Restoration Notes

Restoration Notes have been a distinguishing feature of *Ecological Restoration* for more than 25 years. This section is geared toward introducing innovative research, tools, technologies, programs, and ideas, as well as providing short-term research results and updates on ongoing efforts. Please direct submissions and inquiries to the editorial staff (mingram@wisc.edu and cmreyes@wisc.edu).

An Integrated Approach to Mapping Forest Conditions in the Southern Appalachians (North Carolina)

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Accurate and continuous forest cover information is essential for forest management and restoration (SAMAB 1996, Xi et al. 2007). Ground-truthed, spatially explicit forest data, however, are often limited to federally managed land or large-scale commercial forestry operations where forest inventories are regularly collected. Moreover, such areas are often fragmented by parcels of privately owned forest for which reliable data are unavailable. As a result, analyses and models that rely on these data face a trade-off between using large, ecologically meaningful spatial scales with missing data, or choosing less relevant spatial scales dictated by the availability and completeness of data.

We describe a method to predict missing forest cover data in order to more completely map large areas. Our aim is to create complete forest cover input data for our modeling work, which is part of a larger effort to investigate the utility of landscape models as a decisionmaking tool. We are working in the southern Appalachian region on predamage impact analysis and postdamage restoration of forest landscapes threatened by a variety of invasive pest species (Cairns et al. 2008, Xi et al. 2008).

Our study area is the Grandfather Ranger District (GRD, ca. 777 km^2) of the Pisgah National Forest in western North Carolina. This mountain region consists of varied habits and high plant diversity (SAMAB 1996). The landscape is characterized by hardwood forests composed of a variety of oaks (*Quercus* spp.), maples (*Acer* spp.), American beech (*Fagus grandifolia*), and yellow poplar (*Liriodendron tulipifera*). On the dry slopes and ridges of this region, yellow pines (*Pinus* subgenus *Pinus*) such as Table Mountain pine (*Pinus pungens*), pitch pine (*Pinus rigida*), shortleaf pine (*Pinus echinata*), and Virginia pine (*Pinus virginiana*) may be dominant. Pine forest covers about 14% of the federally managed GRD lands. Within GRD, substantial data gaps (ca. 43.2%) exist for private and local ownerships (Table 1).

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To predict missing forest cover data in GRD, we first created six layers in a Geographic Information System (GIS) to represent the spatially defined variables: 1) Elevation (m); 2) Aspect (°); 3) Slope (% of rising); 4) Topographic (Wetness) Index ($\ln(a/\tan\beta)$, Beven and Kirkby 1979); 5) NDVI (Normalized Difference Vegetation Index) derived from satellite imagery (Landsat, October 26, 2001); and 6) modeled average daily Shortwave Solar Radiation (Wh/m^2/day) in July (aspect). The first four GIS layers were derived from the DEM (Digital Elevation Model) of the region. The Forest Service's Continuous Inventory of Stand Conditions (CISC) survey dataset (SAMAB 1996) provided information for National Forest lands. Cover types are based on which tree species comprise the main forest canopy (see Table 1 for the classification criteria).

To analyze the relationships between these key continuous environmental variables and forest cover types, and to create a complete landscape map for GRD, we worked in a stepwise manner to 1) group the CISC forest stand data hierarchically into three levels of detail: "major forest," "forest," and "subforest" cover types; 2) perform segmentation based on aspect, which delineated relatively homogenous patches of land as the basic analysis units; 3) train the Classification and Regression Tree (CART) models using the CISC data; 4) perform the predictions; and 5) evaluate the results.

We used a data mining tool (See5, RuleQuest Research) that explores large amounts of data in search of consistent patterns and systematic relationships between variables, and then validates the findings by applying the detected patterns to new subsets of data. The CART algorithm of See5 searches for the dependent variable that splits the population of GIS pixels into two groups. The algorithm explains the largest proportion of deviation of the independent variable and creates a decision tree for predicting variables (i.e., the forest cover types) in areas with missing

Table 1. Forest cover classification as described in the U.S. Forest Service Continuous Inventory of Stand Conditions (CISC) database (SAMAB 1996) for the Grandfather Ranger District of Pisgah National Forest, North Carolina. The percentage of the forest cover type within the District is shown in parentheses. "Major forest" refers to conifer vs. hardwood forests; "forest" consists of pine (≥ 70% basal area of trees with dominant and codominant crowns are softwoods), pine-hardwood (51–69% softwood species), hardwood-pine (51–69% hardwoods), and hardwood types (≥ 70% hardwoods); "subforest" is defined by the groups or taxa that dominate much of the forest canopy.

Major forest	Forest	Subforest
1 Pine forests (14.2%)	11 Pine types (7.2%)	111 White pine (3.3%)
		112 Yellow pine (3.9%)
	12 Pine-Hardwoods (7.0%)	121 Hemlock-hardwoods (1.2%)
		122 White pine/Hardwoods (2.9%)
		123 Yellow pine/Oaks (2.9%)
2 Hardwood forests (42.6%)	21 Hardwood-Pines (12.5%)	211 Cove Hardwoods/Pines (2.7%)
		212 Hardwoods-White pine (2.1%)
		213 Oaks/Yellow pine (7.7%)
	22 Hardwood type (30.1%)	221 Oaks (17.9%)
		222 Yellow poplar/Oaks (12.2%)
0 Data gaps (43.2%)	N/A	N/A

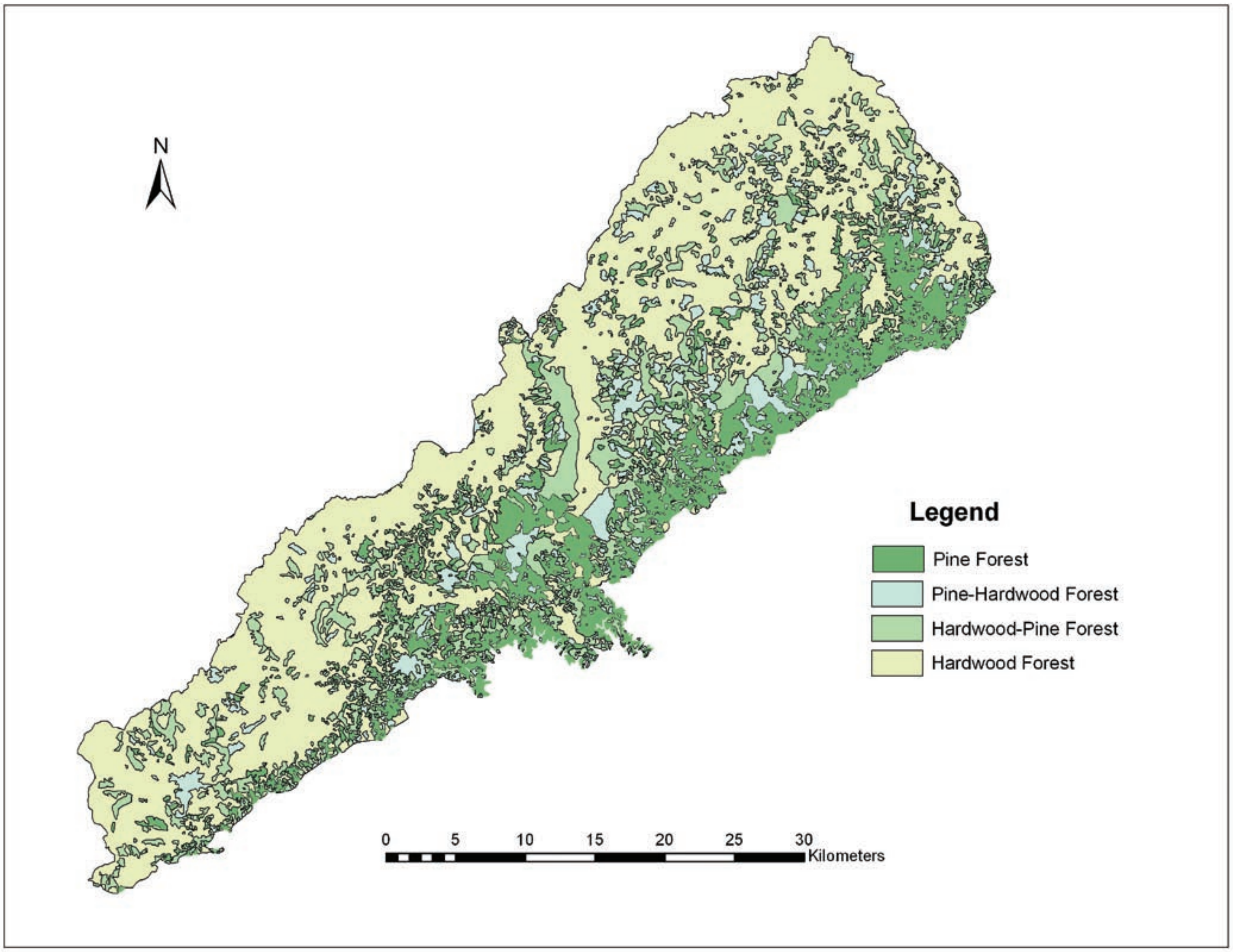

Figure 1. Prediction map of four forest cover types for the Grandfather Ranger District, Pisgah National Forest, North Carolina.

data (Vayssieres et al. 2000). The algorithm has several advantages over traditional classification techniques. In particular, the CART method is strictly nonparametric and does not require assumptions regarding the statistical properties of the input data.

We obtained 77.9% classification accuracy for the “major forest cover types” and 56.7% accuracy for the “forest cover types,” sufficiently reasonable to use the model to interpolate forest cover data. The classification rules generated from CART showed that elevation, slope, and solar radiation were the most important variables for predicting forest cover types in the southern Appalachians. Elevation is the primary determinant of the spatial distribution of tree species in our study area, but NDVI can also be very useful.

This mapping work allows us to generate continuous maps for major forest cover types in the southern Appalachians (Figure 1). By classifying forest cover types at multiple levels of detail, we assess how much supplemental information a predictive model can contribute to the current available forest inventory data. We also plan to use these results to evaluate the insect hazard, invasive plant threats, and forest restoration strategies after southern pine beetle outbreaks in the southern Appalachians (Xi et al. 2007, Cairns et al. 2008, Xi et al. 2008).

Further information about our restoration project is available at http://landscape-restoration.tamu.edu/.

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Mine Land Reclamation Strategies for the Restoration of American Chestnut

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Hundreds of thousands of acres of reclaimed strip mines throughout the central Appalachians are experiencing arrested succession. They remain in the same grassland state produced immediately following reclamation and do not appear to be succeeding back to hardwood forest, the original condition prior to disturbance. We report here methods to accelerate hardwood succession for reclamation purposes and to restore American chestnut (*Castanea dentata*).

Prior to the 1930s, coal mining in the United States was largely an underground affair. Following this period, in part because of large demands associated with World War II, coal mining switched to an aboveground endeavor. The Surface Mining Control and Reclamation Act (SMCRA) of 1977 grew out of a concern for the environmentally degrading effects of surface mining (OSM 2006), becoming the primary regulatory element for active mines and providing guidelines for reclamation of abandoned mine lands.

While SMCRA has had an important impact on the reclamation of mined lands following this period, it created some unanticipated results. Elements of SMCRA required that the land be returned to its original contour and revegetated to reduce erosion. The former requirement resulted in heavy earthmoving equipment often compacting soils, leading to biologically problematic soil bulk densities. The latter requirement encouraged the planting of non-native graminoids and forbs, many of which are highly competitive. The outcome has been grasslands that remain in arrested succession for decades. Rarely have these lands entered into a successional pathway that would return them to natural forest conditions. As a result, surface mining has inadvertently led to wholesale conversion of native hardwood forest into grassland, often dominated by non-native invasive species, throughout much of the Appalachian region.

Preliminary research suggests that the American chestnut can thrive on abandoned mine land soils (Keiffer, McCarthy and Herendeen, unpub. data). The American

Chestnut Foundation will soon begin to deploy blight-resistant American chestnut hybrids as part of a major restoration effort for this species (Jacobs 2007). The objectives of this research are to utilize the basic Forestry Reclamation Approach (FRA; Burger et al. 2005) to assess how different forms of surface conditioning of mined soils relate to the survival and growth of American chestnut (and ultimately other hardwood species). We wish to evaluate both aboveground and belowground performance, as well as ectomycorrhizal relationships. Our primary goal is to merge mineland reclamation with ecological restoration.

Our study site is the Tri-Valley Wildlife Management Area (TVWMA), located in central Ohio (near the city of Zanesville, Muskingum County). The TVWMA, a 7400-ha area devoted to public hunting and fishing, is owned and managed by the Ohio Department of Natural Resources and consists mostly of reclaimed strip mine lands from the late 1970s and early 1980s. The grassland topography is highly dissected and rolling, with small pockets of hardwood forest. The dominant species are red fescue (*Festuca rubra*) and other graminoids, with lespedeza (*Lespedeza* spp.) and autumn olive (*Eleaegnus umbellata*) present sporadically. Various test plantations of softwoods and hardwoods can be found throughout the site.

Three experimental blocks (approximately 30 × 60 m each) were created in the spring of 2007, each containing a control and three treatments in a 2 × 2 factorial design. The treatments were deep cross-ripping (using a 1.0-m ripping bar on a D-6 sized dozer) on 1.5-m centers, plowing and disking using conventional agricultural equipment, and a rip followed by plowing and disking. We divided a total of 1,200 bare-root American chestnut seedlings evenly so that each treatment/block combination included 100 plants, of which roughly 33% were pure American, 33% were 7/8ths hybrids (i.e., 1/8 Chinese chestnut, *Castanea mollissima*), and 33% were 15/16ths hybrids (putatively the most resistant). We planted pure and hybrid seedlings randomly on a grid with 1.5-m spacing interval to roughly correspond to the ripping treatment. All seedlings were originally inoculated at the Ohio Tree Nursery (Marietta OH) with the ectomycorrhizal fungus *Pisolithus tinctorius* (Pt). Each seedling was individually caged with 1.2-m tall poultry netting to prevent deer browsing.

We examined seedlings monthly for survival, growth, and vigor. At the end of the first growing season a subset (10%) of seedlings was randomly selected for detailed biomass analysis and an evaluation of ectomycorrhizal colonization. Fungi found on roots were identified to species based on fungal morphology and molecular sequencing of the ITS region using primers ITS1-F and ITS4. Sequenced samples were compared with known species in the GenBank database using BLAST.

The aboveground seedling dynamics were quite apparent. There was a dramatic difference in survival as a function of the treatments (Cox proportional hazard model, Likelihood = 273, df = 3, $P < 0.0001$). Each of the treatments resulted in 90–95% seedling survival after one year, whereas the control survival was only 40%. Interestingly, we found a similar pattern, though less pronounced, in the type of seedling: the two hybrids experienced 79–81% survival, while the pure American seedlings yielded 68% survival (Cox proportional hazard model, Likelihood = 9.3, df = 3, $P < 0.0001$). Seedling height, stem width, and vigor all paralleled the survival data in terms of general treatment response (one-way ANOVA, all $P < 0.001$). Detailed results and photographs can be found in McCarthy (2008). In sum, American chestnut establishment, survival, and growth were markedly increased by soil surface treatments. The treatments all resulted in significantly ($P < 0.001$) reduced soil bulk densities and ostensibly decreased competition from non-native graminoids and forbs (B.C. McCarthy, pers. obs.). We expect that continued monitoring may show that deep ripping has a pronounced effect in later years as root systems increase in size and begin to utilize a deeper rooting profile.

One curious result of this research was that we never recovered Pt mycorrhizae from the seedlings, despite their being initially inoculated with this species. Of the 120 seedlings sampled, 102 were morphologically identified as having ectomycorrhizal fungi. Of these, 60 root tips were successfully sequenced using ITS primers and identified to genus (the other 42 were deemed inconclusive). A total of 20 taxa were discovered on the roots (*Hebeloma* spp. and *Thelophora* spp. were the most abundant). Either Pt was never retained after inoculation, or the other species competitively displaced it. However, we were encouraged by the high level of fungal diversity encountered. This should aid future restoration efforts utilizing other hardwood seedlings.

In summary, we found that reclaimed strip mines provide a reasonable environment in which to do American chestnut restoration work. This assists mine operators in following regulatory goals while enhancing ecosystem integrity and services and increasing forest habitat in the landscape. Chestnut can be planted and successfully establish on these sites and then disperse in later decades into nearby woodlands (Jacobs 2007). Establishment, though, will involve some sort of soil treatment. We hypothesize that the combined ripping followed by plowing and disking will likely prove to be the best treatment for long-term survival, as seedlings are able to grow and explore deeper into the soil.

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Environmental Restoration in a Tropical Rainforest in Mexico

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The Montes Azules Biosphere Reserve, located in southeastern Chiapas and established in 1978, protects 331,000 hectares of the Lacandona Rainforest, one of the most important regions in Mexico owing to its high biodiversity (de la Maza 1997, Dirzo 1994) and the environmental services it provides. The larger Lacandona region was once comprised of 1.8 million hectares of forest (SEMARNAT 2006), but in the last decades it has suffered significant deforestation and fragmentation (Mendoza and Dirzo 1999, Carabias et al. 2006), causing the loss of two-thirds of the rainforest in the area (SEMARNAT 2006). The region has also been the site of intense conflicts over land tenure and resource use, involving people displaced through complex sociopolitical factors, the Mexican government, and conservation interests. As part of these conflicts, some areas inside the reserve were illegally cleared less than ten years ago for cattle grazing (in the high plains) and agricultural cropland (along the river). A land planning effort facilitated by the federal government over the past several years resulted in the resettlement of 16 communities between November 2004 and February 2005. These areas, almost 290 ha distributed in 10 sites of 2–80 ha, are no longer occupied by people; however, they suffer from impoverished biodiversity and richness and harbor exotic species such as fruit trees and corn and bean crops.

Here, we report on a project to restore the cleared areas. The project involved two federal agencies, the Comisión Nacional de Áreas Naturales Protegidas (CONANP) and Procuraduría Federal de Protección al Ambiente (PROFEPA), the nonprofit organization Natura y Ecosistemas Mexicanos (formerly Espacios Naturales y Desarrollo Sustentable), and *ecoguardas* (local forest rangers) from three Mayan communities, Lacanjáh-Chansayáb (Lacandón), Nueva Palestina (Tzeltal) and Frontera Corozal (Cho'l).

For our study, we selected ten plots ranging in size from 2 to 20 ha (128 ha total): four crop fields in the riparian zone of the Lacantún River (croplands) and six pastures in the highs plains of the Cordón Chaquistero. As part of site preparation, we removed all of the agricultural plants, including trees and crops.

In the design phase, we implemented a Restoration Workshop with participating stakeholders to define the activities and species to be utilized in each site. We discussed seed dispersal and direct sapling plantings from our nurseries to create mixed stands and sapling transplants from remnants to create mixed stands. We selected 12 tree species depending on their ability to grow in degraded areas and their availability.

During January 2005 and March 2006, we collected seeds and saplings from the remnant rainforest and produced saplings in our nurseries. In the study plots, seeds were directly dispersed by hand while saplings were planted throughout the entire area with a minimum distance of 2 m between each plant in a grid arrangement, with a final total density of 2,000–5,000 individuals per hectare. The 12 species were randomly placed throughout the grid according to availability in each site. We measured sapling growth rates for eight months, but not seedlings germinated from seed because of their small height. Statistical comparisons were not reliable because of insufficient replicates. Here we report data pooled for saplings from both nurseries and remnant forest.

Some species grew more than 40 cm in eight months (Table 1), such as trumpet tree (*Cecropia obtusifolia*, 69 cm), balsa (*Ochroma pyramidale*, 43 cm), and kapok tree (*Ceiba pentandra*, 40 cm). These species probably will form a complete canopy within the next three years, considering their growth rates. Growth was slower than in other pastures within similar rainforest in Mexico (Meli and Dirzo unpub. data, Martínez-Garza et al. 2005), probably because we included data from all sites without differentiating riparian zones and pastures. After 14 months, we found that pasture plots were still dominated by perennial grasses, preventing secondary regeneration of native species. We found higher growth rates within the riparian sites (Meli, pers. obs.), suggesting more suitable edaphic conditions and water supply but also less competition because of the absence of grass cover. Meanwhile, the percentage of canopy cover in these sites doubled in one year. We are optimistic that planting combined with natural seeding will increase diversity in the next three years in these sites.

Our results suggest the potential of these 12 species for restoration purposes in the Lacandona region, but more research is necessary on the effects of soil conditions and competition between trees and grasses. We consider all the species desirable, but at later stages we also recommend developing a more diverse scenario. Larger scale restorations can usefully implement other strategies, including attracting dispersers and ameliorating soil conditions.

Table 1. Species transplanted into degraded areas of the Montes Azules Biosphere Reserve in Mexico and their mean (± SE) monthly growth rates in descending order.

Species	Family	Common name		Growth (cm/month)
		Spanish	English	
Cecropia obtusifolia	Cecropiaceae	Guarumo	Trumpet tree	8.61 (0.87)
Ochroma pyramidale	Bombacaceae	Balsa	Balsa	5.37 (0.48)
Ceiba pentandra	Bombacaceae	Pochota	Kapok tree	5.01 (0.52)
Spondias mombin	Anacardiaceae	Jobo	Yellow mombin	3.62 (0.40)
Castilla elastica	Moraceae	Hule	Mexican rubber tree	2.75 (0.30)
Pachira aquatica	Bombacaceae	Sapote de agua	Money tree	1.75 (0.13)
Schizolobium parahybum	Fabaceae (Caesalpinoideae)	Plumillo	Brazilian firetree	1.50 (0.08)
Vatairea lundelli	Fabaceae (Faboideae)	Amargoso	Danto	1.37 (0.16)
Licania platypus	Chrysobalanaceae	Cabeza de mico	Licania	1.25 (0.09)
Ficus sp.	Moraceae	Amate	Fig	0.48 (0.05)
Pouteria sapota	Sapotaceae	Sapote	Naseberry	0.31 (0.006)
Brosimum alicastrum	Moraceae	Ramón	Breadnut	0.08 (0.01)

However, restoration opportunities inside the Reserve are limited by the continued presence of some settlements that are still being negotiated. We consider resolution of land tenure conflicts imperative to the ecological restoration of degraded areas inside the Reserve and its buffer zone.

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Hurricane Effects on the Piedmont Forests: Patterns and Implications

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Restoring forest damaged by catastrophic wind events is a significant task for landowners and forest managers in the eastern United States. A better understanding of disturbance effects and subsequent forest recovery is essential for effective damage assessment and restoration planning (Walker 1999). This is particularly important as ongoing climate change is likely to sustain increased hurricane activity for the foreseeable future (Goldenberg et al. 2001).

Historically, hurricanes have been a large, infrequent natural disturbance causing serious forest damage in the Piedmont region. On September 6, 1996, Hurricane Fran struck the central North Carolina Piedmont and caused severe damage to the Duke Forest, a Long Term Ecological Research area with many years of baseline data on tree, seedling, and herb dynamics, in some cases dating to the early 1930s (Peet and Christensen 1987). The availability of Duke Forest data spanning the years before and following Fran provided a unique opportunity to separate hurricane-induced changes from the background successional dynamics.

We resurveyed all long-term monitoring plots in the Duke Forest during 1997 and 2000. During the surveys, in addition to continuing measurement of vitality, diameter (dbh), height, and survival, we quantified hurricane damage status for each stem in the plots (Palmer et al. 2007, Xi et al. 2008b).

Hurricane Fran's strong winds and high precipitation resulted in a highly heterogeneous pattern of forest

Table 1. Resistance (in descending order) of tree species to hurricane damage for the major trees in the North Carolina Piedmont region, USA, based on our field observations on uprooting and breakage damage in the Duke Forest and consultations with experienced foresters.

Uprooting	Breakage
Common hackberry (*Celtis occidentalis*)	Common hackberry
Eastern redcedar (*Juniperus virginiana*)	Eastern redcedar
American hornbeam (*Carpinus caroliniana*)	Sweetgum
Hophornbeam (*Ostrya virginiana*)	Blackgum
Blackgum (*Nyssa sylvatica*)	Hophornbeam
Sweetgum (*Liquidambar styraciflua*)	Flowering dogwood
American sycamore (*Platanus occidentalis*)	Southern magnolia
Winged elm (*Ulmus alata*)	American hornbeam
Slippery elm (*U. rubra*)	Northern red oak
Northern red oak (*Quercus rubra*)	Water oak
White oak (*Q. alba*)	White oak
Scarlet oak (*Q. coccinea*)	Scarlet oak
Southern red oak (*Q. falcata*)	Southern red oak
Willow oak (*Q. phellos*)	Willow oak
Chestnut oak (*Q. prinus*)	Chestnut oak
Post oak (*Q. stellata*)	Post oak
Black oak (*Q. velutina*)	Black oak
Eastern redbud (*Cercis canadensis*)	Eastern redbud
American beech (*Fagus grandifolia*)	American beech
Tuliptree (*Liriodendron tulipifera*)	Tuliptree
Southern magnolia (*Magnolia grandiflora*)	American sycamore
Loblolly pine (*Pinus taeda*)	Shortleaf pine
Shortleaf pine (*P. echinata*)	Virginia pine (*Pinus virginiana*)
Water oak (*Quercus nigra*)	Loblolly pine
Red maple (*Acer rubrum*)	Black cherry
Southern sugar maple (*A. barbatum*)	Bitternut hickory
Flowering dogwood (*Cornus florida*)	Pignut hickory
Black cherry (*Prunus serotina*)	Shagbark hickory
Bitternut hickory (*Carya cordiformis*)	Mockernut hickory
Pignut hickory (*C. glabra*)	Southern sugar maple
Shagbark hickory (*C. ovata*)	Red maple
Mockernut hickory (*C. alba*)	Black walnut
Black walnut (*Juglans nigra*)	Winged elm
Sourwood (*Oxydendrum arboreum*)	Slippery elm

disturbance across the landscape (Xi et al. 2008a). Uprooting was the dominant damage type for canopy trees owing to soil saturation by the heavy rain immediately before and during the storm. Much of the damage was concentrated at the topographic extremes, particularly along stream bottom areas where the wettest soils were located and on slopes facing the primary force of the winds. Tree damage correlated positively with prehurricane tree size.

Fran significantly increased within-stand spatial heterogeneity as a result of the intense small-scale (subkilometer scale) boundary layer rolling effects (equivalent to the Langmuir cells in aquatic systems; Wurman and Winslow 1998, Xi et al. 2008b). In addition, the hurricane resulted in a substantial increase in tree-gap size and a dramatic rise in understory light. On average, mortality of large trees approximately doubled during the five-year period that spanned the hurricane event as compared to the prehurricane level. Increased mortality of hardwood trees was not confined to 1996, but continued for several years following the hurricane. These significant structural changes in the forest appear likely to substantially influence stand development and future composition.

The hurricane significantly diversified the live-tree size distribution in damaged forest stands. The density and size of trees of the upper canopy layer in both pine and hardwood forests decreased substantially. Small tree (1–3 cm dbh) density increased spectacularly for several rapid-growing, light-demanding hardwood species, such as tuliptree (*Liriodendron tulipifera*) and sweetgum (*Liquidambar styraciflua*). Particularly conspicuous was the increase in red maple (*Acer rubrum*) stem density in smaller size classes, suggesting that the hurricane accelerated the successional trend of more shade-tolerant but fire-sensitive species like red maple replacing oaks (*Quercus* spp.) and hickories (*Carya* spp.) in the eastern United States (Xi et al. 2008b).

Variation in tree species susceptibility to windthrow can partially explain within-stand variation in damage (Table 1). Tree susceptibility is determined by tree canopy characteristics, leaf features, and root system characteristics. Among large trees of the Duke Forest, Fran caused a higher incidence of damage in hardwoods than pines, as hardwoods usually have broad, spreading canopies and flat leaves that can catch the force of the wind much more readily than the smaller canopies and needles of pine trees (Xi et al. 2008a).

The most rapid changes following Fran were in the understory. Seedling density and species richness experienced an immediate drop, followed by a rapid rebound in density and more gradual recovery and enhancement in richness and diversity. Seedling recruitment did not increase continuously over time and overall seedling density was relatively low compared to prehurricane levels. Changes in sapling density and diversity were varied. This observation is consistent not only with the hypothesized relaxation of competition, but also the hypothesis that windthrows contribute greatly to tree diversity in the Piedmont (Peet and Christensen 1987).

Piedmont forests exhibit remarkable resilience to hurricane damage because of advanced regeneration. Canopy gaps created by the hurricane resulted in release of established shade-intolerant and partially tolerant seedlings and saplings (Figure 1). Most seedlings and saplings approximately doubled their relative growth rates after the hurricane, although not uniformly across tree species. Resprouting of damaged individuals and vegetative production of additional shoots were also common.

In contrast to the profound structural changes, hurricane-induced changes in tree species composition and diversity were modest and depended greatly on damage severity, prehurricane stand characteristics, and the temporal and spatial scales of observation. Tree diversity increased slightly or was maintained in most of the damaged forest stands as a result of colonization by light-demanding species. In addition, the disturbed forests experienced an increase in exotic tree species, such as princess tree (*Paulownia tomentosa*) and tree-of-heaven (*Ailanthus altissima*).

Large hurricanes have profound impacts on the Piedmont forests. The occurrence of Hurricane Fran has served to further document and clarify the variable and nonequilibrium nature of late-successional, mixed-aged hardwood forests of the Piedmont. Forest managers may use this information to assess the vulnerability of forest lands to hurricane damage and to design efficient campaigns for mapping tree damage after heavy storms. Our field observations of tree species resistance to hurricanes may help forest managers to select suitable trees when restoring forest damaged by catastrophic wind events. Finally, our research suggests that posthurricane monitoring to detect newly established populations of invasive plant species is important.

Figure 1. This 2001 photograph from the Bormann plot of Duke Forest shows several loblolly pine (*Pinus taeda*) and tuliptree (*Liriodendron tulipifera*) saplings established quickly after Hurricane Fran in 1996. Photo by Weimin Xi

Acknowledgments

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Native Forbs Occurring in Brome Fields within a Mixed-Grass Prairie Landscape (North Dakota)

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Smooth brome (*Bromus inermis*) is a cool-season, perennial grass introduced to North America's prairie regions from Europe in the late 1800s for use as livestock forage (Romo et al. 1990). Its ability to spread and outcompete native prairie flora makes the grass a major obstacle to reestablishing native vegetation in prairie soils that were previously cultivated. However, some native, herbaceous plant species may persist in or reinvade patches of brome, even in seeded monotypes of brome that are several decades old. Knowledge of which forb species can compete with brome successfully can assist land managers in improving seed mixtures used in agricultural field restorations. We believe multispecies seedings that include native forbs may better compete with brome while enhancing the structural diversity of the vegetation for wildlife.

To help provide such information for the northern mixed-grass prairie region, we measured the occurrence of native forb species in tracts of cropland that had been seeded to brome on the 2,200-ha Wilderness Area of the Lostwood National Wildlife Refuge in northwestern North Dakota. The Wilderness Area is mostly mixed-grass prairie interspersed with 0.2- to 10-ha seasonally flooded wetlands. Uplands are rolling to hilly, with thin loam soils. Nine 12- to 120-ha tracts of old cropland are distributed evenly across the area on the less hilly sites (6–15% slopes). The tracts were cultivated from about 1905 to 1950 then seeded to brome during the 1950s (USFWS 1998). Since its establishment in 1976, the Wilderness Area has experienced four prescribed burns, one wildfire, and four seasons of moderate cattle grazing.

We examined the tracts in July and August 2006, when the majority of plant species could be most readily identified. Within each, we delineated an interior zone (> 50 m from native prairie) and an edge zone (< 25 m from native prairie), separated by a 25-m buffer. We randomly selected 15 points in each zone of each tract; points within a given zone were at least 25 m apart. At each point, we recorded species of forbs detected within a 1- × 12-m quadrat. We calculated frequency values for each of the 9 tracts and then determined overall means by species and zone. The Domin-Krajina ratings of cover-abundance allowed us to convert frequencies to cover estimates (Bonham 1989, 129). We used the paired *t*-test to compare edge and interior zones for each species.

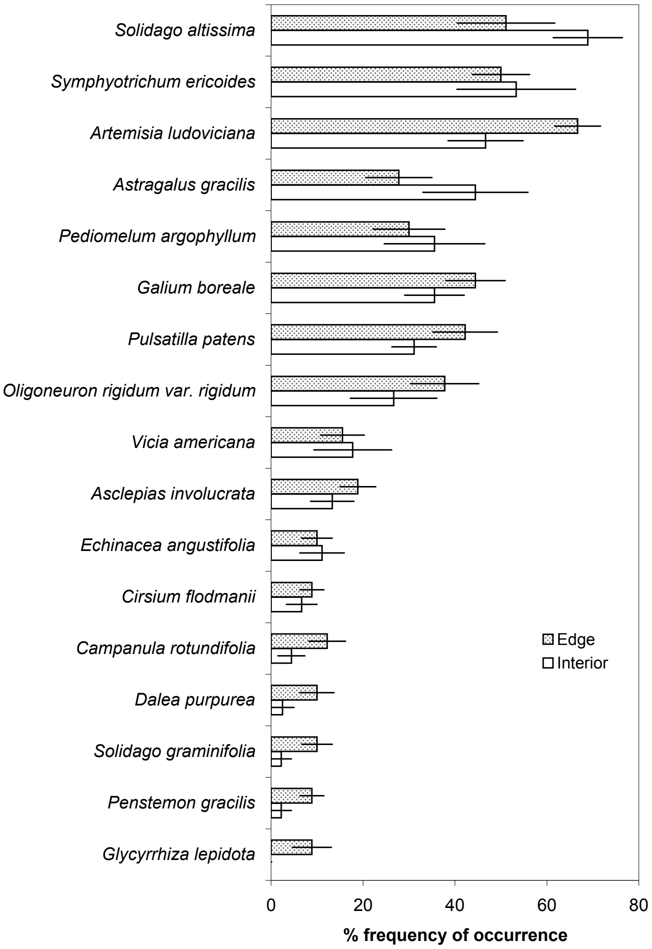

Figure 1. Mean frequency (± SE) of native forb species commonly found in nine cultivated tracts that were seeded to smooth brome (*Bromus inermis*) in the 1950s on the Wilderness Area of Lostwood National Wildlife Refuge, North Dakota, as documented during summer 2006. Not shown are 13 species that were detected less frequently: *Astragalus crassicarpus, Pediomelum esculentum, Thalictrum dioicum, Pulsatilla patens* ssp. *multifida, Liatris punctata, Oxytropis splendens,* and *Fragaria virginiana* were found only in edges; *Polygala alba* and *Lygodesmia juncea* were found only in interiors; *Thermopsis rhombifolia, Artemisia frigida, Erigeron philadelphicus,* and *Symphyotrichum laeve* were found both in edges and interiors.

Brome composed roughly three-fourths of the vegetation cover in the interior and edges of old croplands (range 10%-99%). We detected 30 species of native forbs, all perennials (Figure 1). The edge zones contained 28 species and 22 were in interior zones. Only one species, white sagebrush (*Artemisia ludoviciana*), differed significantly ($p = 0.024$) between zones; it was found more often in the interior.

Eight species of forbs were encountered in at least one-fourth of the plots in both edges and interiors (Figure 1); three of these, all members of Asteraceae, were widely distributed (> 50% frequency). These high occurrences suggest that these species are likely to persist or reinvade regardless of management efforts.

Contrary to our expectations, we did not detect Pacific anemone (*Anemone multifida*), blue flax (*Linum perenne*), or white prairie aster (*Symphyotrichum falcatum*), all species that occur in tracts of old cropland elsewhere in the area (K. Smith and USFWS unpub. data). We may have overlooked the anemone and aster because of their very early and late phenology, but they likely were not widespread in tracts we examined because dormant individuals of the two species typically are conspicuous. In addition, we were unsure why we did not detect purple coneflower (*Echinacea angustifolia*) more frequently, as it appears fairly common in old cropland tracts elsewhere in the area.

When planning seeding mixes for restoring northern mixed-grass prairie in areas prone to invasion by brome, land managers may wish to exclude or minimize some plant species that are likely to persist or reinvade regardless of management efforts, making their inclusion in seed mixes an unnecessary expense. Our survey suggests eight such species of forbs. To improve the likelihood of establishing floristically and structurally diverse native vegetation, managers may wish to focus on seeding native plant species that appear to be moderately persistent and competitive in decades-old, seeded stands of brome. These could include some of the forb species we observed with less than 20 percent frequency.

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Workshop participants on a field trip with LCBS station director Rakan Zahawi to observe the planned biological corridor connecting current station lands with forest remnants spanning a 500-m elevation gradient to the Guaymi indigenous reservation. Photo by Federico Oveido

Tropical Forest Restoration Research Opportunities in Southern Costa Rica

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The Organization for Tropical Studies (OTS) Las Cruces Biological Station (LCBS), a leader in research on tropical restoration and conservation in fragmented landscapes, held a bilingual workshop in late July 2008 with over 60 participants from the United States and Costa Rica. Principal workshop achievements included providing a thematic overview of recent research achievements in conservation and restoration biology at LCBS over the past 10–15 years, integrating and catalyzing future research ideas, and identifying emerging themes in these fields. The workshop was also aimed at building collaborations between U.S. and Costa Rican scientists and institutions; a mix of participants including faculty, graduate students, and government and nonprofit agency representatives from both countries facilitated this process.

The workshop included research presentations and posters, excursions to numerous field sites, and panel discussions. In addition, there was a poster session in Spanish to share research results with local stakeholders. Research talks centered around the themes of natural forest regeneration and restoration in the region; conservation of faunal communities in the fragmented forest and agricultural landscape mosaic; and research agendas of Costa Rican nonprofit organizations and universities in the region. Panel discussions included strategies for facilitating collaborative research among Costa Rican and U.S. institutions; efforts to facilitate communication among scientists, farmers, and local K–12 educators; and an overview of the LCBS land acquisition plan for a corridor spanning a 500-m elevation gradient.

Emergent research themes included studies to monitor forest recovery and the coalescence of smaller forest fragments along recently abandoned riparian corridors in the matrix surrounding LCBS, restoration along an elevation gradient in the context of climate change, and human dynamics of landscape use and change due to external drivers such as foreign land purchase and changing agricultural markets. The need for more studies of biogeochemical and hydrological cycling to complement existing data sets on vegetation and avifaunal community composition in this landscape was highlighted.

To promote further research at LCBS and surrounding areas, there will be a post-workshop call for seed grants (for graduate students at U.S. and Costa Rican institutions) to conduct interdisciplinary pilot studies on themes related to the workshop. For further information on LCBS, a full list of talks and panels from the workshop, and opportunities for restoration research in the region visit the OTS website (www.ots.ac.cr) and click on the link for Las Cruces Biological Station.

Supplemental Irrigation and Fertilization of Transplanted Michaux's Sumac May Not Be a Restoration Panacea (North Carolina)

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Because Michaux's sumac (*Rhus michauxii*) is a federally listed endangered shrub, its actual or potential presence must be considered to avoid negative impacts on all publicly owned lands and in all proposed construction projects involving federal funding. Considerable interest exists in the Southeast in restoring, and ultimately delisting, Michaux's sumac. First recognized as a new species by Andre Michaux in 1794 in North Carolina, Michaux's sumac has always been rare. The species currently ranges from southeastern Virginia to central North Carolina and grows on a wide variety of sites. It is most commonly found on infertile, sandy soils in slight depressions in otherwise flat topography. Most importantly, this shade intolerant species rarely grows more than 80 cm tall and requires an open canopy, usually oak-hickory or pine savanna, where abundant sunlight reaches the forest floor. Historically, occasional surface fires (every 3–6 years) reduced the shade and litter from trees and stimulated root sprouting in Michaux's sumac to form clones. Most reproduction is local and vegetative, because the species is dioecious and insect pollinated, and the other sex is rarely within pollinating distance, resulting in limited seed availability for restoration efforts.

Transplanting is an alternative means of increasing the number of populations. Braham et al. (2006) demonstrated the feasibility of transplanting, but initial survivorship and vigor were low. Survivorship ranged from 25 to 80 percent, and the leaves were only about one-half of the usual size. Although survivorship and vigor are related, our experience indicates that low vigor is more problematic than survivorship, because it severely limits the capacity to form large clones from root sprouts. Vigorous root sprouting from healthy transplants can more than compensate for low survivorship. Because suboptimum water and nutrition may contribute to low vigor, we wanted to test whether irrigation and fertilization after transplanting would improve vigor.

We conducted our test in 2000 at three Piedmont locations in North Carolina, one at Schenck Forest (Wake County) and two at Hill Forest (Durham County). Both properties are owned by the Department of Forestry and Environmental Resources, North Carolina State University. The location at Schenck Forest occurs on Cecil sandy loam soil with an open overstory of loblolly pine (*Pinus taeda*). Both locations at Hill Forest occur on Appling sandy loam soil with an open overstory of mostly white oak (*Quercus alba*). Our study occurred 1–2 years after one clone had been transplanted to these locations as described by Braham et al. (2006). Plants in odd-numbered rows were assigned to the treatment group, and plants in even-numbered rows were the control group. Each plant in the treatment group was irrigated weekly from May 24 to September 24 with 4 L of tap water, poured slowly from a beaker. Every other week we dissolved into the water the standard dose (about 15 g) of Peters® 20-20-20 water-soluble fertilizer. The control group was not irrigated or fertilized. Because the plants were small (4–15 cm tall) and in rows spaced about 2 m apart, the possibility of roots of the control group receiving irrigation or fertilizer was considered negligible. Both groups received natural rainfall throughout the experiment. Owing to the low number of plants in the original transplanting, the number of plants in each group was relatively small: 57 for the treatment and 51 for the control. In late September, well after growth stopped, height growth that occurred in 2000 and length of the longest leaf were measured to the nearest 0.1 cm as indicators of vigor.

Because the height data were normally distributed and the leaf length data only slightly skewed, analysis of variance was used to test for differences between the two groups. Location effects were significant for both height ($p = 0.0001$) and leaf length ($p = 0.0027$) (Figure 1). All plants at Schenck Forest were on average shorter in height (5.4 cm) than those at both locations at Hill Forest (8.3 and 8.1 cm). Leaves at Schenck Forest were significantly

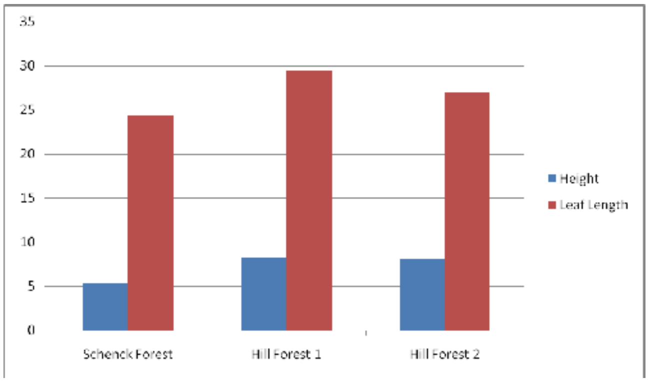

Figure 1. Mean height growth and leaf length (in cm) of Michaux's sumac (*Rhus michauxii*) at Schenck and Hill Forest in North Carolina. Location effects for height (*p* = 0.0001) and leaf length (*p* = 0.0027) were significant.

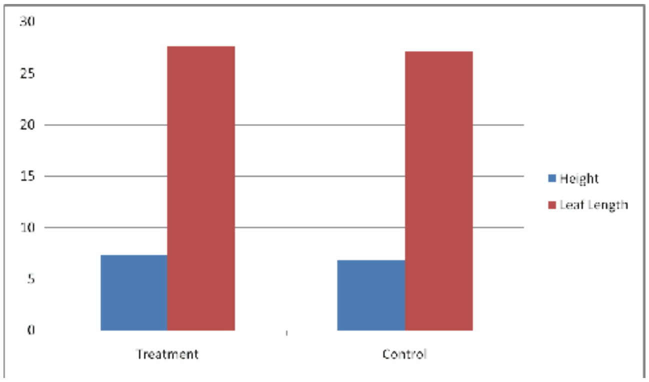

Figure 2. Mean height growth and leaf length (in cm) of treatment (irrigation and fertilization) and control of Michaux's sumac (*Rhus michauxii*) groups at Schenck and Hill Forests in North Carolina. Treatment effects for height (*p* = 0.9347) and leaf length (*p* = 0.5270) were not significant.

shorter (24.4 cm) than those at one location at Hill Forest (29.4 cm), but not the other (27.0 cm). Survivorship (89%) was the same for both groups. Significant location effects were not surprising, since the soil at Schenck Forest has less topsoil owing to erosion from a longer history of previous cultivation. This difference probably accounts for the location effects.

The treatment group grew an average of 7.3 cm in height, while the control group grew 6.9 cm (Figure 2). The longest leaf in the treatment group averaged 27.7 cm, while the control group averaged 27.2 cm (Figure 1). Neither height growth nor leaf length of the treatment group was significantly different from the control at any location, which was surprising, because irrigation and fertilization often increase growth. Because Michaux's sumac grows by repeated flushes from April to August (Thrush 2002), growth of the treatment group should have been responsive to irrigation and fertilization.

The reasons for our results are not completely clear, because they are somewhat contradictory to our earlier experience with fertilization. Although we did not specifically measure growth in the earlier study, retention and color of Michaux's sumac leaves were improved with fertilization in the greenhouse (Braham et al. 2006). However, Michaux's sumac usually grows on relatively infertile sites, and Chapin et al. (1986) have shown that plants adapted to infertile sites may respond less to fertilization than plants adapted to fertile sites. Whether Michaux's sumac forms mycorrhizal associations is not known. If so, fertilization possibly reduced the mycorrhizal populations of the treatment group, offsetting the benefits of fertilization.

Rainfall totals in 2000 were approximately average during most of the growing season, and about 2.5 cm above average in June and July. The native water-supplying capacity and fertility of the soils were adequate in times of average or greater rainfall. Therefore supplemental irrigation was probably not needed 1–2 years after planting in years of average rainfall.

Because Michaux's sumac is endangered, we could not measure root growth. However, we excavated a surprisingly large amount of root biomass in earlier transplant research (Braham et al. 2006). Michaux's sumac may routinely allocate more carbohydrate to belowground biomass, because such preferential allocation would be a decided advantage to a shrub adapted to fire-prone, infertile sites. Roots of the treatment group possibly grew more in response to fertilization. Keyes and Grier (1981) found that Douglas-fir (*Pseudotsuga menziesii*) trees on poor sites had greater root biomass than trees on good sites, possibly in order to absorb sufficient water and nutrients. We favor this hypothesis, which should be tested in future research.

We conclude that supplemental irrigation and fertilization of Michaux's sumac, applied 1–2 years after planting, does not increase aboveground growth under the natural conditions of this study, and therefore may not be a panacea. The practices might be useful, however, during the first year of transplanting, during times of drought, or on very sandy soils.

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