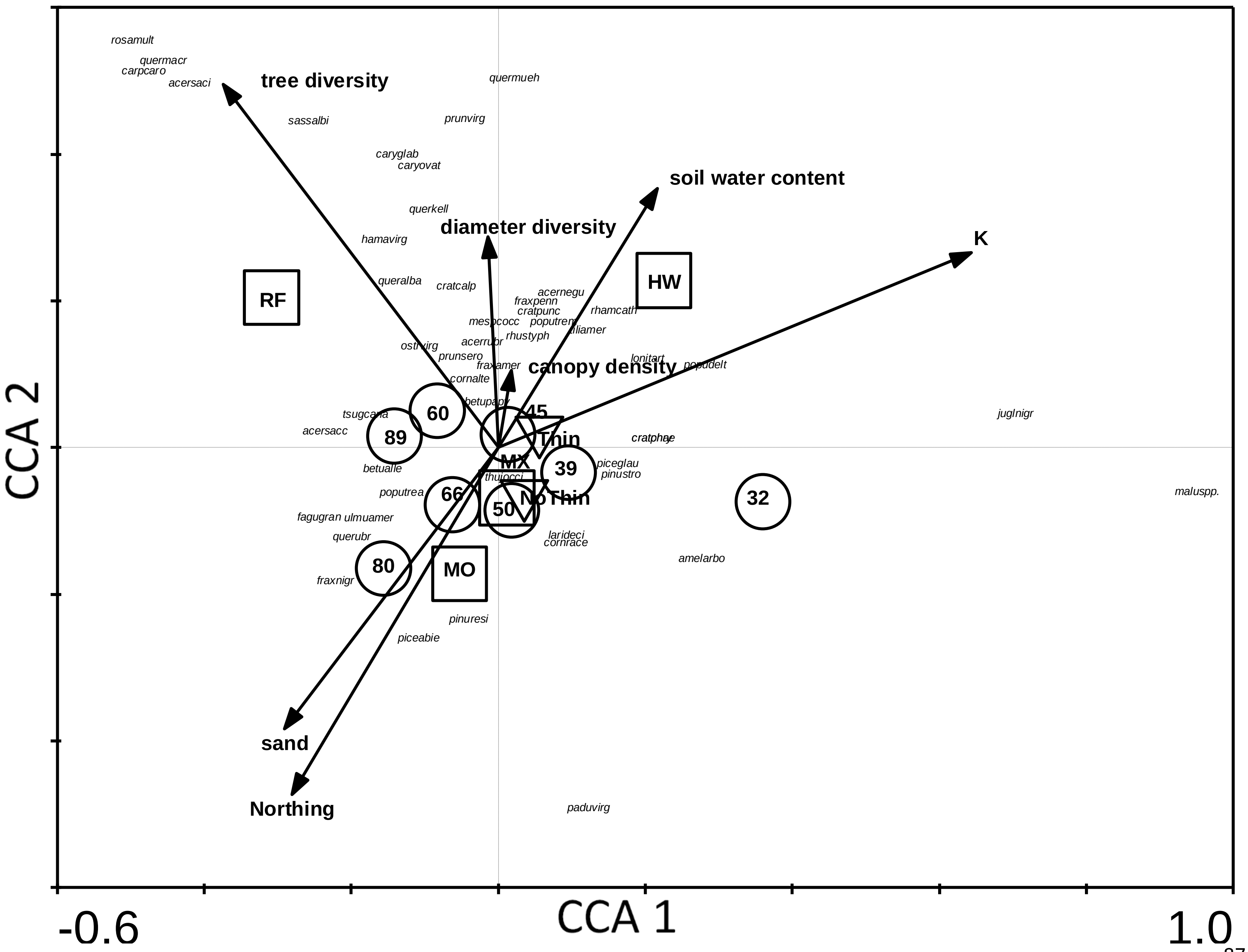
Analysis of the *Asarum* phytometer followed a similar course as *Allium*, except that all three years of data were included and phytometer presence dropped steadily across the three sampling dates. As this drop was observed in both the plantation and the reference stands, and inclusion of sampling date as an explanatory factor complicated models and reduced capacity of other effects, following illustration of these changes over time, time-averaged pattern were analyzed in several steps as per the *Allium* analysis. First, differences between H, S, and R stands were assessed in order to determine best-expected relocation success based on lsmean occurrence in R stands. Second, R data was ignored and *Asarum* occurrence in H and S plantations were compared with respect to relationships with time since afforestation. Third, as phytometers in H stands were found to be unrelated to time since afforestation, patterns of change over the afforestation timescale in S plantations only were focussed on in more detail, with consideration given to differences between thinned and underthinned plantations of both monoculture (MT and MU, respectively) and mixed-softwood (XT and XU) varieties. Analyses all considered the full set of plantation data, including potential for auto-correlation due to repeated measures, but differed mainly with respect to how stand types were defined and contrasts were employed.

Effects of the soil-addition treatments on *Asarum* performance were first evaluated through a generalized linear model of phytometer occurrence across all sampling dates as a function of stand planting-type (H, M, X or R), soil-addition treatment [control (C), sterilized soil (S), or not-sterilized soil (N) added], and the interaction between these. As this revealed no significant effect of soil-addition alone or through interaction with stand type, the second stage of analysis considered plantation-data only and explicitly modelled *Asarum* presence as function of a time since afforestation and the interaction between this factor and sampling date, stand types (incorporating both planting and thinning groups), and soil-addition. In this case, sampling date was taken as a continuous variable, such that contrast statements could be specified to focus on results from the middle of the study (i.e. 20 months after out-planting). This enabled calculation of a slope (and associated significance test) for the relationship between *Asarum* occurrence and time since afforestation for each soil-addition level within each stand type. As such, we were able to declare contrast statements testing within each stand type whether the rate of increasing similarity to R over time since afforestation was different between S, C and N plots. Specific contrasts tested different specific hypothesis related to the idea that lack of soil-level similarity between reference and plantation stands constrains the rate of increasing phytometer-level similarity over time, while accounting for the fact that different types of such relationships may exist in different types of plantations. The final best model for *Asarum* in this context included contrast statements explicitly testing afforestation-time relationships in C vs. S. vs. N plots in both XT and MU stands.

4.3 Appendix C: Supplementary Results of the Afforested Environments Study

Fig 4.2. Canonical correspondence analysis (CCA) for canopy community composition, based on tree importance rather than simple abundance at each site. The analysis explained 33% of species data and 58% of species-environment relationships; the first two canonical axes are plotted here (CCA 1 = 0.433, CCA 2 = 0.327). Eight-letter labels correspond to the Species ID Code defined in the Appendix and represent occurrence of individual species. Geometric shapes represent centroids of site classification variables (circles=age group, squares=afforestation planting treatment, triangles=stand-thinning regime). Sites were classified as reference forests (RF) or placed into age groups that minimized age variance within groups and maximized it among groups, while keeping the number of sites per group more or less consistent. Numeric age-group labels corresponds to the mean age of sites within each group. To show similarities as well as differences among site groups, classification variables were retained in the final model regardless of their significance during Monte Carlo testing. In contrast, environmental regression variables (shown as vectors with lengths proportional to explanatory power) were only retained if p<0.25. The most significant explanatory variables were soil K (p=0.002), stand identity as RF (p=0.002), soil sand content (p=0.002), tree Shannon diversity (p=0.014), geographic UTM Northing (p=0.014), and soil H2O content (p=0.058). The proximity on the plot of one species label to any other or to a site classification centroid is proportional to the likelihood of the first species occurring alongside the second or at a site consistent with the grouping variable respectively. Position of a species label relative to the orientation of the environmental gradients indicated by regression vectors corresponds to the relative quantity (i.e. low ▯ high) of the variable associated with optimal species occurrence.

4.2 Supplementary Results (Appendix C) – Fig. 4.2



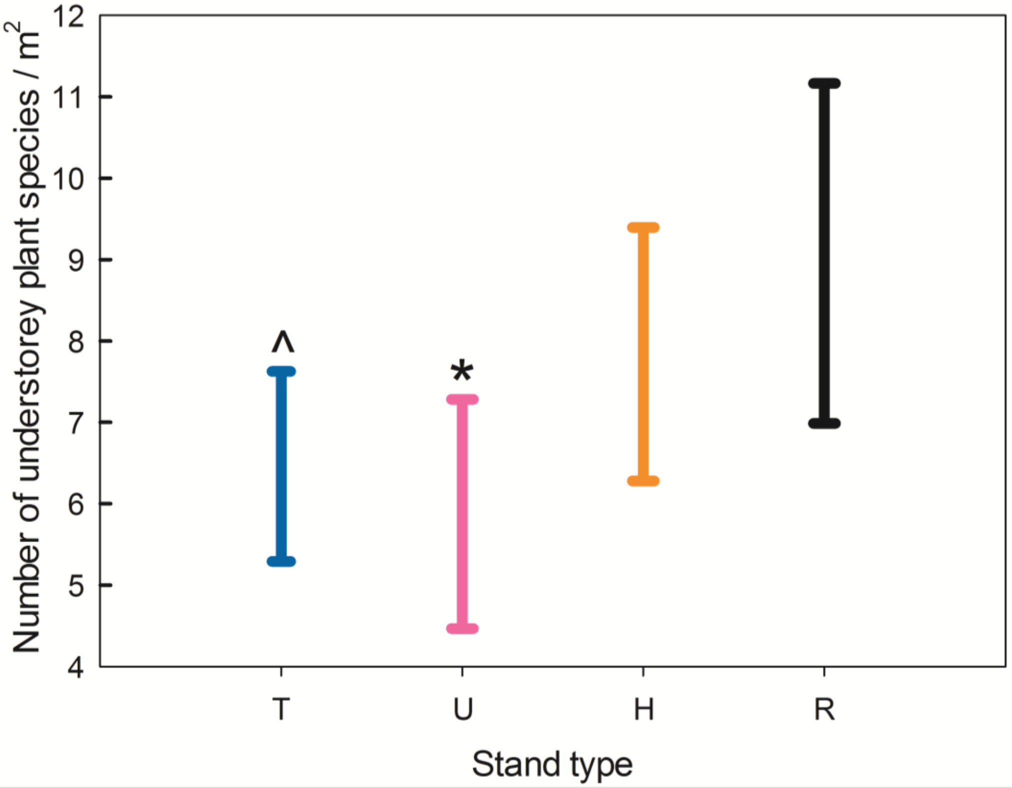
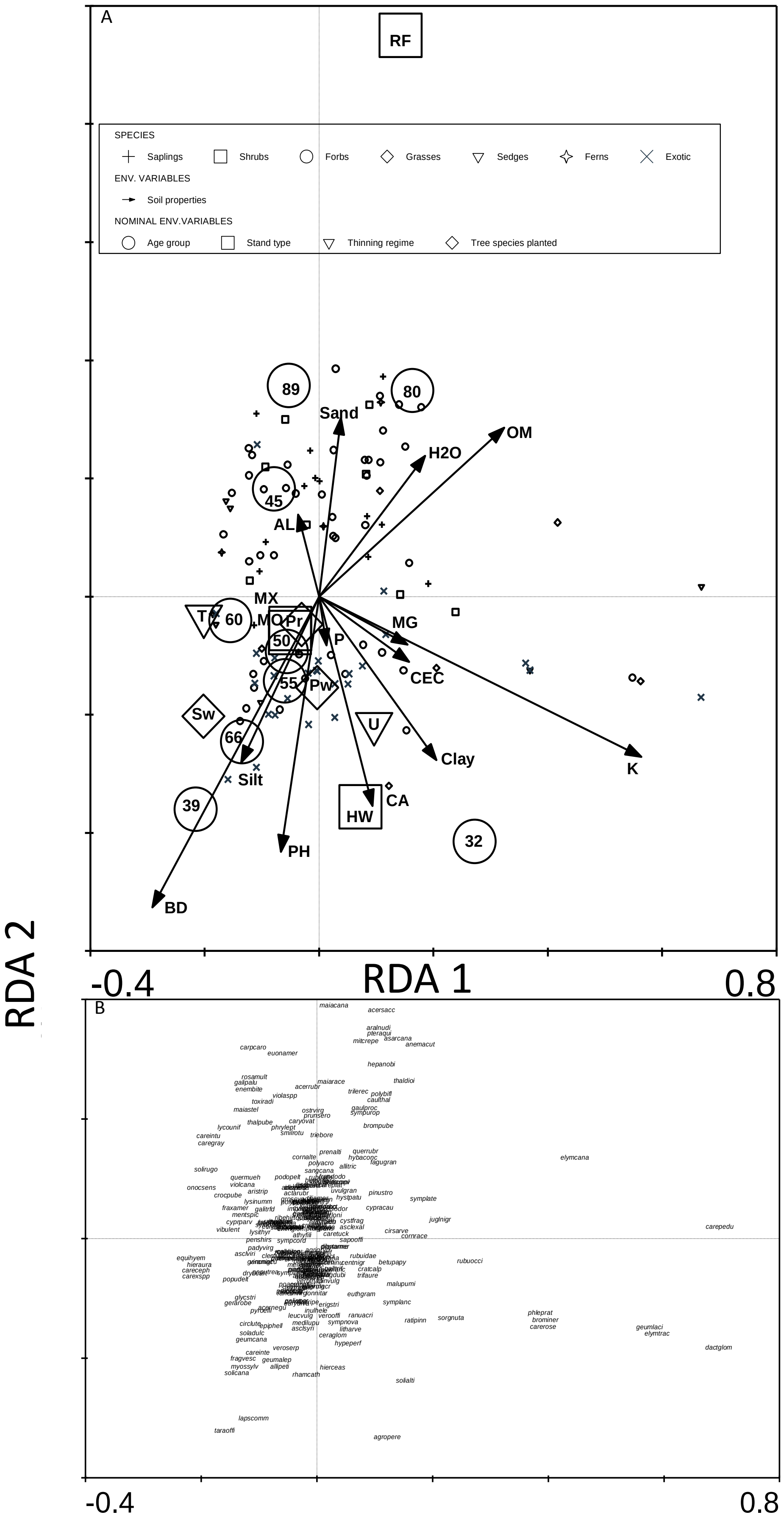


Figure 4.3. Plot-level plant species density in the understorey of plantation stands and reference natural forests. The black-shaded line indicates the average number of understorey plant species found per metre-squared sampling plot within reference mature natural forests (R); blue, pink, and orange-coloured lines indicate results from thinned softwood (T), under-thinned softwood (U) and mixed hardwood softwood (H) plantation stands, respectively. Each vertical bar indicates the 95% confidence interval symmetrically surrounding the least-squared mean value for the stand type shown. Symbols above bars indicate that the mean value for the corresponding stand type differed statistically from that for R (*=$P<0.05$; ^= $0.05<P<0.10$).

Figure 4.4. **Redundancy analysis (RDA) for understorey community composition, based on frequency of species occurrence at each site.** The analysis explained 6.5% of species data and 38% of species-environment relationships; the first two redundancy axes are plotted here (RDA 1 λ = 0.0.018, CCA 2 λ = 0.0.018). In the top panel, small symbols represent occurrence of individual species (classified according to plant life-form) while large symbols represent centroids of site classification variables (see legend inset and Fig. 5 for details). All variables shown in the plot were found to be highly significant ($P<0.005$) in Monte Carlo permutation testing. In the bottom panel, the ordination above is replicated but explanatory variables are not shown and species symbols have been replaced with the species ID codes (for cross-reference with species lists found in the Appendix). The proximity on the plot of one species label to any other, or to a site classification centroid, is proportional to the likelihood of the first species occurring alongside the second or at a site consistent with the grouping variable, respectively. Position of a species label relative to the orientation of the regression vectors corresponds to the relative quantity (i.e. low → high) of the variable that is associated with optimal occurrence of that species.

4.2 Supplementary Results (Appendix C) – Fig. 4.4



4.3 Supplemental Results (Appendix C) – Table 4.2

Table 4.2: List of all vascular plants encountered during the Afforested Environments Study

	Species	ID code	Common name	Family	lifeform	Provincial Rank	Global rank
	Acer negundo	acernegu	Manitoba maple	Sapindaceae	Tall shrub	S5	G5
x	Acer plantoides	acerplan	Norway maple	Sapindaceae	Tree	SE5	G?
	Acer rubrum	acerrubr	red maple	Sapindaceae	Tree	S5	G5
	Acer saccharinum	acersaci	silver Maple	Sapindaceae	Tree	S5	G5
	Acer saccharum	acersacc	sugar maple	Sapindaceae	Tree	S5	G5T?
	Actaea pachypoda	actapach	white baneberry	Ranunculaceae	Forb	S5	G5
	Actaea rubra	actarubr	red baneberry	Ranunculaceae	Forb	S5	G5
	Adiantum pedatum	adiepeda	maidenhair fern	Pteridaceae	Ferns	S5	G5
	Agrimonia gryposepala	agrigryp	Agrimony	Rosaceae	Forb	S5	G5
	Agrostis perennans	agropere	autumn bent grass	Poaceae	Grass	S5	G5
x	Alliaria petiolata	allipeti	garlic mustard	Brassicaceae	Forb	SE5	G5
	Allium tricoccum	allitric	wild leek	Liliaceae	Forb	S5	G5
x	Amaranthus albus	amaralbu	tumble pigweed	Amaranthaceae	Forb	SE5	G5
	Ambrosia artemisiifolia	ambrarte	common ragweed	Asteraceae	Forb	S5	G5
	Amelanchier arborea	amelarbo	downy serviceberry	Rosaceae	Tree	S5	G5
	Anemone acutiloba	anemacut	sharp-lobed liver-leaf	Ranunculaceae	Forb	S5	G5
	Anemone canadensis	anemcana	Canada anemone	Ranunculaceae	Forb	S5	G5
	Anemone quinquefolia	anemquin	wood anemone	Ranunculaceae	Forb	S5	G5
	Anemone virginiana	anemvirg	tall anemone	Ranunculaceae	Forb	S5	G5T
	Aralia nudicaulis	aralnudi	wild sarsaparilla	Araliaceae	Forb	S5	G5
x	Arctium minus	arctminu	common burdock	Asteraceae	Forb	SE5	G?T?
	Arisaema triphyllum	aristrip	small jack in the pulpit	Araceae	Forb	S5	G5T5
	Asarum canadense	asarcana	wild ginger	Aristolochiaceae	Forb	S5	G5
	Asclepias syriaca	asclsyri	common milkweed	Asclepiadaceae	Forb	S5	G5
*	Asclepias viridiflora	asclviri	green milkweed	Asclepiadaceae	Forb	S2	G5
	Asclepias exaltata	asclexal	poke milkweed	Asclepiadaceae	Forb	S4	G5
	Athyrium filix-femina	athyfili	northern lady fern	Dryopteridaceae	Ferns	S5	G5T5
	Aucuparia americana	aucuamer	American mountain-ash	Rosaceae	Tree	S5	G5
x	Berberis vulgaris	berbvulg	common barberry	Berberidaceae	Tall shrub	SE5	G?
	Betula alleghaniensis	betualle	yellow birch	Betulaceae	Tree	S5	G5
	Betula papyifera	betupapy	paper birch	Betulaceae	Tree	S5	G5
*	Blephilia hirsute	blephirs	hairy wood-mint	Lamiaceae	Forb	S1	G5?T5
	Boehmeria cylindrica	boehcyli	false nettle	Urticaceae	Forb	S5	G5
	Bossekia odorata	bossodor	purple-flowering raspberry	Rosaceae	Tall shrub	S5	G5
x	Bromus inermis	brominer	Hungarian brome	Poaceae	Grass	SE5	G4G5T
	Bromus pubescens	brompube	hairy brome	Poaceae	Grass	S4	G5Q
	Callionia simplex	callsimp	old-field cinquefoil	Rosaceae	Forb	S5	G5
	Carex arctata	carearct	drooping wood sedge	Cyperaceae	Terrestrial sedge	S5	G5
	Carex cephaloidea	careceph	head-like sedge	Cyperaceae	Terrestrial sedge	S5	G5
	Carex grayi	caregray	common bur sedge	Cyperaceae	Terrestrial sedge	S4	G4
	Carex interior	careinte	prairie star sedge	Cyperaceae	Terrestrial sedge	S5	G5
	Carex intumescens	careintu	bladder sedge	Cyperaceae	Aquatic sedge	S5	G5
	Carex pedunculata	carepedu	long-stalked sedge	Cyperaceae	Terrestrial sedge	S5	G5
	Carex pensylvanica	carepens	common oak sedge	Cyperaceae	Terrestrial sedge	S5	G5
	Carex rosea	carerose	stellate sedge	Cyperaceae	Terrestrial sedge	S5	G5

4.3 Supplemental Results (Appendix C) – Table 4.2

	Carex spp.	carexspp	carex spp.	Cyperaceae	Terrestrial sedge	S?	G?
	Carex plantaginea	careplan	plantain-leaved sedge	Cyperaceae	Terrestrial sedge	S5	G5
	Carex tuckermanii	caretuck	tuckerman's sedge	Cyperaceae	Aquatic sedge	S4	G4
	Carpinus caroliniana	carpcaro	blue beech	Betulaceae	Tree	S5	G5T
*	Carya glabra	caryglab	red hickory	Juglandaceae	Tree	S3	G5
	Carya ovate	caryovat	shagbark hickory	Juglandaceae	Tree	S5	G5
	Caulophyllum thalictroides	caulthal	blue cohosh	Berberidaceae	Forb	S5	G
x	Centaurea stoebe	centstoe	spotted knapweed	Asteraceae	Forb	SE5	G?
x	Centaurea nigra	centnigr	black knapweed	Asteraceae	Forb	SE?	G?
x	Cerastium glomeratum	ceraglom	mouse-ear chickweed	Caryophyllacaea	Forb	SE2	G?
	Chimaphila umbellata	chimubel	common pipsissewa	Pyrolaceae	Small shrub	S5	G5T5
	Cicuta virosa	cicuviro	water-hemlock	Apiaceae	Aquatic herbs	S4	G4G5
	Circaea lutetiana	circlute	Canada enchanter's nightshade	Onagraceae	Forb	S5	G5T5
x	Cirsium arvense	cirsarve	Canada thistle	Asteraceae	Forb	SE5	G?
	Clematis virginiana	clemvirg	virgin's bower	Ranunculaceae	Tall shrub	S5	G5
	Clinopodium vulgare	clinvulg	wild basil	Lamiaceae	Forb	S5	G?
	Cornus alternifolia	cornalte	alternate-leaved dogwood	Cornaceae	Tall shrub	S5	G5
	Cornus florida	cornflor	Eastern flowering dogwood	Cornaceae	Tree	S4	G5
	Cornus racemosa	cornrace	red panicled dogwood	Cornaceae	Tall shrub	S5	G5
	Cornus sericea	cornseri	red-osier dogwood	Cornaceae	Tall shrub	S5	G5
	Crataegus calpodendron	cratcalp	pear hawthorn	Rosaceae	Tall shrub	S4S5	G5
	Crataegus coccinea	cratcocc	pringle's hawthorn	Rosaceae	Tree	S5	G5
	Crataegus phaenopyrum	cratphae	Washington Hawthorn	Rosaceae	Tall shrub	-	-
	Crataegus punctat	cratpunc	dotted Hawthorn	Rosaceae	Tall shrub	-	-
	Crocion pubescens	crocpube	downy yellow violet	Violaceae	Forb	S5	G5
	Cylactis pubescebs	cylapube	dwarf raspberry	Rosaceae	Forb	S5	G5
	Cypripedium acaule	cypracau	moccasin flower	Orchidaceae	Forb	S5	G5
	Cypripedium parviflorum	cyprparv	small yellow lady's slipper	Orchidaceae	Forb	S5	G5T
	Cystopteris fragilis	cystfrag	fragile fern	Cystopteridaceae	Ferns	S5	G5
	Cystopteris bulbifera	cystbulb	bulbet bladder fern	Cystopteridaceae	Ferns	S5	G5
x	Dactylis glomerata	dactglom	orchard grass	Poaceae	Grass	SE5	G?
x	Daucus carota	dauccaro	wild carrot	Apiaceae	Forb	SE5	G?
x	Dianthus armeria	dianarme	deptford pink	Caryophyllacaea	Forb	SE5	G?
	Diervilla lonicera	dierloni	bush honeysuckle	Caprifoliaceae	Tall shrub	S5	G5
x	Dipsacus fullonum	dipsfull	wild teasel	Dipsacaceae	Forb	SE5	G?T?
	Dryopteris carthusiana	dryocart	spinulose wood fern	Dryopteridaceae	Ferns	S5	G5
	Dryopteris filic-mas	dryofili	male-fern	Dryopteridaceae	Ferns	S4	G5
	Dryopteris intermedia	dryointe	american shield fern	Dryopteridaceae	Ferns	S5	G5
	Dryopteris goldiana	dryogold	goldie's fern	Dryopteridaceae	Ferns	S4	G4
	Dryopteris marginalis	dryomarg	marginal shield fern	Dryopteridaceae	Ferns	S5	G5
x	Duchesnea indica	duchindi	Indian strawberry	Rosaceae	Forb	SE1	G5
	Echinocystis lobata	echicart	wild cucumber	Cucurbitaceae	Forb	S5	G5
	Elymus canadensis	elymcana	Canada wild rye	Poaceae	Aquatic grasses	S4S5	G5
	Elymus trachycaulus	elymtrac	slender wheat grass	Poaceae	Grass	S5?	G5T5
*	Enemion biternatum	enembite	false rue-anemone	Ranunculaceae	Forb	S2	G5
x	Epipactis helleborine	epiphell	common helleborine	Orchidaceae	Forb	SE5	G5
	Equisetum hyemale	equihyem	winter scouring rush	Equisetaceae	Horsetails	S5	G5T5
	Erigeron philadelphicus	erigphil	Philadelphia fleabane	Asteraceae	Forb	S5	G5T
	Erigeron strigosus	erigstrig	daisy fleabane	Asteraceae	Forb	S5	G5

4.3 Supplemental Results (Appendix C) – Table 4.2

	Euonymus americanus	euonamer	running strawberry bush	Celastraceae	Small shrub	S5	G5
x	Euphorbia cyparissias	euphcypa	cypress spurge	Euphorbiaceae	Forb	SE5	G5
*	Eurybia divaricate	eurydiva	white wood aster	Asteraceae	Forb	S2	G5
	Eurybia radula	euryradu	rough-leaved aster	Asteraceae	Forb	SH	G5
	Euthamia graminifolia	euthgram	grass-leaved goldenrod	Asteraceae	Forb	S5	G5
	Fagus grandifolia	fagugran	American beech	Fagaceae	Tree	S5	G5
x	Ficaria verna	ficavern	lesser celandine	Ranunculaceae	Forb	SE1	G?T?
	Fragaria vesca	fragvesc	woodland strawberry	Rosaceae	Forb	S5	G5T?
x	Frangula dodonei	frandodo	glossy buckthorn	Rhamnaceae	Tall shrub	SE5	G?
	Fraxinis americana	fraxamer	white ash	Oleaceae	Tree	S5	G5
	Fraxinus nigra	fraxnigr	black ash	Oleaceae	Tree	S5	G5
	Fraxinus pennsylvanica	fraxpenn	red ash	Oleaceae	Tree	S5	G5
x	Galium sylvaticum	galisylv	wood bedstraw	Rubiaceae	Forb	SE1	G?
	Galium trifidum	galitrfd	small bedstraw	Rubiaceae	Forb	S5	G5T?
	Galium triflorum	galitrif	sweet-scented bedstraw	Rubiaceae	Forb	S5	G5
	Galium aparine	galiapar	Cleavers	Rubiaceae	Forb	S5	G5
x	Galium mollugo	galimull	smooth bedstraw	Rubiaceae	Forb	SE5	G5
	Galium palustre	galipalu	marsh bedstraw	Rubiaceae	Forb	S5	G5
	Gaultheria procumbens	gaulproc	Wintergreen	Ericaceae	Small shrub	S5	G5
	Geranium maculatum	geramacu	spotted crane's-bill	Geraniaceae	Forb	S5	G5
x	Geranium robertianum	gerarobe	herb Robert	Geraniaceae	Forb	SE5	G5
	Geum canadense	geumcana	white avens	Rosaceae	Forb	S5	G5
	Geum aleppicum	geumalep	yellow avens	Rosaceae	Forb	S5	G5
	Geum laciniatum	geumlaci	rough avens	Rosaceae	Forb	S4	G5
	Glyceria striata	glycstri	fowl manna grass	Poaceae	Aquatic grasses	S5	G5
*	Glycyrrhiza lepidota	glyclepi	wild licorice	Fabaceae	Forb	S3	G5
	Grossularia cynosbati	groscyno	prickly gooseberry	Grossulariaceae	Tall shrub	S5	G5
	Gymnocarpium dryopteris	gymndryo	oak fern	Dryopteridaceae	Ferns	S5	G5
	Hamamelis virginiana	hamavirg	witch hazel	Hamamelidaceae	Tall shrub	S5	G5
	Hepatica nobilis	hepanobi	round-leaved hepatica	Ranunculaceae	Forb	S5	G?
x	Hesperis matronalis	hespmatr	dame's rocket	Brassicaceae	Forb	SE5	G4G5
x	Hieracium aurantiacum	hieraura	orange hawkweed	Asteraceae	Forb	SE5	G?
x	Hieracium caespitosum	hiercaes	field hawkweed	Asteraceae	Forb	SE5	G5
*	Hybanthus concolor	hybaconc	green violet	Violaceae	Forb	S2	G5
x	Hypericum perforatum	hypeperf	common St. John's wort	Hypericaceae	Forb	SE5	G?
	Hystrix patula	hystpatu	bottle-brush grass	Poaceae	Grass	S5	G5
	Impatiens capensis	impacape	spotted touch-me-not	Balsaminaceae	Forb	S5	G5
x	Inula helenium	inulhele	Elecampane	Asteraceae	Forb	SE5	G?
	Iris versicolor	irisvers	multi-coloured blue-flag	Iridaceae	Forb	S5	G5
	Juglans nigra	juglnigr	black walnut	Juglandaceae	Tree	S4	G5
	Lactuca biennis	lactbien	tall blue lettuce	Asteraceae	Forb	S5	G5
	Lactuca canadensis	lactcana	Canada lettuce	Asteraceae	Forb	S5	G5
x	Lapsana communis	lapscomm	Nipplewort	Asteraceae	Forb	SE5	G?
x	Leucanthemum vulgare	leucvulg	ox-eye daisy	Asteraceae	Forb	SE5	G?
	Lindera benzoin	lindbenz	Spicebush	Lauraceae	Tall shrub	S5	G5
	Liriodendron tulipifera	lirituli	tulip tree	Magnoliaceae	Tree	S4	G5
x	Lithospermum arvense	litharve	common gromwell	Boraginaceae	Forb	SE5	G?
	Lonicera caerulea	lonicaer	mountain fly-honeysuckle	Caprifoliaceae	Tall shrub	S5	G5
	Lonicera canadensis	lonicana	American fly-honeysuckle	Caprifoliaceae	Tall shrub	S5	G5

4.3 Supplemental Results (Appendix C) – Table 4.2

x	Lonicera tatarica	lonitart	tartarian honeysuckle	Caprifoliaceae	Tall shrub	SE5	G?
	Lycopus uniflorus	lycounif	Tuberous water-horehound	Lamiaceae	Aquatic herbs	S5	G5
	Lycopus americanus	lycoamer	American water-horehound	Lamiaceae	Aquatic herbs	S5	G5
x	Lysimachia nummularia	lysinumm	Moneywort	Primulaceae	Forb	SE5	G?
	Lysmachia thyrsiflora	lysithyr	Yellow loosestrife	Primulaceae	Forb	S5	G5
	Maianthemum canadense	maiacana	Wild lily-of-the-valley	Liliaceae	Forb	S5	G5
	Maianthemum racemosum	maiarace	False Solomon's seal	Liliaceae	Forb	S5	G5T
	Maianthemum stellatum	maiastel	Star-flowered Solomon's seal	Liliaceae	Forb	S5	G5
	Malus spp.	maluspp.	Crabapples	Rosaceae	Tree	-	-
	Malus pumila	malupumi	Common crabapple	Rosaceae	Tree	SE5	G5
	Matteuccia struthiopteris	mattstru	Ostrich fern	Dryopteridaceae	Ferns	S5	G5
x	Medicago lupulina	medilupu	Black medic	Fabaceae	Forb	SE5	G?
x	Melilotus albus	melialbu	White sweet-clover	Fabaceae	Forb	SE5	G?
x	Mentha spicata	mentspic	Spear mint	Lamiaceae	Forb	SE4	G?
	Mespilus coccinea	mespcocc	Scarlet hawthorn	Rosaceae	Tall shrub	S4	G5
	Mitchella repens	mitcrepe	Creeping partridge-berry	Rubiaceae	Small shrub	S5	G5
	Mitella nuda	mitenuda	Naked bishop's cap	Saxifragaceae	Forb	S5	G5
	Monarda fistulosa	monafist	Wild bergamot	Lamiaceae	Forb	S5	G5
	Monotropa uniflora	monounif	Indian-pipe	Monotropaceae	Forb	S5	G5
x	Myosotis sylvatica	myossylv	Woodland forget-me-not	Boraginaceae	Forb	SE4	G5
	Neottia canvallarioides	neotconv	Broad-lipped twayblade	Orchidaceae	Forb	S$	G5
	Onoclea sensibilis	onocsens	sensitive fern	Onocleaceae	Ferns	S5	G5
	Ostrya virginiana	ostrvirg	Ironwood	Betulaceae	Tree	S5	G5
	Oxalis acetosella	oxalacet	true wood-sorrel	Oxalidaceae	Forb	S5	G5
	Padys virginiana	paduvirg	choke cherry	Rosaceae	Tall shrub	S5	G5T
	Parthenocissus quinquefolia	partquin	Virginia creeper	Vitaceae	Tall shrub	S4?	S5
	Penstemon hirutus	penshirs	hairy beard-tongue	Scrophulariaceae	Forb	S4	G4
x	Phleum pretense	phleprat	common Timothy	Poaceae	Grass	SE5	G?
	Phlox divaricate	phlodivi	wild blue phlox	Polemoniaceae	Forb	S4	G5
	Phryma leptostachya	phrylept	slender-spiked lopseed	Phrymaceae	Forb	S4	G5
	Phytolacca americana	phytamer	Pokeweed	Phytolaccaceae	Forb	S4	G5
x	Picea abies	piceabie	Norway spruce	Pinaceae	Tree	SE3	-
	Picea glauca	piceglau	white spruce	Pinaceae	Tree	S5	G5
	Pinus resinosa	pinuresi	red pine	Pinaceae	Tree	S5	G5
	Pinus strobus	pinustrob	eastern white pine	Pinaceae	Tree	S5	G5
x	Pityopsis falcate	pityfalc	golden aster	Asteraceae	Forb	SE1	G3G4
x	Plantago major	planmajo	common plantain	Plantaginaceae	Forb	SE5	G5
x	Plantago lancelota	planlanc	narrow-leaved plantain	Plantaginaceae	Forb	SE5	G5
	Poa compressa	poacompr	Canada blue grass	Poaceae	Grass	S5	G?
	Podophyllum peltatum	podopelt	may-apple	Berberidaceae	Forb	S5	G5
	Polygonatum biflorum	polybifl	smooth Solomon's seal	Liliaceae	Forb	S4	G5
	Polygonatum puescens	polypube	hairy Solomon's seal	Liliaceae	Forb	S5	G5
	Polystichum acrostichoides	polyacro	Christmas fern	Dryopteridaceae	Ferns	S5	G5
	Populus deltoids	popudelt	eastern cottonwood	Salicaceae	Tree	SU	G5T?
	Populus tremuloides	poputrem	trembling aspen	Salicaceae	Tree	S5	G5
	Populus balsamifera	popubals	balsam poplar	Salicaceae	Tree	S5	G5T?
	Populus tremula	poputrea	large-tooth aspen	Salicaceae	Tree	S5	G5
x	Potentilla recta	poterect	sulphur cinquefoil	Rosaceae	Forb	SE5	G?
	Prenanthes altissima	prenalti	tall white rattlesnake-root	Asteraceae	Forb	S5	G5?

4.3 Supplemental Results (Appendix C) – Table 4.2

	Prunella vulgaris	prunvulg	self-heal	Lamiaceae	Forb	S5	G5T?
	Prunus pensylvanica	prunpens	pin cherry	Rosaceae	Tree	S5	G5
	Prunus serotina	prunsero	black cherry	Rosaceae	Tree	S5	G5
	Pteridium aquilinum	pteraqui	eastern bracken fern	Dennstaedtiaceae	Ferns	S5	G5T
	Pyrola elliptica	pyroelli	Shinleaf	Pyrolaceae	Forb	S5	G5
	Pyrola asarifolia	pyroasar	round-leaved pyrola	Pyrolaceae	Forb	S4?	G5
	Quercus alba	queralba	white oak	Fagaceae	Tree	S5	G5
	Quercus muehlenbergii	quermueh	chinquapin oak	Fagaceae	Tree	S4	G5
	Quercus macrocarpa	quermacr	bur oak	Fagaceae	Tree	S5	G5
	Quercus rubrum	querrubr	red oak	Fagaceae	Tree	S5	G5
	Quercus velutina	quervelu	black oak	Fagaceae	Tree	S4	G5
x	Ranunculus acris	ranuacri	tall buttercup	Ranunculaceae	Forb	SE5	G5
	Ranunculus pensylvanicus	ranupenn	bristly buttercup	Ranunculaceae	Forb	S5	G5
x	Raphanus raphanistrum	raphraph	wild radish	Brassicaceae	Forb	SE3	G?
*	Ratibida pinnata	ratipinn	gray-headed coneflower	Asteraceae	Forb	S3	G5
x	Rhamnus cathartica	rhamcath	European buckthorn	Rhamnaceae	Tall shrub	SE5	G?
	Rhus typhina	rhustyph	staghorn sumac	Anacardiaceae	Tall shrub	S5	G5
	Ribes americanum	ribeamer	American black currant	Grossulariaceae	Tall shrub	S5	G5
	Ribes hirtellum	ribehirt	smooth gooseberry	Grossulariaceae	Tall shrub	S5	G5
	Rorippa palustris	roripalu	marsh yellow cress	Capparales	Aquatic herbs	S5	G5T?
	Rosa blanda	rosablan	smooth rose	Rosaceae	Tall shrub	S5	G5
x	Rosa multiflora	rosamult	multiflora rose	Rosaceae	Tall shrub	SE4	G?
	Rubus idaeus	rubuidae	red raspberry	Rosaceae	Tall shrub	S5	G5T
	Rubus allegheniensis	rubuocci	high-bush blackberry	Rosaceae	Tall shrub	S5	G5
	Rubus occidentalis	rubualle	black raspberry	Rosaceae	Tall shrub	S5	G5
	Rudbeckia hurta	rudbhirt	black-eyed Susan	Asteraceae	Forb	S5	G5
	Salix interior	saliinte	sandbar willow	Salicaceae	Tall shrub	S5	G5
	Sambucus canadensis	sambcana	Canada elderberry	Caprifoliaceae	Tall shrub	S5	G5
	Sanguinaria canadensis	sangcana	Bloodroot	Papaveraceae	Forb	S5	G5
x	Saponaria officinalis	sapooffi	bouncing-bet	Caryophyllacaea	Forb	SE5	G?
	Sassafras albidum	sassalbi	Sassafras	Lauraceae	Tree	S4	G5
x	Silene vulgaris	silevulg	bladder campion	Caryophyllacaea	Forb	SE5	G?
	Smilax tamnoides	smiltamn	bristly greenbrier	Smilacaceae	Forb	S4	G5Q
*	Smilax rotundifolia	smilrotu	common greenbrier	Smilacaceae	Forb	S2	G5
x	Solanum dulcamara	soladulc	bitter nightshade	Solanaceae	Forb	SE5	G?
	Solidago juncea	solijunc	sharp-toothed goldenrod	Asteraceae	Forb	S5	G5
	Solidago rugosa	solirugo	rough-stemmed goldenrod	Asteraceae	Forb	S5	G5T?
	Solidago altissima	solialti	tall goldenrod	Asteraceae	Forb	S5	G?
	Solidago canadensis	solicana	Canada goldenrod	Asteraceae	Forb	S5	G5
	Solidago flexicaulis	soliflex	zig-zag goldenrod	Asteraceae	Forb	S5	G5
	Sorghastrum nutans	sorgnuta	Indian grass	Poaceae	Grass	S4	G5
	Spiraea alba	spiralba	narrow-leaved meadow-sweet	Rosaceae	Tall shrub	S5	G5
x	Stellaria graminea	stelgram	common chickweed	Caryophyllacaea	Forb	SE5	G?
	Streptopus amplexifolius	streampl	clasping-leaved twisted-stalk	Liliaceae	Forb	S4S5	G5
*	Strophostyles helvola	strohelv	trailing wild bean	Fabaceae	Forb	S3	G5
	Symphyotrichum cordifolium	sympcord	heart-leaved aster	Asteraceae	Forb	S5	G5
	Symphyotrichum laeve	symplaev	glaucous aster	Asteraceae	Forb	S5	G5T?
	Symphyotrichum lanceolatum	symplate	panicled aster	Asteraceae	Forb	S5	G5T
	Symphyotrichum lateriflorum	sympnovb	calico aster	Asteraceae	Forb	S5	G5T5

4.3 Supplemental Results (Appendix C) – Table 4.2

	Symphyotrichum novi-belgii	sympundu	new York aster	Asteraceae	Forb	S5	G5
	Symphyotrichum undulatum	symplanc	wavy-leaved aster	Asteraceae	Forb	SU	G4
	Symphyotrichum novae-angliae	sympnova	New England aster	Asteraceae	Forb	S5	G5
	Symphyotrichum pilosum	symppilo	white heath aster	Asteraceae	Forb	S5	G5T?
	Symphyotrichum uriohyllum	sympurop	Arrow-leaved aster	Asteraceae	Forb	S4	G4
x	Taraxacum officinale	taraoffi	Dandelion	Asteraceae	Forb	SE5	G5
	Thalictrum dioicum	thaldioi	early meadow rue	Ranunculaceae	Forb	S5	G5
	Thalictrum pubescens	thalpube	tall meadow rue	Ranunculaceae	Forb	S5	G5
	Thuja occidentalis	thujocci	eastern white cedar	Cupressaceae	Tree	S5	G5
	Tilia Americana	tiliamer	American basswood	Tiliaceae	Tree	S5	G5
	Toxicodendron radicans	toxiradi	climbing poison ivy	Anacardiaceae	Tall shrub	S5	G5T
x	Tragopogon dubius	tragdubi	goat's-beard	Asteraceae	Forb	SE5	G?
	Trientalis borealis	triebore	star-flower	Primulaceae	Forb	S5	G5?
x	Trifolium aurem	trifaure	yellow clover	Fabaceae	Forb	SE5	G?
	Trillium erectum	trilerec	purple trillium	Liliaceae	Forb	S5	G5
	Trillium grandiflorium	trilgran	white trillium	Liliaceae	Forb	S5	G5
	Triosteum aurantiacum	triogran	wild coffee	Caprifoliaceae	Forb	S5	G5
	Tsuga Canadensis	tsugcana	eastern hemlock	Pinaceae	Tree	S5	G5
	Ulmus Americana	ulmuamer	White/American Elm	Ulmaceae	Tree	S5	G5
	Uvularia grandiflora	uvulgran	large-flowered bellwort	Liliaceae	Forb	S5	G5
	Vaccinium angustifolium	vaccangu	velvet-leaf blueberrt	Ericaceae	Small shrub	S5	G5
x	Verbascum thapsus	verbthap	common mullein	Scrophulariaceae	Forb	SE5	G?
x	Veronica offininalis	verooffi	common speedwell	Scrophulariaceae	Forb	SE5	G5
x	Veronica serpyllifolia	veroserp	thyme-leaved speedwell	Scrophulariaceae	Forb	SE5	G?T?
	Viburnum recognitum	vibureco	southern arrowwood	Caprifoliaceae	Tall shrub	S4	G5
	Viburnum acerfolium	vibuacer	maple-leaved viburnum	Caprifoliaceae	Tall shrub	S5	G5
	Viburnum lentago	vibulent	Nannyberry	Caprifoliaceae	Tall shrub	S5	G5
	Viburnum opulus	vibuopul	high bush cranberry	Caprifoliaceae	Tall shrub	S5	G5T5
x	Vicia cracca	vicccrac	cow vetch	Fabaceae	Forb	SE5	G?
x	Vincetoxicum nigrum	vincnigr	black swallow-wort	Asclepiadaceae	Forb	SE?	G?
	Viola Canadensis	violaspp	Canada violet	Violaceae	Forb	S5	G5
	Viola sp.	violcana	violet species	Violaceae	Forb		
	Vitis riperia	vitiripe	riverbank grape	Vitaceae	Small shrub	S5	G5
x	Zea mays	zeamays	Indian corn	Poaceae	Grass	SE2	G?

*Notes: All species information derived from Newmaster, S.G. and S. Ragupathy, 2012. Flora Ontario – Integrated Botanical Information System (FOIBIS), Phase I. University of Guelph, Canada. Available at: http://www.uoguelph.ca/foibis); * indicates species is at risk or imperilled in Ontario (i.e. provincial rank = S1-S3); x indicates species not native to Ontario*

Figure 4.5. Microhabitat and soil features of plantation stands and reference natural forests found to be independent of time since afforestation. For each microhabitat feature shown, black-shaded lines indicate average values observed within reference mature natural forests (R) while blue, pink, and orange-coloured lines indicate results from thinned softwood (T), under-thinned softwood (U) and mixed hardwood softwood (H) plantation stands, respectively. Vertical bars indicate the 95% confidence intervals (symmetrically) surrounding the least-squared mean value for each stand type. Symbols above bars indicate that the mean value for the corresponding stand type differed statistically from that for R (*=P<0.05; ^= 0.05<P<0.10). Separate general linear models for each of the seven variables (panels A-B and D-G) exhibited respective F(4,37) values of: 8.62, 244.13, 15.70, 90.06, 65.70, 33.43, and 61.11, with P<0.0001 for all. The respective R^2 values for the models were: 0.12, 0.07, 0.09, 0.16, 0.19, 0.16, 0.18. Dunnett-adjusted comparisons of T, S and H to R for each variable yielded P values which were respectively greater than 0.4026, 0.2316, 0.2161, 0.3875, 0.260, 0.3787, and 0.2138 (i.e. plantations were not different from R with respect to the variables shown).

4.3 Supplemental Results (Appendix C) – Fig. 4.5

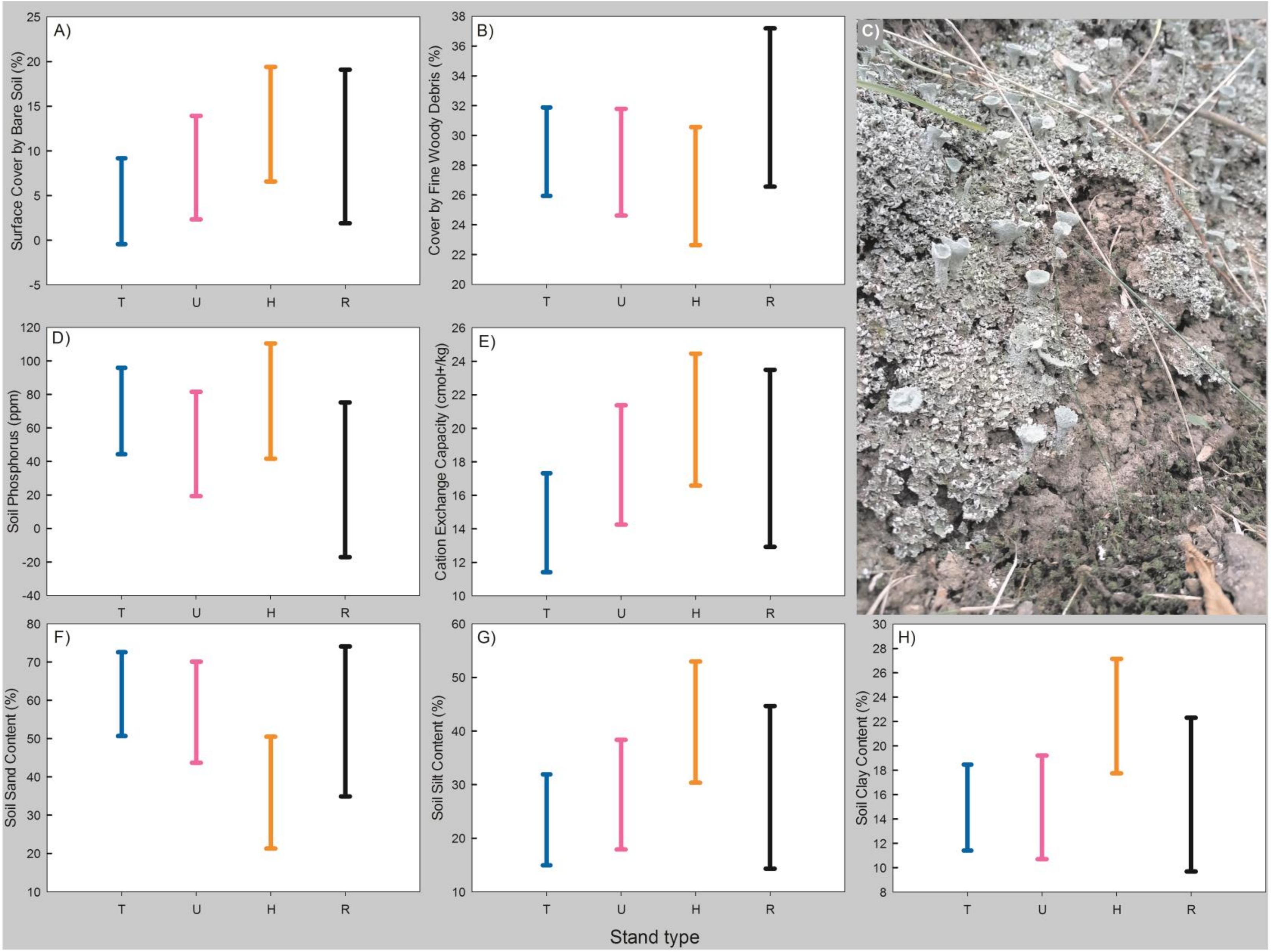


Figure 4.6. Microhabitat and soil features of plantation stands found to be dependent upon time since afforestation. Panel A) shows duff-layer microhabitat conditions typically experienced by *Asarum* (shown) in its natural hardwood understorey environment. Panels B-F) show measured duff-layer and soil features found to vary significantly with time since afforestation in at least one type of plantation stand, as well as average conditions in natural reference forests (R, black lines). Blue, pink, and orange-coloured lines indicate results from thinned softwood (T), under-thinned softwood (U) and mixed hardwood softwood (H) plantation stands, respectively. Solid regression lines indicate a statistically significant relationship between the level of the feature and time since afforestation while the dashed lines indicate no significant relationship. For each such feature, points along the x-axis where significant regression lines intersect the reference line (or would intersect, if extrapolated) correspond to expected time-lags between stand planting and equivalence to reference forests, within the indicated plantation type.

4.3 Supplemental Results (Appendix C) – Fig. 4.6

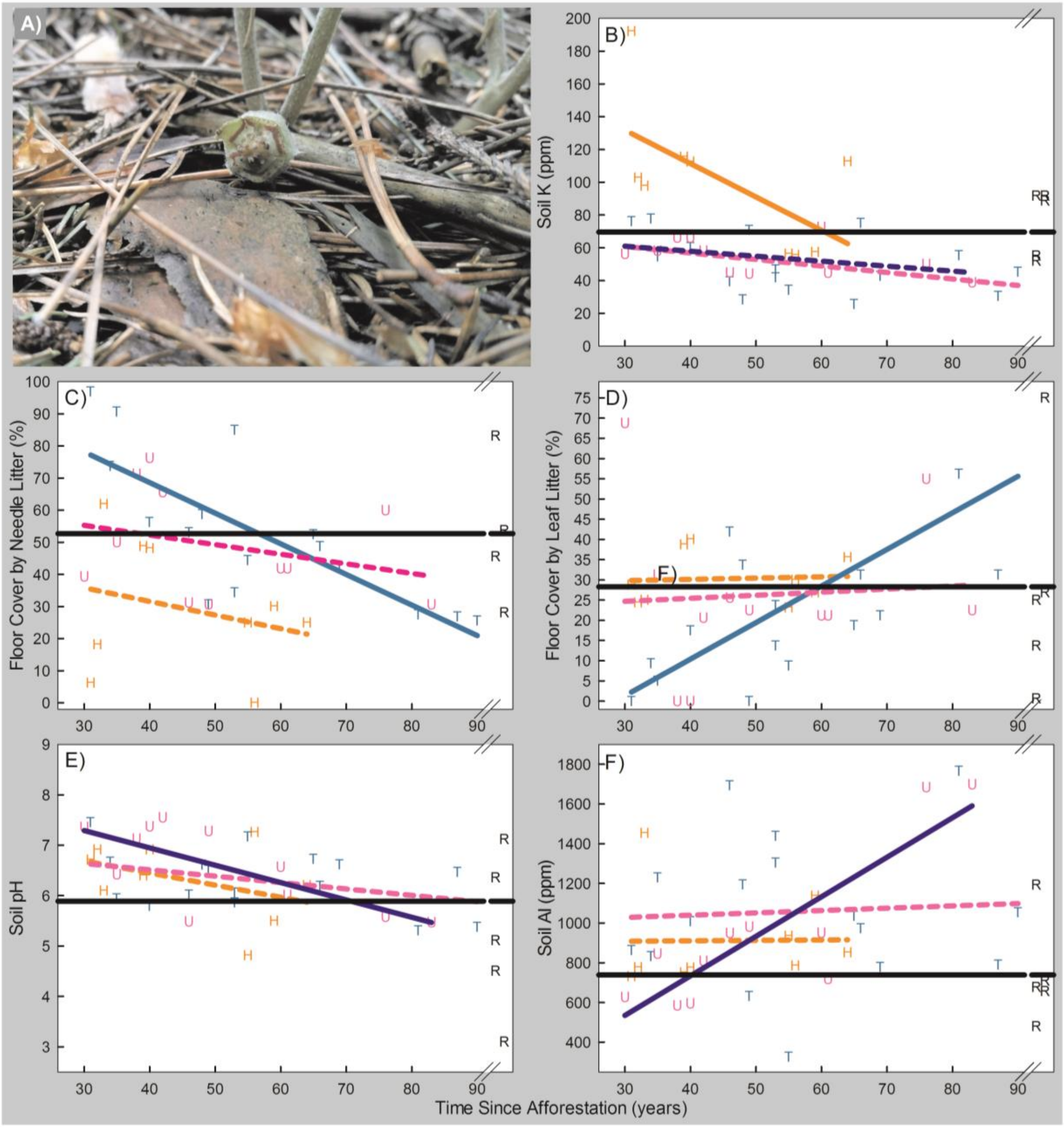


Table 4.3 Generalized Linear Model of *Allium tricoccum* occurrence in herb-relocation plots across stand composition types and sampling years

Analysis Of GEE Parameter Estimates*						
Factor	**Level**		**Est.**	**Std.Err.**	**Z**	**Pr > |Z|**
Intercept	Type	Year	-0.2	0.2	-1.47	0.1426
Stand type	R	.	0.0	0.0	.	.
	M	.	-0.7	0.2	-3.34	0.0008
	X	.	-0.8	0.2	-4.26	<0.0001
	H	.	-1.4	0.2	-5.99	<0.0001
Sampling year	.	1	0.0	0.2	-0.13	0.8964
	.	3	0.0	0.0	.	.
Stand type X Sampling year	R	1	0.0	0.0	.	.
	R	3	0.0	0.0	.	.
	M	1	0.0	0.3	-0.06	0.9561
	M	3	0.0	0.0	.	.
	X	1	0.1	0.2	0.57	0.5690
	X	3	0.0	0.0	.	.
	H	1	0.5	0.3	1.66	0.0974
	H	3	0.0	0.0	.	.
Scale	.	.	1.0	.	.	.

*Model-baseds standard errors are presented; results reflect analysis of 1230 herb-relocation plots sampled in 2012 and again in 2014, with 679 events of *Allium* presence observed across the 2460 trials.

Table 4.4. Generalized Linear Model of *Allium tricoccum* occurrence in herb-relocation plots within hardwood-softwood (H) and softwood-only (S) plantation stands varying in time since afforestation

Analysis Of GEE Parameter Estimates*					
Factor	**Level**	**Est.**	**Std.Err.**	**Z**	**Pr > |Z|**
Intercept	.	-2.6	0.2	-12.63	<0.0001
Stand type	H	1.9	0.5	3.84	0.0001
	S	0	0	.	.
Time X Stand type	H	-0.01	0.01	-1.46	0.1433
	S	0.030	0.004	8.56	<0.0001
Scale	.	1	.	.	.

*Model-baseds standard errors are presented; results reflect analysis of *Allium* occurrence within 1080 herb-relocation plots sampled in S and H plantations in 2012 and again in 2014. Factor Time refers to Time Since Afforestation (i.e. plantation age). In total there were 548 events of wild leek occurrence across the 2160 trials. Model R^{2*}=0.05.

4.3 Supplemental Results (Appendix C) – Table 4.5

Table 4.5. Generalized Linear Model of *Allium tricoccum* occurrence in herb-relocation plots within H, T, and U plantation stands varying in time since afforestation

Analysis Of GEE Parameter Estimates*

Factor	Type	Year	Est.	Std.Err.	Z	Pr > |Z|
Stand type X Sampling year	H	1	-1.6	0.6	-2.87	0.0041
	H	3	0.4	0.6	0.59	0.5531
	T	1	-2.5	0.4	-7.23	< 0.0001
	T	3	-1.9	0.3	-5.87	< 0.0001
	U	1	-1.8	0.4	-4.06	< 0.0001
	U	3	-4.8	0.6	-7.95	< 0.0001
Stand type X Sampling year X Time since afforestation	H	1	0.01	0.01	0.78	0.4346
	H	3	-0.05	0.02	-3.06	0.0022
	T	1	0.031	0.006	5.45	< 0.0001
	T	3	0.023	0.005	4.23	< 0.0001
	U	1	0.009	0.008	1.16	0.2444
	U	3	0.056	0.010	5.78	< 0.0001
Scale		.	1	.	.	.

*Model-baseds standard errors are presented; results reflect analysis of *Allium* occurrence within 1080 herb-relocation plots sampled in hardwood-softwood mixed (H), thinned softwood (T), and underthinned softwood (U) plantations in 2012 and again in 2014. Time Since Afforestation (i.e. plantation age) ranged from 30-90 years. In total there were 548 events of wild leek occurrence across the 2160 trials. Model R^{2*}=0.07.

4.3 Supplemental Results (Appendix C) – Table 4.6

Table 4.6. Generalized Linear Model of *Asarum canadense* occurrence in plantation (P) and reference (R) herb-relocation plots over 3 sampling years

Analysis Of GEE Parameter Estimates*						
Factor	**Level**		**Est.**	**Std.Err.**	**Z**	**Pr > |Z|**
Intercept	**Type**	**Year**	-1.8	0.2	-7.71	< 0.0001
Stand type X Sampling year	P	1	0.0	0.3	-0.1	0.9213
	P	2	-2.0	0.3	-6.29	< 0.0001
	P	3	-2.4	0.4	-6.98	< 0.0001
	R	1	1.2	0.3	4.59	< 0.0001
	R	2	0.4	0.2	1.74	0.0824
	R	3	0.0	0	.	.
Scale			1	.	.	.

*Model-baseds standard errors are presented; results reflect analysis of 1230 herb-relocation plots sampled in 2012, 2013, and 2014 (Years 1-3), with 292 events of *Asarum occurrence* across the 3690 trials. Model R2*=0.10

4.3 Supplemental Results (Appendix C) – Table 4.7

Table 4.7. Generalized Linear Model of *Asarum canadense* occurrence in herb-relocation plots within hardwood-softwood (H) and softwood-only (S) plantation stands varying in time since afforestation

Analysis Of GEE Parameter Estimates*					
Factor	**Type**	**Est.**	**Std.Err.**	**Z**	**Pr > |Z|**
Intercept	.	-4.1	0.3	-14.19	<0.0001
Stand type X Time	H	0.010	0.008	1.29	0.1975
	S	0.026	0.005	5.80	<0.0001
Scale	.	1	.	.	.

*Model-baseds standard errors are presented; results reflect analysis of *Asarum* occurrence within 1080 herb-relocation plots sampled in H and S plantations yearly from 2012-2014. Factor Time refers to Time Since Afforestation (i.e. plantation age). In total there were 187 *Asarum* occurrence events across the 3240 trials. Model R^{2*}=0.02.

Table 4.8. Generalized Linear Model of *Asarum canadense* occurrence in herb-relocation plots within plantations varying in time since afforestation and experiencing different planted composition and thinning management

Analysis Of GEE Parameter Estimates*

Factor	**Planted**	**Thinning**	**Est.**	**Std.Err.**	**Z**	**Pr > |Z|**
Intercept	.	.	-4.3	0.3	-13.75	< 0.0001
Planted composition X Thinning X Time since afforestation	H	U	0.011	0.011	1.07	0.2826
	H	T	0.015	0.009	1.70	0.0883
	M	U	0.033	0.005	6.00	< 0.0001
	M	T	0.021	0.005	4.31	< 0.0001
	X	U	0.026	0.007	3.93	< 0.0001
	X	T	0.039	0.005	7.25	< 0.0001
Scale	.	.	1	.	.	.

*Model-baseds standard errors are presented; results reflect analysis of *Asarum* occurrence within 1080 herb-relocation plots sampled in hardwood-softwood mixed (H), softwood-only monoculture (M), and softwood-only mixed (X) plantations experiencing either regular stand thinning (T) or relative under-thinning (U). Results correspond to average patterns observed over the three sampling years (2012-2014). There were 187 *Asarum* occurrence events across the 3240 trials. Model R^{2*}=0.03.

Table 4.9 Generalized Linear Model of *Asarum canadense* occurrence in herb-relocation plots as a function of stand type and soil addition treatment

Analysis Of GEE Parameter Estimates*						
Factor	**Levels**		**Est.**	**Std. Err.**	**Z**	**Pr > |Z|**
	Soil	**Type**				
Intercept			-1.3	0.2	-5.8	< 0.0001
Soil addition	C		-0.3	0.3	-0.90	0.3697
	N		0.5	0.3	1.61	0.1069
	S		0.0	0.0	.	.
Stand type		H	-2.3	0.5	-4.92	< 0.0001
		M	-2.0	0.4	-5.21	< 0.0001
		X	-1.2	0.3	-4.04	< 0.0001
		R	0.0	0.0	.	.
Soil addition X Stand type	C	H	-1.0	1.0	-1.09	0.2766
	C	M	0.6	0.5	1.20	0.2292
	C	X	0.0	0.4	0.03	0.9731
	C	R	0.0	0.0	.	.
	N	H	-0.2	0.6	-0.33	0.7407
	N	M	0.5	0.5	1.09	0.2764
	N	X	-0.2	0.4	-0.60	0.5477
	N	R	0.0	0.0	.	.
	S	H	0.0	0.0	.	.
	S	M	0.0	0.0	.	.
	S	X	0.0	0.0	.	.
	S	R	0.0	0.0	.	.
Scale			1	.	.	.

*Model-based standard errors are presented; results reflect analysis of *Asarum* occurrence within 1230 herb-relocation plots receiving either no soil (C, control), not-sterilized soil from home reference forests (N), or sterilized soil from home reference forests (S). Stand were either planted with a hardwood-softwood mixture (H), a softwood-only mixture (X) or a softwood monoculture (M), or a naturally-occurring mature hardwood reference forest (R). There were 292 *Asarum* occurrence events across the 3690 trials. Model R2*=0.06.

Appendix D: Literature Cited

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