Changes in the Forests of the Jockey Hollow Unit of Morristown National Historical Park Over the Last 5-15 Years

Emily W. B. (Russell) Southgate

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National Park Service
Boston Support Office
Natural Resources Management
15 State Street
Boston, MA 02109-3572

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EXECUTIVE SUMMARY

The forests of Morristown National Historical Park (MORR) (Morris County, NJ) constitute a major element of the cultural as well as the natural landscape of the park. The forested vegetation of the park is not homogeneous; rather it includes older mature stands of varied species composition and a variety of younger stands that have replaced fields abandoned in the last century or more. These forest stands are undergoing changes today, in a complex interplay between the natural dynamics of forest vegetation and the consequences of human activities. These changes are consistent with park objectives when they result in a continued healthy, ecologically integrated forest. On the other hand, if the changes are tending toward an ecologically impoverished forest, with little resemblance to the vegetation of the 18th century, it may be necessary to intervene to reverse these trends.

Study of the vegetation of the park over the last 15 years suggests that, while superficially many parts of the forests of MORR appear healthy, there is abundant evidence that the future is unpredictable based on current regeneration. Most critical are the secondary successional stands where yellow poplar or black locust dominates. In these forests there are almost no saplings. The future of these forests is in jeopardy, probably in the near (decadal scale) future. The causes of the lack of regeneration are most likely complex, and reversing the problem will require consultation with foresters familiar with similar stands.

In more mature forests, the only abundant saplings are beeches, which occur only in some stands. There are, however, also red maple and black birch saplings. There is a lack of seedlings that grow vigorously or survive for more than a couple of years. The problem here is less critical than in the successional stands, but should be monitored carefully and perhaps experimental management instituted to find out why, other than deer browsing, most seedlings do not survive. Soil chemistry is a potential factor.

The shrub layer of the vegetation presents a complex issue. It appears that over the last 2-3 decades, deer have nearly eliminated most palatable shrubs. It is possible that this lack of shrub understory is appropriate for the historical period, when free-ranging livestock would have decimated the palatable plants. However, non-native shrubs such as Japanese barberry have replaced the native shrubs in many areas, growing in very dense thickets, creating a distinctly an ahistorical forest appearance. The monitoring data suggest that there are underlying reasons why barberry and other invasive non-native species are restricted to some parts of the park, and these reasons should be elucidated to help in control of these shrubs.

Removal of deer has both positive and negative impacts on the understory vegetation of the park. On the positive side, it allows native shrubs and herbs, and possibly tree seedlings, to survive in some areas. On the negative side, it also allows vigorous growth of non-native spiny shrubs such as multiflora rose. Even Japanese

vigorous growth of non-native spiny shrubs such as multiflora rose. Even Japanese barberry does better without deer browsing. It is likely thus that a reduction in deer herd would allow a proliferation of even more non-native shrubs than are already present at the park.

Changes over the last 5-15 years indicate that to establish and maintain a health forest at the park, immediate experimental intervention is important for many of the successional forests. This might include the establishment of larger exclosures, soil studies, and other management options. Testing these options over the next 5-10 years would provide data on which to base more extensive treatments. For example, decreases in the deer herd at least locally and for short periods may be necessary to encourage the reestablishment of healthy seedling growth, but other factors must also be investigated. Complete elimination of deer appears to allow the establishment of native tree, shrub and herbaceous species in some areas.

Understanding the factors that lead to establishment of Japanese barberry and Japanese wiregrass (as well as other invasive species) in some areas rather than others may help in controlling their distribution.

Some of the changes observed over the last few years are consistent with the continuation of a healthy, ecologically integrated forest, primarily in the older forest areas. On the other hand, changes observed in the secondary successional forests appear to be tending toward an ecologically impoverished forest, which will probably lack not only the species, but also the structure of the vegetation of the 18th century. It may be necessary to intervene to reverse these trends.

INTRODUCTION

The forests of Morristown National Historical Park (Morris County, NJ) constitute a major element of the cultural as well as the natural landscape of the park (Russell 2001). The forested vegetation of the park is not homogeneous; rather it includes older mature stands of varied species composition and a variety of younger stands that have replaced fields abandoned in the last century or more (Ehrenfeld 1977). These forest stands are undergoing changes today, in a complex interplay between the natural dynamics of forest vegetation and the consequences of human activities. These changes are consistent with park objectives when they result in a continued healthy, ecologically integrated forest. On the other hand, if the changes are tending toward an ecologically impoverished forest, with little resemblance to the vegetation of the 18th century, it may be necessary to intervene to reverse these trends. I have proposed several approaches to monitoring change in the forests of Morristown National Historical Park (Russell 2001) to provide a scientific basis for making decisions about possible management actions. Here I report on preliminary results from implementing this monitoring; integrating it with results from other studies initiated in the last 15 years.

A healthy forest is characterized by continuing growth of mature trees, regeneration of tree species adequate to replace trees that die, and the existence of a variety of native shrub, herbaceous, fern and moss species in the understory. At Morristown National Historical Park it would also seem critical that the structure of the forest should be that of a natural forest. With a variety of ages and species of trees, shrubs, herbaceous species and dead wood and other litter, the landscape has a feeling of the unmanaged, uncontrolled, semi-natural forest of the 18th century as opposed to a neat, landscaped park of planted trees and shrubs. It may also be important that the flora consist of native species in order to evoke the environmental atmosphere of the time of the encampment (Russell 2001).

At the present time, much of the forest vegetation of the park, both older and successional, consists of mature deciduous trees with native shrub and herbaceous understory. Throughout, however, non-native shrubs and herbaceous species are spreading, and there is an apparent dearth of seedling and small sapling regeneration and of vigorous native herbaceous species. In order objectively to describe changes that are occurring in the vegetation of the park, it is necessary to quantify the changes by developing data sets that are comparable over time. The park has begun to establish the research necessary to accomplish this goal. In this report, I will discuss preliminary results from this research and indicate tentative conclusions that can be drawn from these preliminary results as well as their consequences for the future viability of a healthy forest at the park. Carefully repeated studies are necessary either to avoid accepting apparently obvious, immediate causes for changes, when perhaps other, more subtle causes may be more important, or the rates of change which may themselves change over time.

The data sets collected to quantify change in several aspects of the forests of the Jockey Hollow Unit of Morristown National Historical Park include 13 permanently marked plots established in 1995 to study trees, shrubs and herbaceous layer vegetation, sampled in 1995 and again in 2001; 5 deer exclosures, established in 1987-1988 and sampled annually since then; 14 small, permanently marked plots to study woody seedling establishment and survival, established in 1995 and sampled at least once a year since then; and 2 transects established in 2001 to study replacement of dead trees in two major types of forest. Two of these studies address comparatively specific perceived problems in the park: excessive deer browsing and the spread of non-native species. The original deer exclosures provide comparisons between browsed and unbrowsed plots. The permanent vegetation sampling plots provide comparisons between areas where there is an abundance of non-native species with those where these species are absent or rare, since half were located in areas with high densities of non-native species and half in areas with no or almost no individuals of non-native species. Data from several other deer exclosures established for various other reasons also suggest possible effects of deer browsing.

The small size of the existing data sets precludes rigorous statistical analysis, which would require a very large study. Because the plots have been sited randomly or at least haphazardly, however, they do provide a reasonably unbiased estimate of change, which can suggest potential problems with forest regeneration. These data can be interpreted in terms of consistent trends or very marked differences. The function of statistics is to indicate whether small differences are real or are attributable to sampling procedures, random variation, etc. When the differences are striking, it is not necessary to use statistics to provide support for the conclusions. However, lack of obvious differences does not prove lack of subtler but statistically significant differences.

PERMANENT VEGETATION MONITORING PLOTS

These plots were established in 1995 (?) to compare changes in vegetation over time in areas of the forest where Japanese barberry (Berberis thunbergii) and Japanese wire grass Microstegium vimineum have spread with those where they are absent (or only very sparingly present). Repeated measurements of all plants in the plots at 5-10 year intervals will reveal patterns of change. Comparisons among the plots may suggest different factors, such as age or composition of the forest, presence or absence of non-native species, or aspect, that may contribute to differences in rates and directions of change.

METHODS

Of the 18 plots established in 1995, we were able to locate 13 for resampling in 2001. The data from these 13 provide valuable insight into changes over the last 6 years.

In each 20m x 20m plot, all trees (> about 2 cm **dbh** or diameter at breast height) were measured and tagged in 1995. Within each plot are nested 4 5m x 5m subplots in

which shrub cover and species presence were recorded and 9 1m x 1m subplots in which herbaceous cover and species presence were recorded. A visual estimate of cover appears to provide a repeatable and accurate assessment of plant cover (Brakenhielm and Qinghong 1995). The details of the plots and data collection are described in Ehrenfeld (1999). In 2001, we repeated all of the measurements as described for the original data collection, correcting the tree misidentifications (especially a large number of red maples (Acer rubrum) misidentified as sugar maples (Acer saccharum) and American beech (Fagus grandifolia) mislabeled as black cherry (Prunus serotina)). This modifies the original results in terms of species composition, but not in terms of total basal area. In 2001, unfortunately, neither plot located in the New Jersey Brigade area was located, nor were 4 other plots in the Jockey Hollow Area, leaving 13 of the original 18 plots to remeasure. Thus, the data that I will discuss refer only to plots in the Jockey Hollow Area.

In the field, it was decided that woody stems less than 10 cm tall would be counted in the herbaceous count. Woody stems >/=10 cm and less than 2 m tall were counted in the shrub tallies. In other words, the terms herbaceous and shrub referred to size of the plants, not their growth forms. It was not clear from the descriptions of the data collection in 1995 what the distinction was, since tree species were included in both the shrub and herbaceous data lists. Shrub counts included all species with distinctly shrubby growth form over 10 cm tall. Saplings were all stems >2 m tall and <5 cm dbh. Standing dead trees were tallied in both censuses.

The categories describing the forest floor were expanded in 2001. In 1995, the category "bare soil" apparently included all the surface area not covered with low vascular plants. The categories used in 2001 were bare soil, leaf litter, woody litter (logs), rocks, moss, and tree bases.

RESULTS

Trees: All tree and sapling data are shown in Tables 1 and 2. Figures 1 and 2 show the numbers of individuals of all tree species with 3 or more stems in any plot, by plot. The same 5 species (black birch, red maple, American ash, American beech and red oak) were most common at both sampling periods, but the order differed. Black birch and red maple were the most common trees at both time periods, but American beech trees had become more numerous and American ash less numerous by 2001. There were the same number of red oaks both years. Black locust (Robinia pseudoacacia), a non-native tree, was present only in invaded plots. White ash, an early successional species, was also only in the invaded plots with the exception of one tree in one plot in 1995. American beech, on the other hand, a species of the mature forest, was very common in two of the uninvaded plots and present in only one of the invaded ones.

There were fewer trees in 2001 than in 1995, 151 versus 173, but approximately the same basal area, 24 versus 23.1 m^2/ha, suggesting that tree growth balanced out tree death. There were slightly fewer tree/plot in the invaded than in the uninvaded plots in 1995 (12 vs. 13.9) but the difference had become greater by 2001 (9.7 vs. 13.3). Almost all trees had gained in diameter over the 6 years. Mortality was greatest for flowering

dogwood and small beech trees, as well as ash, red maple and black birch. A few trees classified as saplings (<10 cm dbh) in 1995 had grown over this limit by 2001 and were counted as trees. The stocking of 23 m^2/ha is close to that considered by the US Forest Service to be fully stocked (21 m^2/ha) (DiGiovanni and Scott 1990).

Figure 3 shows the size-class distribution of the trees in all plots for 2001 compared with those that had died since 2001 or that were listed in 1995, not found in 2001 and assumed to be dead. The curve for the live trees approximates the "reverse-j" curve typical of a healthy forest. On the other hand, one large tree in the 50-55 cm dbh size-class had died, but most of the mortality was in the smaller size-classes. Size-class comparison between invaded and uninvaded plots does not show a difference in trees >/= 10 cm dbh (the cut-off between trees and saplings), but does show a large difference in the sapling size class, with the uninvaded plots having many more saplings (Figure 4). Only one of the total of 62 oaks (Quercus spp.), hickories (Carya spp.) or beeches, genera characteristic of mature forests, died between 1995 and 2001.

This rosy picture of a healthy forest is compromised, however, by the sapling data (Figure 4). The diversity of stems among the saplings is much lower than among the trees; there were 17 species of trees present and only 11 sapling species in 2001. However, there were approximately the same number of total saplings in the plots in 2001 as in 1995, due to a balance between loss and recruitment, especially of beech saplings, but also one hickory. Of the sapling species, 3 (flowering dogwood, ironwood (Carpinus caroliniana) and shadbush (Amelanchier arborea)) do not generally reach canopy status, and only 3 canopy trees, beech, red maple, and black birch were common. These are, however, the three species that were most common in the canopy layer as well in 2001. Oaks were missing (except for one individual) in the sapling layer, as is typical of oak-dominated forests throughout the northeastern U.S. This "oak regeneration problem" has been studied extensively, but no satisfactory explanation has yet been found, nor any management recommendations other than intensive management of cover and stocking (Lorimer 1989).

The difference between the uninvaded and the invaded plots, however, is very striking in terms of the saplings. There are almost no saplings in the invaded plots (Figure 4), and those that are in this size class are mostly flowering dogwood. This does not augur well for the future of these stands, as it indicates an almost complete dearth of regeneration of canopy trees.

Shrub layer: Some data on shrubby species that had stems greater than 2 cm dbh were collected as part of the tree data in the whole plots. This included spicebush (Lindera benzoin), witch hazel (Hamamelis virginiana) and blackhaw (Viburnum prunifolium). Witch hazel occurred only in uninvaded plots, often in large numbers, and the other two only in invaded plots. All decreased in numbers of stems between 1995 and 2001. Spicebush, which was present in only low numbers in 1995 was gone by 2001, although it was still present in one plot in the shrub category.

Data collected in the shrub subplots shows consistency in the distribution of barberry from 1995 to 2001 (Tables 3 and 4). It was present in only 2 subplots of one

uninvaded plot in 1995 and was still present only in these in 2001, still in low density. It was present in one subplot of another uninvaded subplot in 1995 and in another subplot of the same plot in 2001, again at very low density. Barberry was absent in all other subplots of the uninvaded plots both in 1995 and in 2001. Thus barberry did not appear to have spread by 2001 into areas where it had been absent in 1995. This is consistent with Ehrenfeld's observations on remeasured points between 1994 and 1995 (Ehrenfeld 1999). However, in contrast to her data, the data collected here indicate a large increase in the numbers of barberry stems between 1995 and 2001.

There were no other non-native shrub species in uninvaded plots in 1995, and only one stem each of Oriental bittersweet (Celastrus orbiculatus) and burning bush (Euonymus alatus) in 2001. The plots with barberry and Microstegium, however, had other non-native shrub species in both 1995 and 2001, with more species in 2001. Wineberry (Rhus phoenicolasius) stems were especially numerous in 2 of the invaded plots, and appeared to be increasing in density.

Diversity but not density of native shrub layer species apparently increased from 1995 to 2001, with 23 species of native species in 2001 in the invaded plots and 12 in the uninvaded plots. Shadbush was the only one of these species to be present in 1995 and not in 2001. The shrub layer species included tree "seedlings" – i.e., woody plants taller than 10 cm. but less than 2 cm dbh, suggesting that there is some regeneration of tree species, including oaks and hickories that was not recorded in the sapling layer. There were considerably more native shrub layer species in the invaded than in the uninvaded plots. While there was an increase in number of species between 1995 and 2001, there was a decrease in the numbers of stems. Only a few species, e.g., a species of cherry (Prunus sp.) and an unknown woody shrub, increased between 1995 and 2001.

Herbaceous layer: As with the barberry, Microstegium remained where it had been in 1995, not spreading into uninvaded plots or subplots (Table 5). There was generally lower cover of Microstegium recorded in 2001 than in 1995, but that could be due to the time of sampling. However, as this grass grows rapidly over the summer and data in 2001 were collected at the end of the summer, when it is at its greatest extent, this seems an unlikely explanation. With an annual such as this, weather conditions over the summer may affect its cover.

There were some notable changes in herbaceous species over the 6 years. The extent of white snakeroot (Eupatorium rugosum), a species that deer do not eat, shrank between 1995 and 2001. In plot CE2, it appears that another inedible herbaceous species, garlic mustard (Alliaria officinalis) had replaced it. In 1995, there was an unknown in several subplots of plot BE, where garlic mustard was present in 2001, suggesting that garlic mustard may have been more widespread in 1995 than the data show. This was the most common non-native herbaceous species, other than Microstegium in 2001.

There were no other non-native herbaceous species in the herb layer in either year. There were more non-native and native species in both categories of plots in 2001 compared with 1995, but this may be due to time of year or better taxonomy. It seems unlikely that these mostly perennial herbaceous species only appeared in this time period.

One fruiting stalk of jack-in-the-pulpit (Arisaema triphyllum) was found among the Microstegium in one plot in 2001.

CONCLUSIONS

As an overall picture the forests of the Jockey Hollow section of Morristown National Historical Park represent young, well-stocked stands. There is a good diversity of species present. Between 1995 and 2001 the total number of trees in the plots decreased, but because the remaining trees grew and much of the mortality was in the smaller class sizes, the basal area/ha remained approximately the same. This is typical of a young, maturing forest.

However, there is a significant difference between plots with invasive species (Japanese barberry and/or Japanese wire grass) and without these species, or with very few individuals of them. The uninvaded plots fit the pattern described for the overall forest. The invaded plots, on the other hand, are almost completely lacking in regeneration of canopy species. They had 4 sapling stems in 6 plots in 2001, while the uninvaded plots had 85. However, most of the saplings in the uninvaded plots were beeches found in only two plots. The other plots had an average of only 5 saplings/plot. The number of tree stems/plot decreased from 12.0 to 9.7 between 1995 and 2001 in the invaded plots, but only from 13.9 to 13.3 in the uninvaded plots.

Based on the loss of trees and the lack of saplings in the invaded plots, it appears that these areas may be in danger of losing their forest canopy as canopy trees die and have no replacements. Even the uninvaded plots have only a low number of saplings, except in areas dominated by beech saplings.

DEER EXCLOSURES

It is apparent in the park that deer browsing has reduced the populations of palatable species. Deer exclosures can allow us to address the potential for these species to recover in the complete absence of deer browsing. Several specific questions may be addressed by comparing data from deer exclosures with controls to evaluate the impact of deer exclusion:

1) Do exclosure and control plots become more dissimilar in species composition over time as palatable species begin to recolonize the exclosures?

2) Does the number of native species change in the deer exclosures as compared with the controls over time?

3) Does the cover of non-native species change in the deer exclosures as compared with the controls over time?

4) Are there certain palatable and unpalatable species whose distribution is significantly affected by preventing deer browsing?

The plots serve to demonstrate the most salient effects of complete elimination of deer browsing in several vegetational contexts. They can also serve as comparisons for inferences about the effects of deer browsing in other, unfenced, study plots. For example, if it appears that deer browsing causes a certain change in an unfenced plot, but the same change occurs within the deer exclosures, other explanations must be sought.

METHODS

In 1987 and 1988 the park established 5 20' x 20' deer exclosures, to demonstrate the impact of deer browsing on the vegetation of different kinds of forest. Each included an unfenced control plot located just outside the exclosure and surrounded by a high wire supported by posts to allow comparable bird perches for the control and the exclosure. Two exclosures were placed in mature oak or oak/yellow poplar stands (#1 and #5), one in a black locust/white ash successional stand (#4), one in a white ash/yellow poplar successional stand (#2), and one in an open field (#3).

Once a year, generally during late summer, the plots were all censused. From 1988-1990 one 1x1 m subplot was censused per exclosure and per control. From 1991 to 1997, 3 randomly selected 1x1 m subplots were censused for each exclosure and control. From 1998-2001, 9 contiguous 1x1 m subplots were censused. These changes in sampling intensity may yield increasing species counts from 1990 to 1991 and from 1997 to 1998, but the comparisons between exclosure and control would still be valid.

The Coefficient of Community (CC) was used to compare species composition between exclosure and control for each plot. This is calculated

$$2C/(A+B) \times 100,$$

where C = the number of species in common, A = the number of species in the exclosure, and B = the number of species in the control. If the exclosure and the control were identical, the CC would be 100; if they were completely different it would be 0.

In some plots in some years there were unknown species, or species identified only to genus (or family in the case of grasses). These were all included in the total number of species in a plot and in the number of species in common. For the distinction between native and non-native, only identified species were included, except that all unidentified species of Rubus, Viola, and Galium were assumed to be native. Data from 1988 were not included in the analyses because problems of species identification were more severe than in other years. It is unlikely that there were significant changes between 1988 and 1989 that would affect the interpretation. Throughout the project there were questions of species identifications; it may be appropriate in the future to collect individuals from near the plots as voucher specimens for identification.

To illustrate the number of native species and their cover, numbers were graphed against time, with one line each indicating the total number of native species in the control and in the exclosure, and one each for the number of native species with cover greater than 5% in any subplot. A similar plot was prepared for non-native species.

RESULTS

By 1994 the cover of multiflora rose (Rosa multiflora) and, to a somewhat lesser extent, Japanese barberry (Berberis thunbergii) had become so dense in exclosure #3 in the open field that it was no longer feasible to enter the plot to sample the vegetation. By 2001 it was also not feasible to enter exclosures #2 and #4 because of the dense growth of the same two shrubs. Only a small number of subplots had been censused in these exclosures 2000 because of the density of the shrubs. The data comparisons below will thus not include plot #3, but will include # 2 and #4 through the year 1999.

For the four plots for which the CC was calculated, the similarity in species decreased after a few years, from 50-60% of the species being shared down to 10-30% (Figure 5). In the subsequent several years, however, the similarity recovered to a greater or lesser extent. This variability is to be expected, as some species are rare or appear for a few years in the exclosure or the control plot. There is a suggestion, however, that the difference is robust in plot 5, located in oak-dominated forest, where palatable species such as woodland aster (Aster divaricatus) and blueberry (Vaccinium angustifolium) became more consistently present in the exclosure than in the control plots. That the recovery is slow is not surprising, as several studies have shown that woodland flora is slow to recolonize secondary forests, perhaps because of lack of propagules (Whitney and Foster 1988, Russell 1997 pp. 41-42). Similar problems would probably occur in recolonizing areas where the species have been eliminated by browsing. Blueberry, for example, rarely fruits in this forest, so its recolonization of the exclosure plots would be expected to be slow.

In at least two of the plots (#1 and #4), however, the number of native species with >5% cover increased in the exclosures compared with the control, browsed plots (Figure 6), perhaps because vegetative propagules were already present in the soil. In #4 the consistency of the data for the last three years of data collection is distinctive, with 6 native species (enchanter's nightshade (Circaea quadrisulcata), American ash, spicebush (Lindera benzoin), Virginia creeper (Parthenocissus quinquefolia), jumpseed (Polygonum virginiana) and violets (Viola spp.)) with >5% cover in the exclosure and only 1 (Cardamine pensylvanica – possibly the non-native weed, C. impatiens) in the control. While the number of native species is consistently higher in the exclosures than in the controls (except for 2000 AD in plot #2), it was higher at the start of the experiment, so apparent differences in species composition between exclosures and control plots do not appear to be related to deer browsing. This is the kind of anomalous result that would most likely require a large number of plots for statistical analysis because of the difficulties of establishing comparable experimental and control plots.

Common increasers in cover and numbers in the exclosures have been the shrubs, multiflora rose (Rosa multiflora), brambles (Rubus spp.) (especially the non-native Rubus phoenicolasius), and, to some extent, Japanese barberry. The non-native vine, Japanese honeysuckle (Lonicera japonica), and to some extent in the last few years, oriental bittersweet (Celastrus orbiculatus), have also increased more in the exclosures than in the control plots. The native vine, Virginia creeper (Parthenocissus quinquefolia), is now much more common in the exclosures than in the control plots, as are the native herbs,

woodland aster, bellwort (Uvularia spp.) and enchanter's nightshade. Seedlings marked in Exclosure #1 that have survived since 1995 include 6 maple-leaved viburnum (Viburnum acerifolium), 8 spicebush, 2 sassafras (Sassafras albidum), 1 hickory, 1 black locust, and 1 tree-of-heaven (Ailanthus altissima).

The most notable decreaser in the exclosures is Microstegium vimineum, which decreased from 80-100% cover when the plots were set up in plots 2, 3 and 4, to just a trace in the exclosures by the mid-1990's, while remaining as important as ever in the control plots. The other major decreaser in the exclosures was white snakeroot, a native herb that is poisonous to deer. Meanwhile, in the control plots over this time period Cardamine pensylvanica, garlic mustard and a non-native species of Oxalis* increased, as did barberry in control plot 1. It is likely that the plant identified as C. pensylvanica is the non-native weed C. impatiens.

DISCUSSION

The most salient effect of excluding deer is the large increase in palatable, or even only slightly palatable, spiny shrubs in open or young successional stands. In the local area, these consist mainly of non-native shrubs that deer may be keeping under control. It will be worthwhile to continue monitoring these plots from the outside to discern whether trees are able to survive under the shrubs and eventually to overtop them.

The data suggest that preventing deer browsing may encourage an increase in the cover of native plant species, but that the response is neither consistent nor quick. However, it is apparent that some native seedlings that do not survive outside the exclosures are able to survive for longer in the exclosures. Of interest also is the decrease in some unpalatable species such as white snakeroot and Microstegium vimineum in the exclosures. Further research is needed to identify the factors that lead to this result.

WOODY SEEDLING STUDY

The origin of almost all trees in the forests of the park is from seed reproduction. It seemed appropriate, therefore, to assess the production and viability of woody seedlings. The spring and early summer of 1994 were excellent years for the production of oak seedlings, a very sporadic occurrence. This led to establishing plots to track the fate of these seedlings as well as others that appear.

METHODS

In 1994, I established 14 3x3 m permanently marked plots randomly arranged on a straight, 530 m long, line between Cemetery Road and Lewis Morris County Park,

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* The cosmopolitan weed O. stricta may be native to North America. It is difficult to distinguish from the non-native O. europea (Gleason and Cronquist 1991)

running north from the Maintenance Road and ending at the boundary between the National and County parks. The plots were set up in 1994 in response to the high numbers of oak seedlings that spring. In each plot, I identified and counted all woody (shrub and tree) seedlings. The counts have been repeated 1-4 times each summer, from 1994 until 2001. An attempt was made for each seedling to establish whether it was new that year or was more than a year old, i.e., whether it had survived for a year or more. To this end, a colored toothpick was used to mark each seedling, with colors coded to years. This was not completely successful, as many toothpicks were buried under the leaf litter and moved by small animals. The remaining markers have, however, provided some means of tracking seedlings from one year to the next.

RESULTS

Oak was the most highly variable seedling producer. There have been no such excellent years for oaks since 1994, and all oak seedlings from that year subsequently died, most in their first year (Figures 7-10). Over half the oak seedlings were red oak (Q. rubra), with chestnut oak (Q. prinus) and black oak (Q. velutina) also common (15 and 19%), and a few white oaks (Q. alba) (6%). In contrast, the proportions of oak trees growing within 10 meters of the center of each plot were approximately 1/3 each white, chestnut and red with slightly fewer black oaks (Figure 11).

All major trees in the local forest except beech are represented in the seedlings: oaks, hickories, red maple, black birch, tulip tree, ash. The distribution of these taxa both spatially and temporally (from year to year) varies greatly, so that counting seedlings one year and in a few lots may give a very skewed result. In one year, for example, there was very abundant germination of black birch, such that they were practically uncountable. The large majority of these seedlings did not survive to the next year. Seeds also germinated over the summer for some species, especially tulip tree.

Seedlings that survived here for more than one year were generally still very small. Some red maple and black birch seedlings that were 3-4 years old were only about 2-3 cm long, with much of the length being stem lying just under the litter layer. Insect damage to seedlings was ubiquitous, including insects that ate in from the margins of leaves, holes in the leaves, leaf rollers and leaf miners. Most leaves with such damage appeared still healthy, though there were a few that had been eaten down to the midrib. The consequences for survival cannot be determined from these data.

There was no damage apparently caused by small mammals such as mice, voles and rabbits. Lower stems were all intact. It may be that the seedlings never became large enough to attract such herbivores. There was no evidence that deer browsed seedlings other than oak and hickory, but none of these seedlings has become successfully established, growing larger than a few centimeters, over the 8 years of this study.

The oak and hickory seedlings were almost all browsed down to the terminal bud, and many sprouted again two or more times. It is likely that deer caused such damage, though there were also oak seedlings in the exclosures that had apparently similar damage. Some seedlings had leaves that appeared to have been eaten back to the midrib,

most likely by insects, which would have thriven with the copious numbers of oak seedlings in 1994, dropping from the mature trees to feast on the tender foliage of the seedlings. One hickory seedling hidden in a mat of Japanese wire grass grew to the top of the mat, about 20 cm, suggesting that being under it protected the seedling from predation. It had disappeared by the next year, however.

DISCUSSION

Almost all tree species present in the forest produce propagules, which are able to germinate and establish seedlings in some years. Many produce abundant seedlings. However, most of the seedlings do not survive for more than 2-3 years, and those that do survive almost never become large even in 2-3 years. The causes of the death of the seedlings have not been determined.

Beech is the main exception to the production of seedlings, but other data indicate that it is nevertheless spreading in the forests, by the establishment of occasional seedlings and further spread by vegetative means. White oaks are producing fewer seedlings than their proportion in the forest, while red oaks are producing more. Thus the apparent shift from white toward red oaks is at least partly related to seed production and success, as well as possible problems with survival after seedling development.

"DEAD TREE" TRANSECTS

Replacement of dead trees was studied in two 20 m wide transects. Data from these transects suggests patterns of mortality of trees in the park.

METHODS

The starting point of each transect was randomly located along a stretch of the tour road. The first, called the mature forest transect, extended 160°N through forest mapped as mature (mostly oak-dominated) forest in 1977. To provide a contrast, the second was placed in a mostly yellow poplar-dominated stand, between the maintenance area and the parking lot and comfort station. This was referred to as the successional forest transect. The mature forest Transect was 500 m long and the successional forest 250 m.

For each transect, any dead trees were noted by how far along the transect they were and on which side of the midline they were located. Dead trees were any that were either standing dead or had >50% of the crown dead, or down. Stumps with no visible trunk on the ground were not counted. For each tree an attempt was made to identify it and to estimate or measure its diameter at approximately 1-2 m from the ground. Only trees that would probably have been part of the canopy before they died were included in the tally. A record was made of any tree (>/= 10 cm dbh) or sapling (<10 cm dbh, >1 m tall) within 5 m of the center of the dead tree. This would include any smaller trees that would be expected to fill the canopy gap left by the dead tree.

In addition, to characterize the forest through which the transects ran, every 50 m, a point-centered quarters sample of the surrounding forest vegetation was made. To do this, a point was marked on the midline and the species, dbh (for trees) and distance from the point were recorded for the closest tree and the closest sapling in each quadrant around the point. This provided an estimate of the density of trees and saplings, species composition, and basal area.

RESULTS

The transect in the mature forest was 500 m long, giving an area of 1 ha. We found 53 dead trees in this area. Of these, 22 resulted from tip-up, where the roots had been pulled out of the soil as the tree fell. All of these trunks were rotted beyond recognition. The remaining dead trees had been broken off or were still standing. Most of these were red maple or black birch. The average diameters of the trees within 5 m of the dead trees and of the dead trees that I could measure were about 24 cm. The scatter plot of tree diameters, however, illustrates that the live trees included more larger trees than the dead ones (Figure 12). There were more saplings than trees within 5 m of the dead trees in this area.

The transect in the yellow poplar forest was 250 m long, giving an area of 0.5 ha. We found 27 dead trees in this area, giving about the same density of dead trees as in the mature forest. Of these, 8 were tip-ups, and, as in the mature forest, almost all of these were rotted beyond recognition. The average dbh of the trees within 5 m of the dead trees in this area was 32 cm, while the diameter of the dead trees was 39 cm. There were fewer trees within 5 m of the dead stumps than in the mature forest, and an average of 0 saplings. There were no trees or saplings within 5 m of 6 of the 27 dead trees, and 19 had no saplings.

The point-centered-quarters data indicate that there are about as many saplings/ha as trees in the more mature forest (distance to nearest tree is 5.1 m and to nearest sapling is 6.4 m). The largest trees were 3 oaks and a yellow poplar. Of the trees within 5 m of dead trees in this transect, 8 of the 22 trees >40 cm dbh were oaks. However, beeches predominated in the sapling layer, suggesting a shift from oak to beech in the future.

The situation was quite different in the successional forest. The closest sapling was twice as far from the point as the closest tree in the point-centered-quarters sampling of this forest (13.9 m as compared with 6.6 m). This is consistent with the very low numbers of saplings within 5 m of the dead trees. The most numerous trees were yellow poplar, which were also the largest trees, while red maple was the most common sapling.

CONCLUSIONS

These results suggest that while there has been adequate replacement of dead trees in the mature forest, the situation in the successional forest is not as good. The lack of saplings and low number of trees within 5 m of the dead trees is disturbing. The average size of the trees is larger, but there are fewer saplings, suggesting that as the large trees die, there will be no native saplings or smaller trees to replace them. This condition will encourage the spread of light-tolerant, unpalatable, generally non-native shrubs and herbs, and probably unpalatable non-native trees, such as tree-of-heaven (Ailanthus altissima) as well. The data also are consistent with a shift from oak dominance to beech dominance in the mature forest.

OVERALL CONCLUSIONS

1. Forest trees in the Jockey Hollow section of the park are continuing to mature, increasing in average basal area as they thin.

2. Sapling regeneration is abundant only where beech is common. It is especially thin in many successional stands, where yellow poplar or black locust dominates the tree layer.

3. Saplings are less diverse than forest tree species, even where saplings are fairly abundant, and suggest future changes in the species composition of the forests.

4. Most of the native tree species are reproducing sporadically by seed.

5. Tree seedling growth is generally poor, whether the seedlings are browsed by deer or not. The causes of this poor growth are as yet undetermined, but may include soil chemistry and shade. In no part of the forest that was studied are there sufficient seedlings surviving to replace saplings as they mature into forest trees.

6. Deer reduce the cover and abundance of many palatable species, both native and non-native.

7. Complete elimination of deer encourages the luxuriant growth of many shrubs, such as multiflora rose and wineberry, in successional stands. In mature forests, with perhaps poorer soils and a more closed canopy, elimination of deer had more subtle effects on increasing cover of native species (McShea and Rappole 2000).

8. Non-native shrubs such as Japanese barberry may in some cases substitute for native shrubs more palatable to deer as nesting sites for birds (study by Steve Gates, 1999).

9. The distribution of Japanese barberry and Japanese wire grass appears to have remained stable in the sampled areas of the forest from 1995 to 2001, though the barberry has become denser over that time period. The distinction between "invaded" and "uninvaded" plots has remained constant.

10. Invaded plots are more likely to be in areas with less oak and beech and more yellow poplar and black locust than uninvaded plots (Kourtev et al. 1998). Yellow poplar and black locust are characteristic of areas that have been cleared sometime in the past although prior agriculture is not documented in all of these stands (Ehrenfeld 1977, Russell and Schuyler 1988). Invaded sites also have higher pH than uninvaded sites (Kourtev et al. 1998), suggesting that these better soils were chosen for more intensive land use in the past.

11. There is a complex interaction between the spread of unpalatable species and deer browsing. Elimination of deer can eliminate or significantly reduce such species, both native, e.g., white snakeroot, and non-native, e.g., Japanese wire grass. The reasons for this are yet to be elucidated.

12. Some successional stands are almost completely lacking in regeneration of trees, so that as trees die they are not being replaced.

IMPLICATIONS FOR THE PARK

While superficially many parts of the forests of MNHP appear healthy, there is abundant evidence that the future is unpredictable based on current regeneration. Most critical are the secondary successional stands where yellow poplar or black locust dominate. In these forests there are almost no saplings. The future of these forests is in jeopardy, probably in the near (decadal scale) future. The causes of the lack of

regeneration are most likely complex, and reversing the problem will require consultation with foresters familiar with similar stands.

In more mature forests, the only abundant saplings are beeches, which occur only in some stands. There are, however, also red maple and black birch saplings. There is a lack of seedlings that grow vigorously or survive for more than a couple of years. The problem here is less critical than in the successional stands, but should be monitored carefully and perhaps experimental management instituted to find out why, other than deer browsing, most seedlings do not survive. Soil chemistry is a potential factor.

The shrub layer of the vegetation presents a complex issue. It appears that over the last 2-3 decades, deer have nearly eliminated most palatable shrubs. It is possible that this lack of shrub understory is appropriate for the historical period, when free-ranging livestock would have decimated the palatable plants. However, non-native shrubs such as Japanese barberry have replaced the native shrubs in many areas, growing in very dense thickets, creating a distinctly an ahistorical forest appearance. The monitoring data suggest that there are underlying reasons why barberry and other invasive non-native species are restricted to some parts of the park, and these reasons should be elucidated to help in control of these shrubs.

Removal of deer has both positive and negative impacts on the understory vegetation of the park. On the positive side, it allows native shrubs and herbs, and possibly tree seedlings, to survive in some areas. On the negative side, it also allows vigorous growth of non-native spiny shrubs such as multiflora rose. Even Japanese barberry does better without deer browsing. It is likely thus that a reduction in deer herd would allow a proliferation of even more non-native shrubs than are already present at the park.

Changes over the last 5-15 years indicate that to establish and maintain a health forest at the park, immediate experimental intervention is important for many of the successional forests. This might include the establishment of larger exclosures, soil studies, and other management options. Testing these options over the next 5-10 years would provide data on which to base more extensive treatments. For example, decreases in the deer herd at least locally and for short periods may be necessary to encourage the reestablishment of healthy seedling growth, but other factors must also be investigated. Complete elimination of deer does appear to be allowing the establishment of native tree, shrub and herbaceous species in some areas.

Understanding the factors that lead to establishment of Japanese barberry and Japanese wiregrass (as well as other invasive species) in some areas rather than others may help in controlling their distribution.

Some of the changes observed over the last few years are consistent with the continuation of a healthy, ecologically integrated forest, primarily in the older forest areas. On the other hand, changes observed in the secondary successional forests appear to be tending toward an ecologically impoverished forest, which will probably lack not

only the species, but also the structure of the vegetation of the 18th century. It may be necessary to intervene to reverse these trends.

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Figure 2. Numbers of trees in permanent vegetation monitoring plots in 2001.

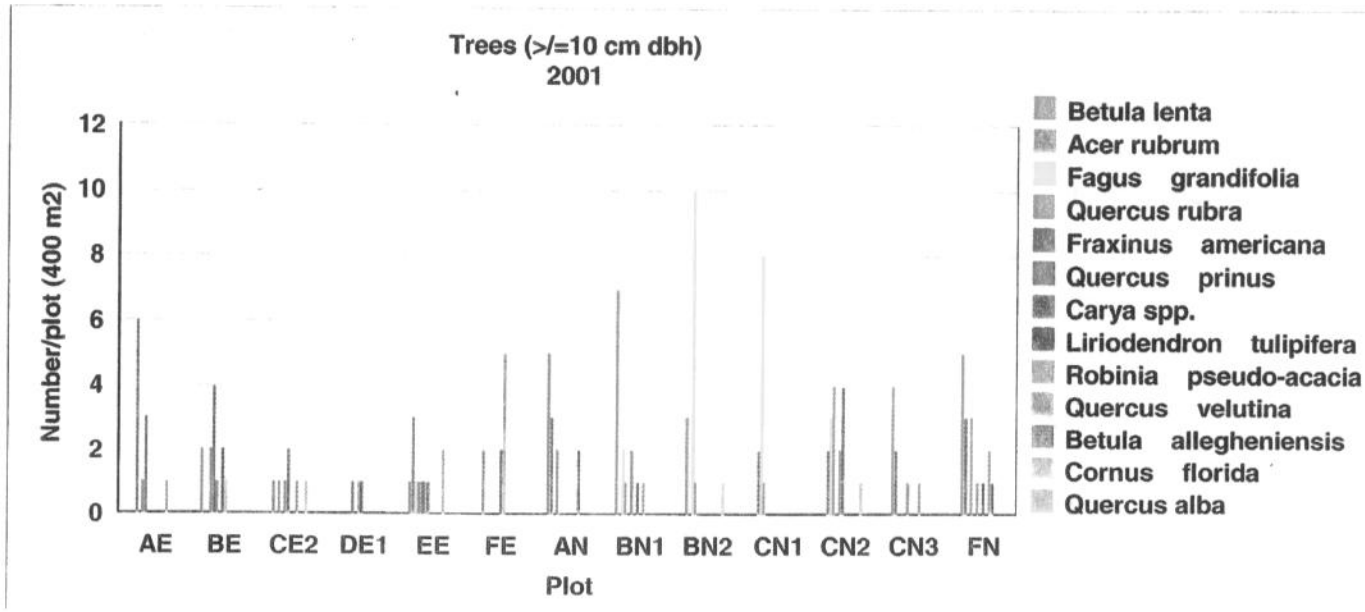


Figure 3. Tree sizes (dbh) in permanent vegetation monitoring plots in 2001.

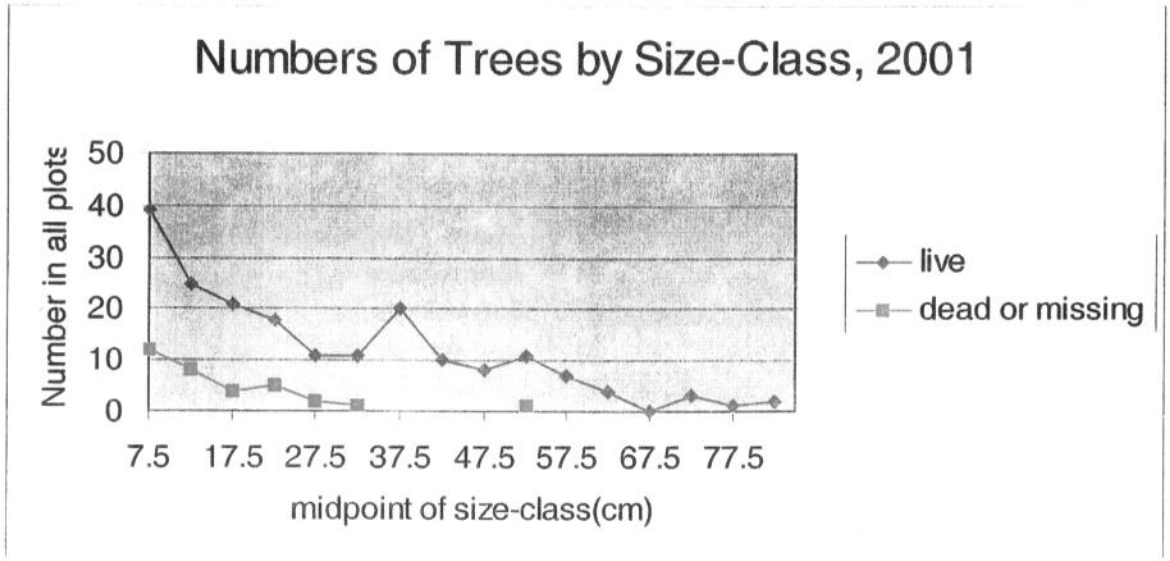
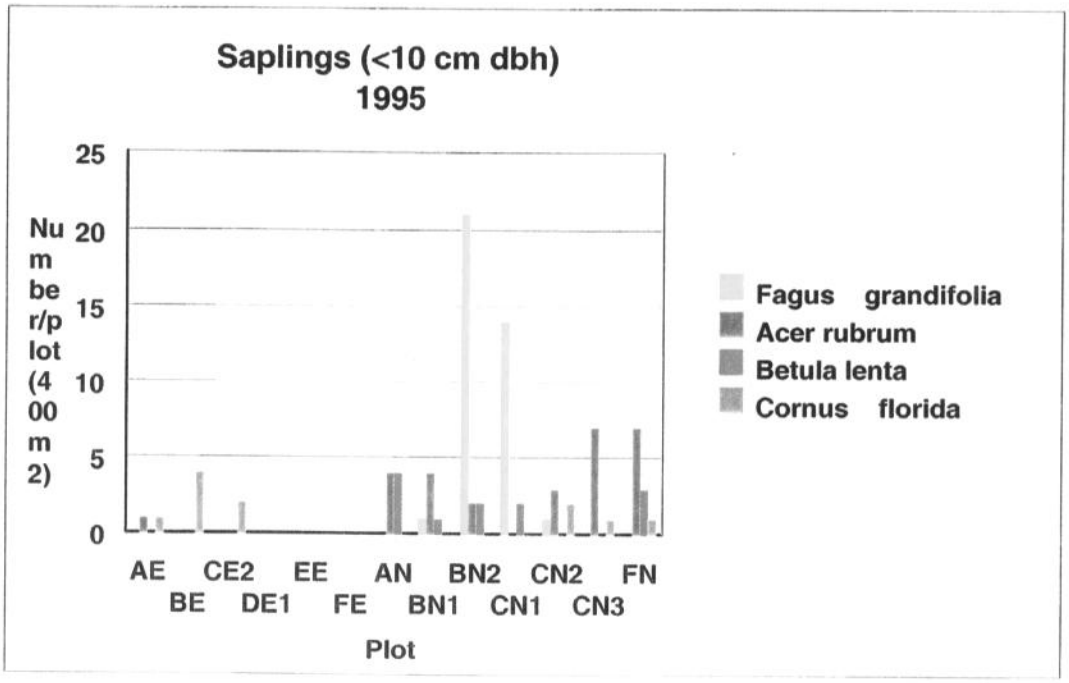


Figure 4. Sapling data for permanent vegetation monitoring plots.

4a. Data for 1995



4b. Data for 2001

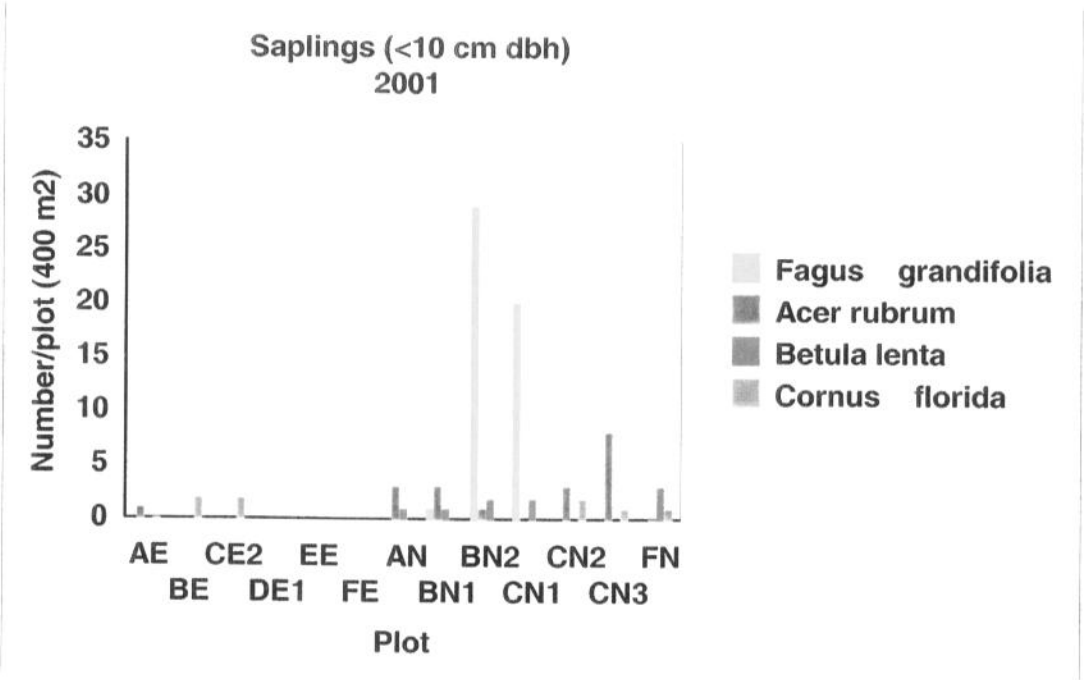


Figure 5. Community coefficients for deer exclosures. CC = 2C/(A + B), where A = number of species in exclosure, B = number of species in control and C = number of species in common

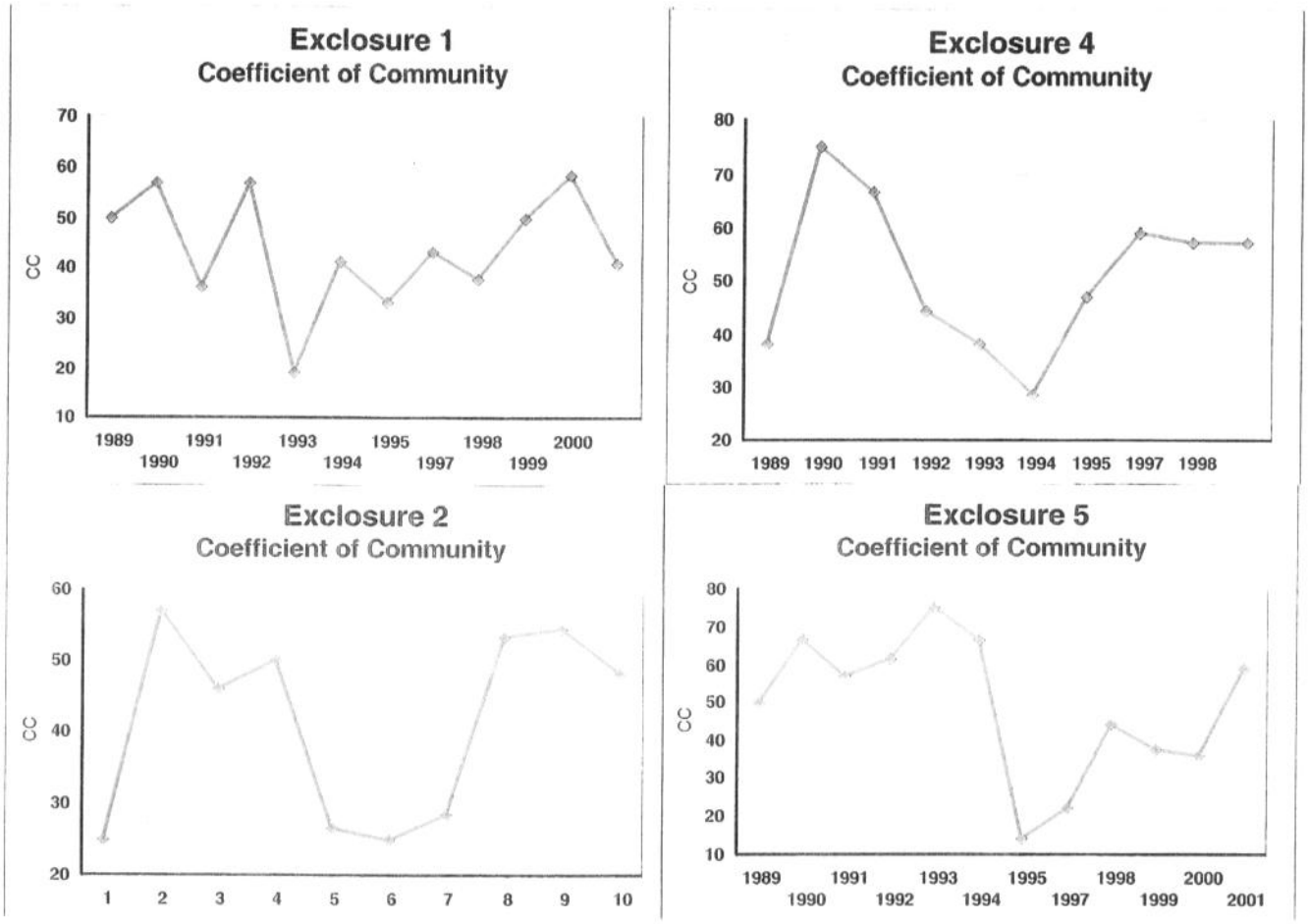


Figure 6. Native species in exclosures and control plots. Data expressed as total number of species in each and number of species with greater than 5% cover in any subplot.

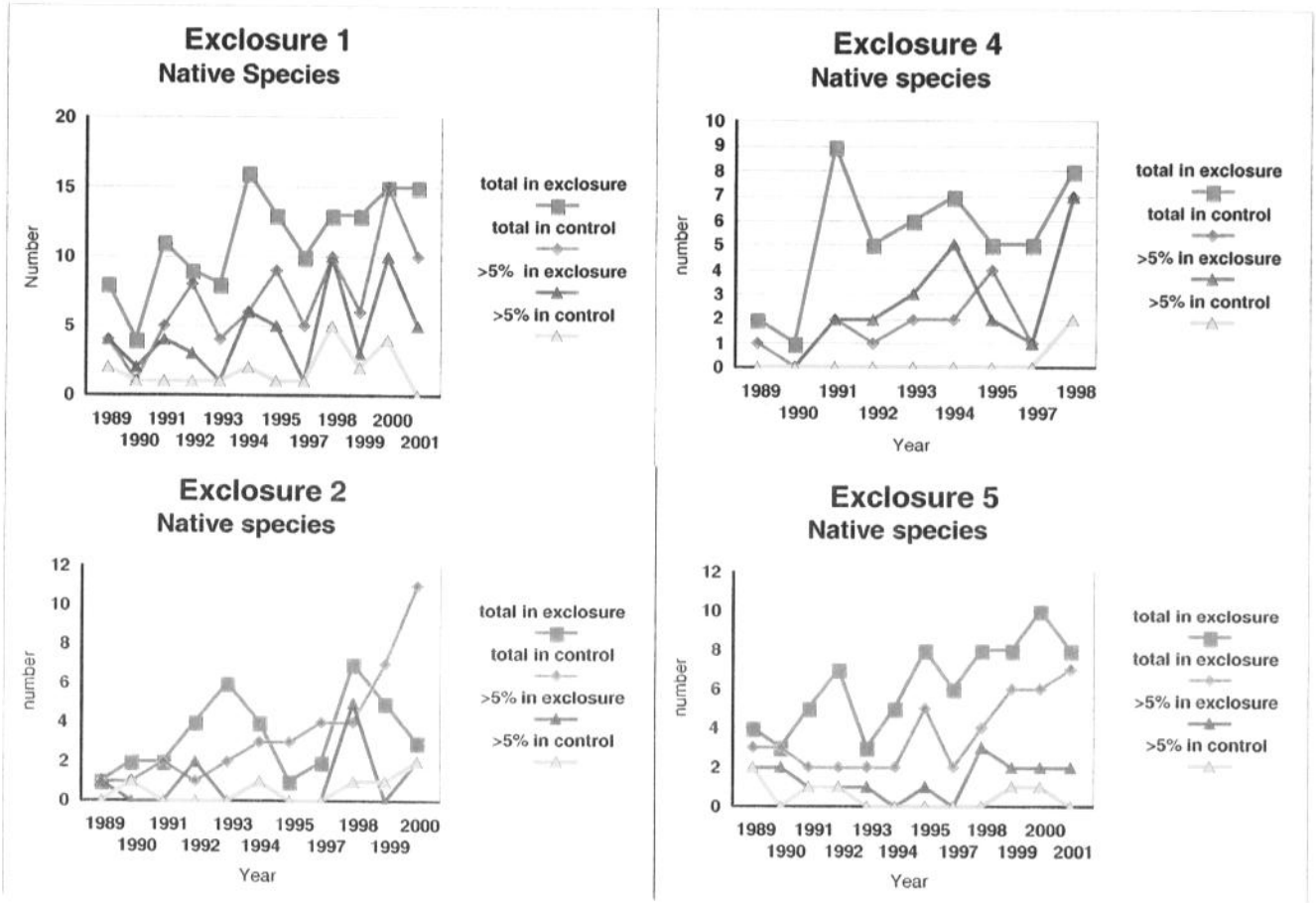


Table 1. Numbers of trees and saplings in permanent vegetation monitoring plots in 1995.
Plots with codes ending in N are not invaded by exotics; those ending in E have exotics.

TREES > /= 10 CM DBH

PLOT	AE	BE	CE2	DE1	EE	FE	AN	BN1	BN2	CN1	CN2	CN3	FN	total	CE1	NJE	DN1	EN	NJN	TOTAL
SPECIES																				
Betula lenta		3			1		5	7	3			4	7	30		7		2	2	41
Acer rubrum	7		2		3	2	4	1	1	3	2	2	1	28	6			4	6	44
Fraxinus americana	3	6	2	5	1			1					4	22		3	4			29
Fagus grandifolia					1			1	10	8	2			22		1		2	5	30
Quercus rubra	1	1			1		2	1	1	1	4		3	15		1	6			22
Cornus florida	1		3		2						2			8	4	1	1			14
Quercus prinus		1	1		1			2			2	1	1	9		1		2		12
Carya spp.			2	1	1						4			8					1	9
Liriodendron tulipifera		2		1		2		1					1	7				1		8
Robinia pseudo-acacia		1				6								7						7
Quercus velutina			1					1				2	1	5		1				6
Betula allegheniensis							2						1	3				4		7
Quercus alba			1						1		1			3	1			1	1	6
Prunus serotina				1		1								2						2
Ulmus americana	1			1										2					1	3
Acer saccharum		1												1				1		2
Sassafras albidum				1										1						1
Prunus avium																	3			3
Nyssa sylvatica																		2		2
TOTAL	13	15	12	10	11	11	13	15	16	12	17	9	19	173	11	15	14	19	16	248

SAPLINGS < 10 CM DBH

PLOT	AE	BE	CE2	DE1	EE	FE	AN	BN1	BN2	CN1	CN2	CN3	FN	total	CE1	NJE	DN1	EN	NJN	TOTAL
SPECIES																				
Fagus grandifolia								1	21	14	1			37		1		8	5	51
Acer rubrum	1						4	4	2		3	7	7	24		2				26
Betula lenta							4	1	2	2			3	11	1	2			1	15
Cornus florida	1	4	2								2	1	1	11	1	1	2	3		18
Carya spp.								2				1		3			2			5
Amelanchier arborea									1					1				2		3
Betula allegheniensis							1							1				6		7
Castanea dentata									1					1						1
Liriodendron tulipifera													1	1				1		2
Prunus avium		1												1						1
Quercus rubra		1												1				1		2
Ulmus americana	1													1						1
Fraxinus americana																	5			5
Quercus velutina															1					1
Nyssa sylvatica																		3		3
Juniperus virginiana															1					1
DEAD	9			1		1	4	3	1	1	1		1	22	1	1	6			30
TOTAL ALIVE	3	6	2	0	0	0	9	8	27	16	6	9	7	93	4	6	9	24	6	142

Table 2. Numbers of trees and saplings in permanent vegetation monitoring plots in 2001.
Plots with codes ending in N are not invaded by exotics; those ending in E have exotics.

TREES > /= 10 CM DBH

PLOT	AE	AN	BE	BN1	BN2	CE2	CN1	CN2	CN3	DE1	EE	FE	FN	total
SPECIES														
Betula lenta		5	3	7	3				4		1		7	30
Acer rubrum	7	4		1	1	2	3	2	2		3	2	1	28
Fraxinus americana	3		6	1		2				5	1		4	22
Fagus grandifolia				1	10		8	2			1			22
Quercus rubra	1	2	1	1	1		1	4			1		3	15
Cornus florida	1					3		2			2			8
Quercus prinus			1	2		1		2	1		1		1	9
Carya spp.						2		4		1	1			8
Liriodendron tulipifera			2	1						1		2	1	7
Robinia pseudo-acacia			1									6		7
Quercus velutina				1		1			2				1	5
Betula allegheniensis		2											1	3
Quercus alba					1	1		1						3
Prunus serotina										1		1		2
Ulmus americana	1									1				2
Acer saccharum			1											1
Sassafras albidum										1				1
TOTAL	13	13	15	15	16	12	12	17	9	10	11	11	19	173

SAPLINGS < 10 CM DBH

PLOT	AE	AN	BE	BN1	BN2	CE2	CN1	CN2	CN3	DE1	EE	FE	FN	total
SPECIES														
Fagus grandifolia				1	21		14	1						37
Acer rubrum	1	4		4	2			3	7				7	24
Betula lenta		4		1	2		2						3	11
Cornus florida	1		4			2		2	1				1	11
Carya spp.				2					1					3
Amelanchier arborea					1									1
Betula allegheniensis		1												1
Castanea dentata					1									1
Liriodendron tulipifera													1	1
Prunus avium			1											1
Quercus rubra			1											1
Ulmus americana	1													1
DEAD	9	4		3	1		1	1		1		1	1	22
TOTAL ALIVE	3	9	6	8	27	2	16	6	9	0	0	0	12	93

Table 3. Shrub data from permanent vegetation monitoring plots, for invaded plots.

Changes 1995 to 2001 (by subplot =25 m2)

		BARBERRY			RUBUS			# NON-NATIVE SPP (EXCL BARBERRY)		# NATIVE SPP		NATIVE SPECIES STEMS	
		1995	2001	2001-1995	1995	2001	2001-1995	1995	2001	1995	2001	1995	2001
PLOT	SUBPLOT												
AE	NE	0	0	0				0	0	0	0	0	0
	NW	0	0	0				0	1	1	0	14	0
	SE	27	92	65				0	2	1	4	1	4
	SW	19	32	13				0	2	1	2	64	2
BE	NE	78	106	28				0	0	0	0	0	0
	NW	17	26	9				0	0	0	0	0	0
	SE	93	122	29				0	0	0	0	0	0
	SW	10	39	29				0	1	0	4	0	15
CE2	NE	34	110	76				0	1	1	0	2	0
	NW	13	42	29				0	2	0	1	0	6
	SE	1	59	58	1	4	3	1	2	1	0	84	0
	SW	6	87	81		2	2	0	2	0	4	0	4
DE	NE	0	0	0	6	74	68	0	3	0	1	0	10
	NW	16	30	14	1	21	20	2	3	0	2	0	6
	SE	0	0	0	15	66	51	0	2	0	4	0	8
	SW	6	28	22	7	46	39	0	3	0	1	0	1
EE	NE	79	104	25	2		-2	2	1	1	7	25	20
	NW	53	96	43	8	8	0	1	1	0	6	0	10
	SE	5	7	2		4	4	0	2	1	10	32	31
	SW	8	0	-8		2	2	0	2	0	0	0	0
FE	NE	63	255	192				0	3	0	1	0	1
	NW	177	324	147				0	2	0	1	0	2
	SE	125	250	125				0	2	0	1	0	1
	SW	31	102	71				0	2	0	2	0	3

1995	SPECIES 2001
X	RUBUS PHOENICOLASIUS
	EUONYMUS ALATUS
X	CELASTRUS ORBICULATUS
	AILANTHUS ALTISSIMA
	ROSA MULTIFLORA
	LIGUSTRUM VULGARE

1995	SPECIES 2001
	ACER RUBRUM
	ACER SACCHARUM
X	BETULA SP
X	CARYA SP
	CORNUS FLORIDA
X	FAGUS GRANDIFOLIA
	FRAXINUS SP
X	HAMAMELIS VIRGINIANA
X	LINDERA BENZOIN
	LIRIODENDRON TULIPIFERA
	NYSSA SYLVATICA
	PARTHENOCISSUS QUINQUEFOLIA
	PRUNUS SP
	QUERCUS PRINUS
	QUERCUS RUBRA
C	QUERCUS SP
	RHODODENDRON PERICLYMENOIDES
	RHUS SP
	SASSAFRAS ALBIDUM
	TOXICODENDRON RADICANS
	VACCINIUM SP
	VIBURNUM PRUNIFOLIUM
	VITIS SP

Table 4. Shrub data from permanent vegetation monitoring plots, for uninvaded plots.

		BARBERRY			# NON-NATIVE SPP (EXCL BARBERRY)		# NATIVE SPP		NATIVE SPECIES STEMS	
PLOT	SUBPLOT	1995	2001	2001-1995	1995	2001	1995	2001	1995	2001
AN	NE	0	0	0	0	0	3	1	9	7
	NW	0	0	0	0	0	2	0	15	0
	SE	0	0	0	0	0	1	0	75	0
	SW	0	0	0	0	0	1	2	69	43
BN1	NE	0	0	0	0	0	4	0	4	0
	NW	3	6	3	0	0	1	4	6	6
	SE	8	10	2	0	0	2	5	15	16
	SW	0	0	0	0	0	3	0	25	0
BN2	NE	0	0	0	0	0	1	1	17	23
	NW	0	0	0	0	0	2	1	297	108
	SE	0	0	0	0	0	1	3	89	68
	SW	0	0	0	0	0	3	1	55	7
CN1	NE	0	0	0	0	0	1	1	8	4
	NW	3	0	-3	0	0	1	0	141	0
	SE	0	0	0	0	0	1	1	2	11
	SW	0	2	2	0	0	1	1	83	36
CN2	NE	0	0	0	0	0	1	2	2	12
	NW	0	0	0	0	0	1	1	9	5
	SE	0	0	0	0	0	2	1	42	4
	SW	0	0	0	0	0	3	1	6	2
CN3	NE	0	0	0	0	1	0	1	0	1
	NW	0	0	0	0	0	0	1	0	1
	SE	0	0	0	0	0	0	1	0	1
	SW	0	0	0	0	0	0	0	0	0
FN	NE	0	0	0	0	0	0	0	0	0
	NW	0	0	0	0	0	0	2	0	2
	SE	0	0	0	0	0	1	1	7	7
	SW	0	0	0	0	1	1	2	26	3

SPECIES	1995	2001
CELASTRUS ORBICULATUS		
EUONYMUS ALATUS		

SPECIES	1995	2001
ACER RUBRUM		
ACER SACCHARUM	X	
BETULA LENTA	X	
CARYA SP	X	
FAGUS GRANDIFOLIA	X	
FRAXINUS SP	X	
HAMAMELIS VIRGINIANA	X	
LIRIODENDRON TULIPIFERA		
NYSSA SYLVATICA		
QUERCUS PRINUS		
QUERCUS SP	X	
VACCINIUM SP		
AMELANCHIER ARBOREA		

Table 5. Herbaceous species in permanent vegetation monitoring plots.

Native herb layer species in invaded plots	species
1995	"fern"
	Carex
	Eupatorium rugosum
	Quercus rubra
2001	Acer rubrum
	Arisaema triphyllum
	Brassicaceae
	Carex
	Carex 1
	Carex 2
	Eupatorium rugosum
	Liriodendron tulipifera
	Rhus sp.
	Vitis sp.

Native herb layer species in uninvaded plots	species
1995	"fern" = Polystichum?
	Carex
	Eupatorium rugosum
	Mitchella repens
	MYAN ?
2001	Acer rubrum
	Acer saccharum
	Betula lenta
	Carex
	Chimophila maculata
	Epifagus virginiana
	Eupatorium rugosum
	Liriodendron tulipifera
	Mitchella repens
	Poaceae
	Polystichum acrostichoides
	Quercus sp
	Vitis sp.

Non-native species in invaded plots Not including Microstegium (in all)	species
1995	Alliaria officinalis (probably)
	Lonicera
2001	Alliaria officinalis
	Allium vineale
	Euonymus alata
	Lonicera
	Oxalis

Non-native species in uninvaded plots Not including Microstegium (trace in two plots)
none

As the nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally owned public lands and natural and cultural resources. This includes fostering wise use of our land and water resources, protecting our fish and wildlife, preserving the environmental and cultural values of our national parks and historical places, and providing for enjoyment of life through outdoor recreation. The department assesses our energy and mineral resources and works to ensure that their development is in the best interests of all our people. The department also promotes the goals of the Take Pride in America campaign by encouraging stewardship and citizen responsibility of the public lands and promoting citizen participation in their care. The department also has a major responsibility for American Indian reservation communities and for people who live in island territories under U.S. administration.

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