Seven-Year Survival of Perennial Herbaceous Transplants in Temperate Woodland Restoration

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Abstract

Little is known about restoring the perennial herbaceous understory of Midwestern deciduous woodlands, despite the significant and widespread degradation of remnants due to human activities. Because many woodland understory species have reproductive characters that make re-establishment from seed slow or difficult, we investigated transplanting as a strategy for introducing 24 species to a degraded early-successional woodland in central Iowa, U.S.A. Plants were planted in single-species groups of generally four individuals, and then monitored for survival five times over a 7-year period, and for flowering during the first year. After 7 years, persistence of these groups was 57% averaged across species. Survival in years 5–7 does not reflect individuals that spread beyond the original planting units by self-sowing or vegetative spread and is therefore a minimum estimate of the abundance of many species at the site. Mean percent flowering was 72% across single-species groups for 15 species monitored. We consider these survival and flowering rates acceptable indicators of establishment success, especially given drought conditions at our site in the first few years and lack of weed control beyond the first year, and evidence that transplanted species were establishing outside the original planting locations. Additional work is needed to investigate regional differences in transplant success, and methods for sustainable production of species are not suitable for introduction by seed. We caution that our results do not necessarily apply to the restoration of rare species.

Key words: deciduous hardwood forests, perennial herbaceous species, reintroduction, transplantation, woodland restoration.

Introduction

Following European settlement, much of the North American hardwood forest east of the Missouri River was cleared for timber products and converted to agricultural use. Although there has been widespread reforestation in the eastern and southeastern states, this has not been the case in the more agricultural Midwest. For example, forested area in Iowa declined from around 2.7 million ha prior to European settlement to a low of 0.65 million ha in 1974. Although forest area had increased somewhat to 0.85 million ha by 1990, reforestation is not widespread (Jungst et al. 1998). Similarly, in Illinois, forested area declined from 38% in 1820 to 7% in 1980, with deciduous forests restricted to steep slopes and/or infertile land (Iverson 1988).

The forested remnants throughout the region are increasingly fragmented and surrounded by agricultural and urban areas (Potts et al. 2004), increasing the likelihood that ground layer species will be extirpated. For example, fragments may have higher light levels, wind speeds, and heat loads at the edges, resulting in loss of interior species that require moist closed canopy sites (Saunders et al. 1991; Jules & Rathcke 1999 and references therein). Cattle grazing has also been widespread in the region, with up to 90% of forests grazed at one time and with 20–65% of farm woodlands still being grazed as late as 1982 (Whitney 1994). Moderate cattle grazing, where a closed canopy is maintained, shifts the understory composition toward weedy and invasive species while tending to reduce populations of forest species that do not tolerate disturbance and that are associated with more specialized habitats, particularly moist soils (Pettit et al. 1995; Mabry 2002). Where cattle grazing has been intensive enough to open the canopy, the understory is even more dramatically simplified or converted entirely to non-native cool season grasses that may make sites unsuitable for natural recolonization after grazing has ceased (DenUyl et al. 1938; Cross 1981; Whitney 1994). Although time series data on vegetation recovery following grazing are not available, data from central Iowa do suggest that sites not grazed for 15 years are similar to those currently and more recently grazed (Mabry 2002).

Additionally, throughout the Midwest, human activities and human-caused landscape changes are intensifying (Potts et al. 2004), changes that have also been widely associated with loss of native species. These include invasion by exotic species, historically high deer populations, timber harvesting, and trails associated with recreation, and urbanization (Hoehne 1981; Sharpe et al. 1986; Robinson et al. 1994; Drayton & Primack 1996; McGuinness

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& deCalesta 1996; Bratton & Meier 1998; McLachlan & Bazely 2001; Carlson & Gorchov 2004; Rooney et al. 2004).

In sum, many remnant and disturbed woodlands throughout the Midwest are unlikely to have a full complement of native understory species. Furthermore, many woodland species have limited dispersal potential, which is especially exacerbated in highly fragmented landscapes (Peterken & Game 1984; Dzwonko & Loster 1992; Matlack 1994; Vellend 2003; Mabry 2004). Thus, we cannot rely on succession or natural dispersal to restore many species to degraded sites. This combination of past and present disturbance and dispersal limitation suggests that an active restoration program is needed.

There are several challenges for developing a restoration program for woodland herbaceous species. First, there are few published reports on techniques to use when restoration strategies such as canopy thinning and prescribed fire are unsuccessful or insufficient for restoring diversity (Mottl 2000; Lane & Raab 2002). Most woodland perennial herb restoration literature originates from the United Kingdom, where experiments involve different species and different land use and land management systems (Buckley & Knight 1989; Packham & Cohn 1990; Francis et al. 1992; Street & Mond 1992; Cohn & Packham 1993), and it is unclear the degree to which the restoration practices are transferable. Information on restoring Midwestern forests is included in Packard and Mutel (1997), Sauer (1998) and Thompson (1992), but these do not evaluate specific reintroduction techniques for herbaceous species.

Second, life history characters of many woodland species make restoration by seed, a common strategy for prairie and wetland species, expensive, slow, and not feasible because (1) many woodland species produce low numbers of seeds, particularly less common species (Mabry 2004); (2) seeds often have exacting germination requirements and low viability (Cullina 2000); and (3) some species may take many years to reach reproductive maturity (Bierzychudek 1982). Many regional native seed nurseries do not yet offer shade-tolerant seed mixes, and, although some native nurseries offer savanna mixes, these have limited potential for woodland restoration because they tend to favor sun-loving prairie species over those that thrive under mixed sun and shade to closed canopy habitats.

Introducing woodland perennials as transplants is an option that may overcome these barriers. If successful, transplanting can effectively bypass the stage from seed to seedling and the very high mortality associated with it (Harper 1977), in addition to reducing the time required for plants to achieve a size where they can begin spreading through the restoration site vegetatively and by seed. Despite the potential contribution of transplantation as a technique for reintroduction, there are little data assessing its effectiveness (Maunder 1992), and there are concerns over its use as a tool for conserving rare species (Howald 1996).

A general understanding in restoration is that site characteristics will have a large influence on transplant success (Maunder 1992; Primack 1996; Bratton & Meier 1998; Drayton & Primack 2000). Indeed, the distributions of woodland herbaceous perennials have been related to a wide range of environmental variables, including nutrients (Pigott & Taylor 1963), leaf litter (Sydes & Grime 1981), aspect (Hutchins et al. 1976), microrelief (Hicks 1980; Rogers 1982; Beatty 1984), soil moisture (Hicks 1980; Rogers 1982), soil depth (Hicks 1980), canopy composition (Hicks 1980; Beatty 1984), and light (Anderson et al. 1969; Moore & Vankat 1986; Reader & Bricker 1992). However, information from this literature has not converged on a subset of consistently important factors that could serve as a general guide for distributing species within a restoration site. Thus, there is a need to evaluate the feasibility of using information on the relationships between transplant performance, light, and other abiotic factors in planning restorations.

Our study evaluates the feasibility of introducing shade-tolerant perennial species to highly degraded wooded sites by quantifying transplant success rates over 7 years. We also examined whether light levels and soil characteristics were correlated with survival and flowering of these species. Few ecological field studies continue for more than 5 years (Tilman 1989); thus, our data provide a much more robust test of transplant survival than has generally been available.

Materials and Methods

Background and Study Site

The study was conducted at Camp Dodge, an Iowa Army National Guard base located in Johnston, Iowa, U.S.A. (e.g., the Midwest region). Much of the base lies within an old river terrace above a tributary of the Des Moines River, which borders the base on the east. The climate of the study area is midcontinental, with average winter temperatures of −6°C (22 F) and summer (June–August) of 22°C (72 F). The average frost-free growing season is 171 days. Total annual precipitation is 848 mm (33.4 inches), 73% of which occurs from April to September. (USDA 1941; Polk County Soil Survey 1960; Waite 1967).

The study site is an old homestead of approximately 3 ha on a gently sloping, well-drained west-facing slope (referred to as the Betz Site restoration). It was acquired in the early 1990s in order to restore it and adjacent areas to native plant communities. The Betz site was an ideal location to test the feasibility of establishing species that occur in shade to mixed sun–shade environments in our region because it had a variable canopy cover of Black walnut (*Juglans nigra*), Hackberry (*Celtis occidentalis*), White mulberry (*Morus alba*), and Red elm (*Ulmus rubra*) that could be used to examine the influence of various light levels on woodland perennial establishment and survival. In addition, an inventory of plots prior to the study revealed that the preexisting understory lacked herbaceous species that typify undisturbed woodlands in the region (Eilers & Roosa 1994; Mabry 2002) and was mostly composed of generalist and weedy species, including *Carex* species (*C. sparganoides*, *C. davisii*), Honewort (*Cryptotaenia canadensis*), White snakeroot (*Eupatorium rugosum*), Cleavers (*Galium aparine*), White avens (*Geum canadense*), Anise root (*Osmorhiza longistylis*), Clustered snakeroot (*Sanicula gregaria*), and Stinging nettle (*Urtica dioica*). Finally, because our site was an old homestead and not a woodland remnant, we could be confident that none of our target species existed at the site either in the seed bank or as emerged plants, and this was confirmed by the preintroduction inventory summarized above. The isolation of the site also ensured that our results would not be confounded by dispersal of target species into the site.

Thirteen 4 × 5.5–m permanently marked plots were subjectively distributed across the Betz site in May 1998. Plot locations were chosen to represent a range of light environments yet have relatively consistent light availability within the plot. Prior to planting, the plots plus an adjacent 1-m-wide buffer strip were sprayed with RoundUp^{TM} (glyphosate). By early June most of the plot vegetation was dead, except for *G. canadense*, *Carex* species (sedges), Gooseberry (*Ribes missouriensis*), and Black raspberry (*Rubus occidentalis*), which appeared unaffected by the herbicide application. No further treatments were used to remove these species, and gas-powered string trimmers were used to cut down the standing litter and stems for faster decomposition and to ease transplanting.

Species Choice

Because few shade-tolerant species were available from Iowa nurseries and we wished to use local genetic sources (e.g., Iowa sources), we were limited to species that could be purchased, grown readily from seed, or transplanted from local source populations sufficiently large to withstand a harvest. Thus, our ideal set of target species was balanced against these constraints and reflects a realistic restoration scenario. As best we could determine at the time, all the selected species are associated with forests and open woods in our region, although some are also associated with more open sites (Gleason & Cronquist 1991; Eilers & Roosa 1994).

None of the species selected would be considered rare overall in the state, but are, nevertheless, good candidate species for restoration. With the possible exceptions of Side-flowered aster (*Aster lateriflorus*) and Tall windflower (*Anemone virginiana*), none of the species included readily recolonized isolated forests or woodlands that are either secondary woods or highly disturbed. Although we cannot cite published studies to support this (because there were none at the time, and available local literature is limited to the grazing study mentioned below), our extensive collective experience with this system and unpublished data support the contention that these species are virtually absent from the sort of disturbed woods that are the focus of this work. In addition, the grazing study by Mabry (2002) confirmed that seven species included in our study are very highly sensitive to cattle grazing: Jack-in-the-pulpit (*Arisaema triphyllum*), Wild ginger (*Asarum canadense*), Dutchman's breeches (*Dicentra cucullaria*), Wild geranium (*Geranium maculatum*), Jumpseed (*Polygonum virginiana*), Zig zag goldenrod (*Solidago flexicaulis*), and Elm-leaved goldenrod (*S. ulmifolia*), and that none of the species included in the study are associated with grazed sites.

Planting and Field Methods

There were three sources of plant material (Table 1). Nursery seedlings of 11 species were purchased as plugs or 2-inch pots from Ion Exchange Native Seed and Plant Nursery, Harper's Ferry, in northeast Iowa (no information was available concerning genotypic variation within the species), and were maintained in a greenhouse at Iowa State University for 2 weeks prior to planting. Seven species were harvested from natural populations at Camp Dodge or central Iowa woodlands; individuals had from one to three leaves (1- or 2-inch clumps for *Carex*) and were maintained in the Iowa State greenhouse for 2-4 weeks prior to planting. Bulbs of one species, *D. cucullaria*, were harvested at Camp Dodge and were intentionally introduced, and a second set was unintentionally introduced to the site when bulbs were included with *A. triphyllum* harvested from a natural Camp Dodge population. Plants of Blunt-lobed woodsia (*Woodsia obtusa*) were grown from spores collected in central Iowa according to standard techniques for fern propagation. Five species grown from seed collected in central Iowa were sown in a soil mix of equal parts of mineral soil, sand, and milled peat, and were maintained in a greenhouse until they had several true leaves. We recognize that differences in stock sources may be important for restoration; however, for simplicity we use the term "transplant" throughout to indicate that plants were moved to the site, irrespective of source.

Prior to planting, each 4 × 5.5–m plot was divided into four 1 × 4–m planting strips and three 0.5-m walkways between strips to provide access to plants during monitoring. Each planting strip was further divided into sixteen 0.25-m^2 units (Fig. 1). Generally, four individuals of a single species were randomly assigned to a 0.25-m^2 planting unit, although for five species three individuals were planted per unit, and for two species five individuals were planted (Table 1). The number of units per species and individuals per unit were determined by the number of plants available for each species. Although availability was limited for some species because of budget constraints, limited seed, and constraints on transplanting from natural populations at Camp Dodge, we included all the species available in our study because although we

Table 1. Species planted as transplants at the Betz Site restoration, Camp Dodge, Iowa.

Species	No. Plots	No. Groups	No. Ind.	Transplant Source
Agastache sp.	2	3	3	propagated from purchased seed
Anemone virginiana	4	7	3	propagated from central Iowa seed
Aquilegia canadensis	2	8	4	harvest from central Iowa woodland
*Arisaema triphyllum**	11	51	3	harvest from Camp Dodge
*Asarum canadense**	12	57	4	harvest from Camp Dodge
*Aster lateriflorus**	13	61	4	nursery stock plugs
*Carex pensylvanica**	12	60	4	harvest from Camp Dodge
*Dicentra cucullaria**	7	42	5	harvest from Camp Dodge
Dicentra cucullaria (U)	11	51	NA	harvest from Camp Dodge
Elymus hystrix	7	31	4	nursery stock 2-inch pots
Elymus virginicus	2	6	3	nursery stock 2-inch pots
*Geranium maculatum**	9	36	4	nursery stock plugs
*Hydrophyllum virginianum**	12	57	4	harvest from Camp Dodge
*Lobelia siphilitica**	13	61	4	nursery stock plugs
Mitella diphylla	2	7	4	propagated from central Iowa seed
*Phlox divaricata**	10	45	4	harvest from Camp Dodge
*Polygonum virginianum**	4	9	4	propagated from central Iowa seed
*Prenanthes alba**	4	19	4	nursery stock plugs
*Rudbeckia laciniata**	13	62	4	nursery stock plugs
*Rudbeckia triloba**	7	32	4	nursery stock plugs
*Solidago flexicaulis**	3	11	4	nursery stock plugs
Solidago nemoralis	4	24	4	nursery stock plugs
Solidago ulmifolia	2	2	5	propagated from central Iowa seed
*Veronicastrum virginicum**	9	35	4	nursery stock plugs
Woodsia obtusa	3	6	3	propagated from central Iowa spores

No. Plots, number of plots/species; No. Groups, number of 0.25 m^2 planting units/species; No. Ind., number of individuals in a group; *, species quantified for flowering; U, unintentionally planted; NA, not applicable.

were interested in robust quantifiable results, we were also interested in creating a more complete restoration than would have resulted from a purely experimental situation.

Most species were transplanted during a 2-week period beginning 8 June 1998. Watering was unnecessary because it rained frequently during the 2-week period. As they became available, some plants were added later in the summer and were watered shortly after planting. Plots were weeded periodically in 1998 to eliminate competition from weedy species emerging from the seed bank but were only weeded once in May 1999 and were not weeded at all in 2000–2005.

In 1999, photon flux density (PFD; 400–700 nm) was measured in μmoles m^{-2} s^{-1} at four locations per plot (Fig. 1) using Campbell Scientific 21X data loggers and quantum sensors (either Li-Cor 190SA sensors or similar sensors constructed by personnel in the ISU Department of Botany). The data loggers recorded PFD each minute and stored averages over 15-minute intervals for each sensor. Sensors were attached to vertical dowel rods 50 cm above the ground, just above the tallest species in the plots. Data were recorded from June 30 to July 17, and from September 15–18. Sampling in September was necessary to replace data lost from June. Readings were taken from approximately 1000–1400 local time to include a 4-hour interval with peak daily solar irradiance. A sensor and data logger were placed in the open at 50 cm aboveground to record total photosynthetically active radiation (PAR) (used to calculate percent PAR in the plots). In August 1999 four soil cores per plot (one from each planted strip) were taken and pooled into a single sample for analyses of phosphorous, potassium, nitrogen (nitrate-nitrogen), percent organic matter, and pH. Analyses were carried out by the Iowa State University Soil-testing Lab, Ames, Iowa.

Survival was monitored for 24 species in 13 plots in 1999, 2000, and 2004–2005. In 2003, the same species were monitored in 11 plots (two plots were accidentally mowed and could not be sampled). Because flowering is an indicator of seed production and site establishment, 15 of the species were monitored for flowering in 1999 and 2000, although data from three planting units of Oak sedge (*Carex pensylvanica*) were recorded for flowering for the first time in 2003, and were also included. Not all species were monitored for flowering due to time constraints and because not all species had flowered. However, we do not believe this biased our results; based on what we have observed subsequently about the species that were not scored for flowering, their exclusion probably resulted in an underestimate of percent flowering. Because of the difficulty in tracking individual stems within units after the second year, survival data were based on presence or absence of species in the 0.25-m^2 planting units. Species in the planting units are hereafter referred to as "groups." So, that flowering data were presented comparably; they are also presented by planting unit. That is, planting units (with one species per unit) were scored for survival and

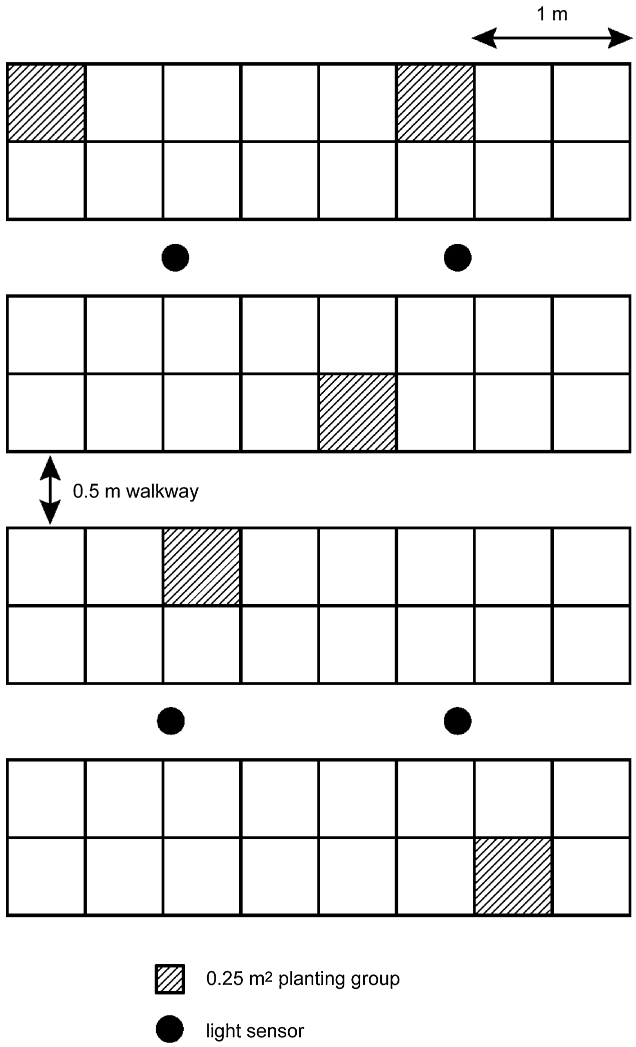

Figure 1. Plot layout and locations of light sensors for a study of transplant survival. One plot comprises four planting strips.

flowering. Survival data for 1999 are based on the last census date for each species, which varied according to species-specific phenologies. Survival data for 2000 and 2003–2005 are based on surveys conducted in April and May, and thus may somewhat underestimate *A. lateriflorus*, which has its peak growth in summer. All other species were easily recognized in April and May. Seedlings and mature plants outside the planting units and plots were also noted to document species that spread by seed and rhizomes.

Data Analysis

Differences among years in the percent groups surviving were compared using 95% confidence intervals. To ensure our results were not biased by including species with small sample size, we also analyzed our data with only species occurring in at least nine, or 70%, of the plots. Results were very similar and are not presented here. Pearson Product–Moment correlations were calculated to evaluate the association between environmental variables, and survival and flowering. Table-wide significance levels were adjusted using the sequential Bonferroni method (Rice 1989). We recognize that our study included only one site and our statistics should not be extrapolated to other sites; however, confidence intervals and correlation p values are presented to aid in interpreting the patterns observed in the descriptive presentation of our data (Oksanen 2001).

Results

Mean survival of the groups of 24 species was 91% in year 1, 85% in year 2, and over 50% in years 5–7 (Table 2). Confidence intervals suggested that survival differed between the two early and the three later samples but did not differ from years 1 to 2 or 5 to 7 (Fig. 2). Survival in years 5–7 does not reflect individuals that spread beyond the original planting units by self-sowing or vegetative spread (see below) and is therefore a minimum estimate of the abundance of many species at the site. Brown-eyed Susan (*Rudbeckia triloba*) is a biennial or short-lived perennial, so it was not included in calculating survival after the first year. Two species, *Lobelia siphilitica* and Gray goldenrod (*Solidago nemoralis*), did not survive to year 5 possibly because the habitat was too dry for *Lobelia* and too shaded for *S. nemoralis* (Eilers & Roosa 1994). When survival was tabulated without these two species, overall survival of the remaining species was around 60% (Table 2). For species surviving to year 7, those with three individuals per planting group had survival nearly the same as those with four (61.9 vs. 61.6%), indicating that this difference in number of individuals per group did not influence the results.

Mean percent flowering of groups for the 15 species monitored was 72% (Table 3). In addition, the eight other angiosperm species were observed flowering in the plots, although the numbers of groups with flowering individuals were not recorded. Spore production by *Woodsia obtusa* was not observed over the course of the study.

By year 7, 17 of the 24 species were spreading by seed and/or rhizomes (Table 2). Although not quantified, the expansions of *Carex pensylvanica, Dicentra cucullaria,* Bottlebrush grass (*Elymus hystrix*), *Hydrophyllum virginianum,* Woodland phlox (*Phlox divaricata*), Cut-leaved coneflower (*Rudbeckia laciniata*), *R. triloba*, and *S. ulmifolia* were particularly notable. For example, *Carex, Elymus, Hydrophyllum*, and *R. laciniata* had formed some patches of at least 3 m^2, and the other species appeared as scattered individuals that established outside their original planting units or outside the plots. *Anemone virginiana*, Virginia wild rye (*Elymus virginicus*), Bishop's cap (*Mitella diphylla*), white lettuce (*Prenanthes alba*), Culver's root (*Veronicastrum virginicum*), and *W. obtusa* persisted vegetatively, although they did not appear to be spreading beyond the original planting units. As noted above, only *S. nemoralis* and *L. siphilitica* failed to persist.

Table 2. Mean percent survival of species groups at the Betz Site restoration, Camp Dodge, Iowa.

Species	*1999*	*2000*	*2003*	*2004*	*2005*
Agastache species*	NA	100.0	66.7	66.7	33.3
Anemone virginiana	NA	100.0	33.3	28.6	71.4
*Aquilegia canadensis**	NA	100.0	62.5	37.5	50.0
*Arisaema triphyllum**	100.0	100.0	60.9	49.0	88.2
*Asarum canadense**	100.0	91.2	59.6	50.9	52.6
*Aster lateriflorus**	100.0	95.1	13.7	14.8	16.4
*Carex pensylvanica**	100.0	70.0	78.0	66.1	61.0
*Dicentra cucullaria**	88.1	85.7	13.3	42.9	33.3
Dicentra cucullaria (U)*	NA	74.5	67.7	49.0	58.8
*Elymus hystrix**	92.6	96.8	51.6	61.3	58.1
Elymus virginicus	NA	40.0	100.0	83.3	83.3
*Geranium maculatum**	100.0	100.0	90.6	88.9	94.4
*Hydrophyllum virginianum**	100.0	98.2	100.0	100.0	96.5
Lobelia siphilitica	98.4	62.3	0.0	0.0	0.0
Mitella diphylla	100.0	100.0	14.3	66.7	50.0
*Phlox divaricata**	100.0	100.0	62.5	62.2	73.3
*Polygonum virginianum**	88.9	88.9	50.0	100.0	88.9
Prenanthes alba	42.1	68.4	42.9	36.8	52.6
*Rudbeckia laciniata**	100.0	93.5	59.6	53.2	51.6
*Rudbeckia triloba**	71.9	NA	NA	NA	NA
*Solidago flexicaulis**	100.0	100.0	54.5	54.5	54.5
Solidago nemoralis	58.3	12.5	0.0	0.0	0.0
*Solidago ulmifolia**	100.0	100.0	100.0	100.0	100.0
Veronicastrum virginicum	88.6	88.6	67.7	54.3	62.9
Woodsia obtusa	NA	83.3	33.3	33.3	33.3
Mean survival	91.0	85.4	53.5	54.2	56.9
Standard deviation	16.3	21.9	30.0	28.1	28.0
Mean survival without *Lobelia*, *S. nemoralis*	92.5	89.7	58.3	59.1	62.0
Standard deviation	15.1	15.0	26.3	23.7	22.9

*, species that spread by seed or vegetatively; U, unintentionally planted; NA, not sampled. Variation in survival for some species in 2003–2005 may be due to slightly less sampling intensity in 2003 and year-to-year variation in emergence for some species.

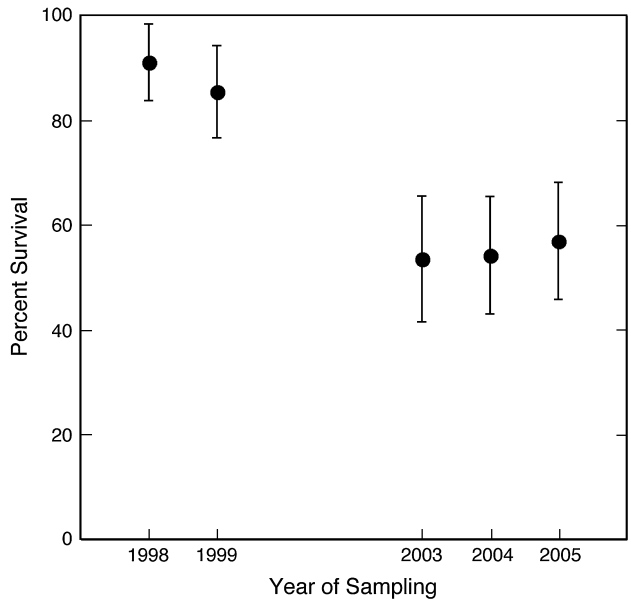

Figure 2. Mean percent survival of species groups sampled five times over 7 years at the Betz Site restoration, Camp Dodge, Iowa. Bars are 95% intervals. Sample size was 13 plots in all years, except for 2003, where it was 11 (see text for explanation).

Mean percent flowering of groups for the 15 species monitored was positively correlated with plot percent PAR ($p = 0.031$; Table 4). There were few strong correlations between the other measured environmental variables and flowering or survival, and none was consistent from year to year (Table 4). The strongest association was between nitrogen and percent survival in year 6 ($p = 0.036$). However, after adjusting for multiple comparisons this correlation was not statistically significant at the table-wide level of 0.01. See Appendix 1 for environmental factor values by plot.

Discussion

Significant challenges exist for restoring native perennial herbaceous diversity to degraded woodlands. Fire and other management techniques may help to restore diversity to sites with a moderate disturbance history when there is a remaining seed bank and/or vegetative propagules (Ladd 1991; McCarty 1998). However, for new secondary woodlands and heavily disturbed sites, where the ground cover has been converted to non-native cool season grasses, or where there is an understory dominated by extreme habitat generalists and species that readily invade

Table 3. Percentage of groups flowering for 15 species, Betz Site restoration, Camp Dodge, Iowa.

Species	*n*	*Percent (1999–2000)*
Arisaema triphyllum	50	32.0
Asarum canadense	52	96.2
Aster lateriflorus	61	96.7
Carex pensylvanica	60	78.3
Dicentra cucullaria	42	19.0
Geranium maculatum	36	94.4
Hydrophyllum virginianum	57	100.0
Lobelia siphilitica	61	60.7
Phlox divaricata	45	95.6
Polygonum virginianum	9	77.8
Prenanthes alba	19	42.1
Rudbeckia laciniata	62	100.0
Rudbeckia triloba	29	75.9
Solidago flexicaulis	12	100.0
Veronicastrum virginicum	35	17.1
Mean flowering		72.4
Standard deviation		30.6

n, number of transplant groups.

disturbed sites, restoration will likely require species introductions, particularly in highly fragmented regions.

Our 7-year survival data for 24 shade-tolerant perennial herbaceous species introduced into a highly degraded woodland suggest that it is feasible to increase woodland herbaceous perennial diversity by introducing transplants. We observed over 50% 7-year survival across 783 groups of transplants, despite absence of any effort to control weeds beyond the first 2 years, and below normal rainfall in 2000 and 2002–2003. High flowering percentages across 15 species, stable percent survival between years 5 and 7, and the spread of several species outside the original planting units also indicated transplant success. The results are particularly encouraging for species like *Arisaema triphyllum, Asarum canadense, Carex pensylvanica, Dicentra cucullaria, Geranium maculatum, Hydrophyllum virginianum*, and *Phlox divaricata* because these have one or more of the following characters that make reestablishment by seed slow or problematic: low seed production, lack of long-distance dispersal mechanisms, intolerance of dry storage, complex germination requirements, and slow growth (Bierzychudek 1982; Cullina 2000; Mottl & Mabry 2004). For species with one or more of these characters, our results indicate that transplanting may be the most promising means of reintroduction.

Table 4. Pearson product–moment correlation coefficients between flowering, survival, and measured environmental variables for species transplanted to the Betz Site restoration, Camp Dodge, Iowa.

	Light	*N*	*P*	*K*	*Organic Matter*	*pH*
Percent flowering	0.598 0.031	−0.087	0.371 0.212	0.209	−0.189	−0.016
Percent survival in year 1	−0.065	−0.078	0.281	0.237	0.133	0.295
Percent survival in year 2	−0.509 0.076	0.055	0.285	0.278	0.183	0.441 0.132
Percent survival in year 6	−0.028	−0.585 0.036	0.256	−0.386 0.192	−0.500 0.082	0.076
Percent survival in year 7	−0.314 0.296	−0.341 0.254	0.046	−0.074	−0.027	0.281

Numbers below the correlation coefficients are *p* values, shown only for values ≤0.300. Data for years 6–7 do not include *Lobelia siphilitica* or *Solidago nemoralis* (see text for explanation).

However, results from other transplant studies have been mixed. In an apparently successful case of reintroduction, Ellarson and Craven (1982) transplanted one, two, or an unspecified number of 14 shade-tolerant species into six 2 × 2–m plots located in northern Wisconsin, U.S.A., and monitored survival over 5–8 years. Only one species did not survive, and species were present in at least five of the six plots 8 years after transplanting. Three were reported to be well established and reproducing, with no information on reproduction in other species given. Glitzenstein et al. (2001) reported 5-year survival ranging from 73–82% for three species native to longleaf pine savannas of the southeastern United States. In contrast, Drayton and Primack (2000) found that 3-year survival of eight New England forest species averaged only 28 and 17% for seedlings and mature adults, respectively. There was no posttransplant care given in the latter study, which occurred in a dry period and may explain the low survival. Level of transplant care was not indicated in the other two studies. Whether this regional variation arises from a small number of comparisons and differences in site preparation and postplanting transplant care, or is due to intrinsic regional differences is worthy of further investigation. In addition, our results apply to only one site, and success rates should be evaluated across a range of sites within the region. With the exception of one rare herbaceous species in the study by Glitzenstein et al. (2001), the species in our study and those cited above did not include rare plants, and it should not necessarily be assumed that our results would apply to translocations to mitigate for loss of truly rare species (Howald 1996).

A significant challenge for restoring woodland perennial herbaceous species is acquiring plant materials for restoration. Many species, particularly spring ephemerals and sedges, are not suited for reintroduction by seed because they tend to produce few seeds per plant, have a narrow window between seed maturation and seed drop, do not tolerate dry storage, may require time-consuming cleaning, and grow slowly (Bierzychudek 1982; Cullina 2000; Mottl & Mabry 2004). Transplanting from natural populations cannot be deemed a success unless source populations are able to replace lost individuals through sexual or vegetative reproduction. Further research is needed to determine what species have this capacity and, for those that do, to determine the replacement rate of harvested individuals, important information for sustainable harvest. Identifying woodland species that could be efficiently introduced by seed would also improve the success and feasibility of large-scale woodland herbaceous restoration.

We found that light and flowering were positively related in the year that flowering was measured and that soil nitrogen was negatively related to survival in year 6; however, these relationships were not significant at the table-wide level. The weak correlations observed in our study should be taken with the caveat that our sample size was small and our study of environmental variables was observational rather than experimental. Thus, we did not test light and nutrients replicated across a range of levels where other factors were controlled. In addition, conclusions should not be inferred from a single point in time because plant response to the environment may change over time (Tilman 1989).

Because forest perennial herbs have often been associated with light and nutrients, and a wide variety of other environmental and disturbance factors, restoration often emphasizes deliberately planting species to match them to their preferred microenvironments or "safe sites" (Maunder 1992; Primack 1996; Bratton & Meier 1998; Drayton & Primack 2000). Our study suggested, however, that detailed knowledge of variation in light and other abiotic factors may not be needed to undertake woodland perennial restoration, at least at the small scale of our 3-ha study area.

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Appendix 1. Environmental variables associated with 13 transplant plots at the Betz Site restoration, Camp Dodge, Iowa.

Plot	*Light (% PAR)*	*N (ppm)*	*P (ppm)*	*K (ppm)*	*Organic Matter (ppm)*	*pH (ppm)*
1	15	1	46	84	2.4	7.45
3	15	3	50	118	3	7.3
5	7	5	72	156	2.7	7.15
6	10	3	46	114	2.4	7.35
7	11	3	64	117	2.6	7.2
9	11	3	60	116	2.5	7.05
12	11	2	48	54	2.4	6.85
13	5	2	26	95	3.5	7.35
15	38	3	30	116	2.4	6.8
19	25	2	76	232	4.5	6.8
20	9	1	52	107	2.6	6.6
21	11	2	104	154	2	6.75
23	54	2	100	126	2.2	6.75