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What Is Local?

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Introduction: What Is Local?

A few years ago, I read some reports on conservation and population genetics that led me to question whether I, as the director of a native plant nursery in a large urban area, ought to pay closer attention to the issue of localness in plants. The more I read about the issue, the more pressing it became, and the more questions I began to ask myself. In restoring and managing the fragmented natural areas of the city, what constituted “local”? How far should we travel to collect seed for our projects? Were two naturally occurring populations of a species from two different parks in the Bronx once part of the same population? Had fragmentation and physical isolation resulted in two (genetically) separate populations? Was a species population in the Bronx the same as a population on Staten Island? And if so, could they be used interchangeably? Could plants from any one source in the five boroughs of New York be used anywhere else in the city without concern?

As I amassed more questions and noticed that none of the nursery’s users was presenting me with similar concerns, it became apparent to me that there was not a great deal of awareness of these issues, and I decided to organize a one-day symposium on May 23, 2006, at the American Museum of Natural History in New York City entitled “What Is Local? Genetics and Plant Selection in the Urban Context.”

I resolved to bring together some of the authors whose work had been guiding me for a day of discussion in front of a wider audience. Six speakers presented to a sold-out audience. Gerry Moore, Director of Science at Brooklyn Botanic Garden, set the context for the discussions by presenting a paper on the changing flora of New York City. Susan Mazer of the University of California, Santa Barbara, provided the basic genetic framework for the discussion and argued for careful consideration when translocating plants to a restoration site. Arlee Montalvo from the University of California, Riverside, detailed a methodology for making appropriate plant selections. Julie Etterson from the University of Minnesota, Duluth, discussed how climate change might alter the discussion of plant translocation. Steven Handle from Rutgers University approached the day’s proceedings from the perspective of urban restoration ecology. I concluded with a brief discussion of the policy roadblocks to implementing a more advanced approach to plant procurement in the urban context.

Since then we have hoped to bring this discussion to an even larger audience. One idea to make the Power Point presentations and audiotapes of the day’s proceedings available via the Internet is in the planning stages and should be implemented in the coming months.[1] A second opportunity was presented by our symposium cosponsors at Brooklyn Botanic Garden to pursue the discussion in an issue of its online journal, *Urban Habits*, devoted to the same subject, which we present to you here.

Several things have become clear to me since our symposium. First, the issues are complex and best addressed by individual projects. Second, a great deal more needs to be learned, and answers to the simple question, “What is local?” are likely to continue to evolve through time. Third, knowledge and experience should be shared as widely and among as many people as possible to address the issue.

Most importantly, I feel that we are asking the right question and that no matter how daunting the pursuit of the right answer, the effort will be worth it. I hope this issue of *Urban Habitats* will encourage readers to pursue their own answers to the question, "What is local?" and broaden the dialogue, leading ultimately and, I hope rapidly, to the implementation of improved restoration practices.

Edward Toth
Guest Editor

[1] Readers may contact me at edward.toth@parks.nyc.gov to learn how and when to access these.

What Is Local? An Introduction to Genetics and Plant Selection in the Urban Context

by Carrie Pike

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The urban landscape comprises myriad isolated green spaces inhabited by an assortment of vegetation types. To many city dwellers, these green spaces interrupt the monotony of concrete and steel and foster deep social attachment between city dwellers and nature. To a conservationist, these vegetation islands provide unique opportunities to restore ecological function to degraded areas by revegetating them with native plants. However, the restoration ecologist faces many challenges unique to the urban landscape. A significant investment in site preparation may be needed to offset the impacts of abiotic factors including compacted soil, drought, and air pollution. Seedlings that survive to maturity are not guaranteed immunity from these abiotic stresses, as is evident in the tree dieback and declines that plague many city landscapes. Abiotic factors are rarely the sole causal agent of urban tree declines since the presence of a multitude of other factors, such as insects and disease, are associated with symptoms of this decline. For example, drought in combination with insect defoliation predisposed black oak (*Quercus velutina*) to decline (Pike et al., 2001), while defoliation and several pathogens were causal agents in the decline of English oak (*Quercus robur*) (Marçais and Bréda 2006). The impacts of biotic factors on plant health may be heightened or lessened in urban areas depending on the ecology of the insect or disease. In these examples, black oaks in urban areas were more susceptible to damage from gall wasps than trees surrounded by contiguous forest. In contrast, the pathogens inciting decline in *Quercus robur* are less problematic in urban settings where soil disturbance prevent *Armillaria* fungus from spreading great distances.

Environmental stresses, to a limited extent, can be managed through site preparation. Some insects and diseases can be managed with pesticides or other integrated approaches. However, long-term sustainability of a restoration planting, beyond the generation that is planted, can only be realized if the basic requirements for reproduction of plant species are met. The consequences of reproductive failure may be immediate and obvious—for example, if seed fails to form—or delayed until subsequent generations. Processes causing reproductive failures are exacerbated, for some species, when plants are isolated from potential mates—a common occurrence in the fragmented urban environment.

Plant reproduction commences when a pollen grain unites with its female counterpart. Pollen is disseminated by a number of vectors, both biotic (such as insects) and abiotic (e.g., wind). Pollen grains from wind-pollinated plants, such as conifers, can travel long distances before landing, allowing isolated stands to remain reproductively viable even when the nearest individual is some distance away. Some plant species are self-compatible, and their reproduction does not depend on the nearby presence of an unrelated mate. Many species, however, are obligate "out-crossers," and seeds that are produced from an inbred cross are either aborted or result in plants with reduced fitness. Orchid species are obligate out-crossers and provide an interesting demonstration of the potentially damaging effects of inbreeding depression on fragmented populations (Izawa et al. 2007). If plants are dependent on local pollen sources, then it is essential that a variety of unrelated families be available to minimize the effects of inbreeding depression (see Leimu et al. 2006, for a discussion of population size and fitness for a variety of plant species).

Inbreeding depression is the reduction in fitness that occurs when related individuals mate; and it can have significant consequences for a planting's long-term reproductive success. It is most likely to occur in small or isolated plantings that lack contiguous land masses for gene flow or pollen exchange. The relatedness of seed in a given seed source (or seed lot) used in a restoration planting can vary greatly. One seed lot may contain seed bulked from a single plant, while another may contain seed collected from an assortment of plants. Understanding seed collection protocols is essential to ensure that seed from a variety of unrelated seed sources are used, which in turn may reduce the incidence of inbreeding depression for future generations of plants.

Outbreeding depression can also contribute to reduced fitness, and occurs when a local, established population crosses with introduced material of the same species (Hufford and Mazer 2003). Native plants that have survived local stresses have likely evolved traits that maximize their adaptation. Outbreeding depression occurs after the native and introduced material breed, and the offspring of these "hybrid" crosses contain a combination of adapted and non-adapted traits. Outbreeding depression may be mitigated by planting seedlings grown from seed procured from a nearby source where environmental conditions are similar to that of the planting site. The effects of outbreeding depression can take decades to be realized, since future generations are affected.

Inbreeding and outbreeding depression are not the only genetic factors to consider in optimizing the sustainability of a planting. Lynch (1991) provides a theoretical comparison of inbreeding and outbreeding effects. Rogers and Montalvo (2004) present a comprehensive discussion of the importance of biodiversity and genetics in plants and planting programs. Extrinsic factors, such as matching seed source to the planting location, also play a role in determining reproductive success. In plant species with high levels of biodiversity, such as conifers with continental-wide ranges, matching the correct seed source to a site is critical to ensure survival and reproductive longevity. Seed-transfer zones, delineated from climate, soil, elevation, and occasionally from common-garden data, can be employed to assist in matching seed to a particular geographic area. Ying and Yanchuk (2006) provide an informative summary of the history and methodology behind seed zones established in British Columbia for forestry applications. However, for many species, the distance that seed lots can be safely "moved" without risking maladaptation is not as well established. In the absence of clear seed-transfer designations, local seed sources provide the best insurance against the deployment of plants that are not suitably adapted for environmental conditions in the restoration planting.

Conservationists face a new hurdle in today's world—the emerging threat of climate change. Research on the implications of climate change for the evolution of native plants (see example in

Etterson and Shaw 2001) is a burgeoning field of study. A plant's success in a novel climate will be determined by its ability to disperse, breed, and adapt to its new surroundings (Davis et al. 2005). A plant's ability to tolerate changes in its environment, or its plasticity, is dependent on its genetics. More research is necessary to inform our understanding of plasticity and tolerance in plants, both in urban and rural landscapes. A general strategy to maximize biodiversity both within and among species will improve the chance that genes for adaptation are present in the population facing dramatic environmental changes.

Historically, plants migrated northward as temperatures rose and glaciers retreated (Davis and Shaw 2001). Plants attempting to migrate today face significant impediments that are both of human (urbanization and agriculture) and nonhuman origin (lakes and rivers). These barriers can be quickly overcome through a restoration effort. Should seed sources from southern locales be favored in northern areas over local sources given the forecasts of increasing temperature? This notion of "assisted migration" in which distant seed sources are favored over local sources in anticipation of climate change is controversial (see McLachlan et al. 2007). Restorers are faced with the need to balance risks of introducing a seed source with the potential maladaptation that might result from climate change. Weather is notoriously erratic; global mean temperatures may be rising, but day-to-day fluctuations can create stressful conditions for plants that are far removed from their origin. In addition, plants that are moved great distances also risk being out of sync with the photoperiod to which they have adapted. Finally, seed or seedlings from distant locations may introduce fungi, insects, or other "volunteers" that could be harmful to flora and fauna inhabiting the planting site. A quantitative approach relying on data from common garden trials can assist in determining appropriate seed sources for future climate scenarios. This method is demonstrated for black spruce (*Picea mariana*) in Lesser and Parker (2006). The risks associated with assisted migration may be easier to justify for plant species that face extinction in a specialized or unique population or extinction of the species as a whole.

Restoration of native plants is a costly but valuable investment across a fragmented and often degraded urban landscape. To maximize planting success in the short- and long-term, efforts should be made to incorporate the genetic infrastructure of desired plant species into restoration plans. For example, plants that tolerate inbreeding may only require a small number of individuals to reproduce successfully in the future. Other plants might benefit from the inclusion of numerous unrelated families, or additional plant species that support populations of local bees or other pollen vectors. In the absence of clear seed-transfer guides, local seed sources should be utilized to improve the likelihood that seed will be adapted to its novel environment. Climate change brings a new set of challenges to conservation planning. For plants that face extinction, radical measures may be needed to ensure their preservation. For all other plants, restoration can enhance the diversity of existing plant communities, which in turn may offset the potential for inbreeding and provide the plant community with the genetic tools needed to thrive and evolve to the changing climate. The extra steps needed to ensure the long-term sustainability of a restoration effort will provide the greatest benefit to future generations of plants and the people who treasure them.

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Practical Seed Source Selection for Restoration Projects in an Urban Setting: Tallgrass Prairie, Serpentine Barrens, and Coastal Habitat Examples

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Abstract

Anthropogenic activities have dramatically altered native plant communities through both habitat reduction and habitat fragmentation. Awareness of these changes has lead to an increased interest in restoring extirpated populations and augmenting remnant communities in urban, suburban, and agricultural landscapes. Ecological restoration frequently requires seeds of component species, and the choice of local, nonlocal, and cultivar seed sources could affect the success of a restoration project. In this article, we describe restoration projects conducted in tallgrass prairie, eastern serpentine barrens, and coastal South Carolina to illustrate practical advice on seed-source selection. We advocate the use of locally collected seed if available, but we acknowledge that nonlocal sources from similar ecological settings (via ecological matching), geographically local sources from different habitats, or unrestricted seed sources may be appropriate depending on the goals of the specific restoration project.

Key words: restoration ecology, local ecotype, *Muhlenbergia sericea*, sweetgrass, Gullah, community participation, tallgrass prairie, eastern serpentine barrens, coastal habitat

Introduction

When discussing plant community restoration, interested parties typically raise the following questions: What is a local seed source? How far can we go to collect plant material for our restoration project? Can we purchase seeds rather than collect them? Individuals interested in restoring, reestablishing, recreating, or augmenting a historical native plant community will likely be very interested in identifying and selecting local ecotypes. Seminal research and recent literature reviews have established the scientific justification for selecting local seed sources for restoration projects and offer general guidelines for select species and ecosystems (Clausen and Heisey 1958; Hufford and Mazer 2003; Joshi et al. 2001; Kawecki and Ebert 2004; Lesica and Allendorf 1999; Leverich 2005; Linhart and Grant 1996; McKay et al. 2005; McMillan 1959;

Montalvo et al. 1997; Packard and Mutel 1997; Rogers and Montalvo 2004; Schaal and Wilkinson 2001). In this paper, we outline some commonsense considerations for seed-source selection when conducting plant community restoration projects within urban settings. To illustrate our points and provide real world examples, we share lessons learned about tallgrass prairies, serpentine barrens, and coastal habitats.

Defining Goals

From our experience and that of others, we argue that the first step to any successful restoration project—and, accordingly, to seed source selection—is to consider an array of goals and clearly define the context of the project and project priorities. We provide an overview of suggested questions to consider when planning and implementing a restoration project in urban areas and wildland-urban interfaces (Table 1). In the following section, we explore five of the more common goals for habitat restoration and reclamation. By examining these goals, we provide a range of options for restoration.

***Goal One*:** Establishing a historical plant community is a challenging goal for anyone pursuing ecological restoration using historical local genotypes because in many regions there are few situations where organizers will find remnant populations with historical genotypes on-site.

***Goal Two*:** Many individuals conducting ecological restoration focus on reestablishing native plant communities with local, but not necessarily historical, genotypes that would have once occupied the restoration site. Selection of plant material that has evolved under similar ecological conditions as the proposed restoration site should have genetic combinations (genotypes) that are more likely to be adapted to present ecological conditions than genotypes that have evolved under other ecological settings. For example, selecting seeds from a population in a wet habitat to restore a wetland community is most likely a better ecological match of plant material than selecting seeds of the same species from a dry habitat.

***Goal Three*:** How should project organizers proceed if there are no local native sources or if local sites are too small to provide sufficient seeds for the entire restoration project? Without sufficient local native seed sources, restoration managers may choose to collect seeds of the desired species from different ecological settings or purchase seeds from a native plant seed supplier. When lacking sufficient seed sources, organizers have three possible approaches. Restoration professionals may decide to use the local seeds but may have to establish seed increase plots on-site and use a multiple-year approach to generate sufficient seeds for the entire project. Two potential negatives to this approach are: (a) multiple seasons needed to generate sufficient seeds may not fit within the time constraints of the funding source, and (b) danger of founder effects due to the small original gene pool used to establish the seed increase plots. Founder effects occur when a few individuals are used to establish a new population, with the resulting population containing only a fraction of the original genetic diversity. Negative effects associated with founder effects, genetic drift and potential inbreeding depression, can be reduced by establishing seed increase plots and/or restoration projects with seed from many individuals collected from across the original native plant community.

Additionally, restoration professionals may choose to collect seeds of target species from a relatively close geographic area but with less emphasis on ecological matching. For example, Cook County, Illinois, has approximately 67,000 acres of prairie, savanna, wetlands, and forest managed by the Cook County Forest Preserve District, and these natural areas could be potential seed sources for restoration efforts in the greater Chicago metropolitan area. The same can be said for Westchester County, New York, and its 39 county-owned natural areas, which could be seed sources for restoration projects in the greater New York City metropolitan area (see fcwc.org/directory/wcoppnc.htm).

Restoration professionals can also decide to purchase seed from a native plant seed producer, regardless of the origin of the original seed source. When using warm and cool season grasses in an urban project in the Northeast, participants can purchase cultivated varieties of these grasses that originated from Kansas, Nebraska, Oklahoma, and Illinois and are produced in large seed increase plots throughout the United States. This large-scale seed production results in relatively inexpensive high quality seed in sufficient quantities to do large-scale plantings. Some of these cultivated varieties have been developed as native forage crops and may not, however, be genetically and ecologically equivalent to native populations in this same region (Gustafson et al. 1999; 2004a; 2004b; Gustafson et al. 2005). Care should be taken in choosing seed because seed purchased from commercial sources may have a tendency to dominate restorations (Baer et al. 2004) as has been the case for Blackwell switchgrass under some circumstances (Schramm 1978).

There are naturally occurring patterns of adaptation across large geographic areas for species with fairly large species ranges (Gustafson et al. 2002; McMillan 1959) or a range of different ecological settings (Huff et al. 1998; Rice and Knapp 1998;). Rogers and Montalvo (2004, Table 10.3) nicely summarize 17 grass, 37 forb, 11 shrub, and 10 tree species studies that have investigated local adaptations or genetic differentiation in at least one state of the Forest Service Region 2 (Colorado, Kansas, Nebraska, South Dakota, Wyoming). While some of these native species do have extensive ranges, it may not be appropriate to assume that the patterns of local adaptation in these states accurately reflect selection dynamics in other regions of the country.

***Goal Four*:** If the potential restoration site has been so dramatically altered (via strip mining, decommissioned landfills, phosphate mines, etc.) that few plants occur there, then the restoration practitioner may select native species that have the potential to grow on that site. This is not a restoration project per se, but more of a reclamation project where the goal is to establish a plant community on a highly disturbed site. Fresh Kills Landfill, on Staten Island, is a 2,200-acre landfill that officially closed in 2001 after it received debris from the World Trade Center. The City of New York is currently planning a large-scale reclamation/restoration of the Fresh Kills Landfill site to create a world-class park (see nycgovparks.org/sub_your_park/fresh_kills_park/html/fresh_kills_park.html). Reclamation of strip mines, decommissioned landfills, and brownfields offers an opportunity for restoration professionals to incorporate native plant species into large-scale reclamation projects. Depending on the duration and intensity of the original disturbance, the types of plant communities that can be established on these sites may provide valuable ecosystem functions that will enhance or create recreational opportunities, habitat for wildlife, and a focal point for the propagation of native plant species for use in these local urban landscapes.

***Goal Five*:** The use of nonnative species does not constitute a native plant community restoration project. We chose to include the nonnative species option in our generic goals simply to establish the range of possibilities for seed-source selection. We do not, however, promote the use of nonnative species in any project that has the goal of restoring native plant communities.

In the following section we discuss three examples of habitat restoration or reestablishment of native plant communities in areas where they no longer exist. Using only locally adapted genotypes (ecotypes) to restore the community is desirable, yet often local native populations no longer exist in areas planned for restoration. In such situations, having clearly defined goals and a sound understanding of the ecology of the plant community—including species composition as well as disturbance dynamics—can help guide selection of plant materials and improve restoration project success.

Below we discuss three habitat examples with implications for the urban environment. In each case, we emphasize key elements of interest to restoration practitioners, and we provide an overview and a project example. In the case of the coastal habitat, we focus our efforts on a native species used for restoration by long-term residents in the region. Since this case has added socioeconomic relevance, we provide more detail of the context of these restoration efforts.

Case One: Tallgrass Prairies

Preservation and conservation of native grasslands has steadily increased during the last three and half decades, although efforts to restore degraded or extirpated communities are hampered by the scarcity of remnant sites. Remnant North American tallgrass prairies, for example, currently occupy < 0.01% of their historical range (Packard and Mutel 1997), with many of the highest quality remaining remnants as small pioneer cemeteries and linear-shaped railroad right-of-ways (Figure 1). Conversion of the nutrient rich native grasslands into row crop agriculture has reduced the size of the remaining grasslands and increased the distance between native sites beyond many species dispersal distances. Organizations and private individuals have taken an active role in restoring native communities throughout North America, with many scientists and restorationists agreeing that matching ecologically appropriate genotypes to restoration site conditions will increase the likelihood of a successful project. The problem lies in the fact that there are very few remaining remnant grasslands and that many of them are very small (≤ 5 acres).

Project: A local elementary school in central Illinois wanted to restore a section of tallgrass prairie along the public bicycle path behind the school. The Freedom Prairie, as it is known, is located south of Colleen Hoose Elementary School, along Constitution Trail, in Normal Illinois. Restoration of this small (10 meters by 50 meters) prairie began in the spring of 1990 and was sponsored by the John Wesley Powell Chapter of the Audubon Society. McLean County Illinois once had 683,136 acres of tallgrass prairie; by the time of this restoration project, only one pioneer cemetery prairie (~5 acres) state-protected remnant was left (Anderson 2006). There were, however, several native prairie communities along railroad right-of-ways and several restored prairies established by The Nature

Conservancy, local universities, and civic groups interested in promoting prairie and savanna ecosystems. The goal of this school's project was to establish a small-scale historical plant community that included local ecotypes (Goal #2). The initial planting of the Freedom Prairie was accomplished by hand broadcasting native warm season grass seed purchased from commercial grower and elementary school children planting native prairie plant root stocks obtained from the Illinois Department of Natural Resources State Nursery, Mason County, Illinois. As with many restoration projects, one can increase plant species diversity by planting seeds of additional native species collected from local remnant sites. If the project had been to establish a tallgrass prairie on a 100-acre parcel of land taken out of row crop or pasture production, then the planning and implementation of the project would have been much more complex.

Case Two: Eastern Serpentine Barrens

The eastern serpentine barrens are historical fire-dependent grasslands of the mid-Atlantic region of North America. These communities are characterized by unique plant assemblages, globally endangered barrens aster (*Symphyotrichum depauperatum*) (Figure 2), hairy chickweed (*Cerastium arvense* ssp. *velutinum* var. *villosum*), and shallow soils with high levels of magnesium, nickel, and chromium in concert with low phosphorus, calcium, and potassium (Brooks 1987; Gustafson et al. 2003; Gustafson and Casper 2004; Gustafson and Latham 2005; Latham 1993). It is believed that eastern serpentine grasslands once covered approximately 100,000 acres of the mid-Atlantic prior to European settlement, but currently there are fewer than 26 serpentine sites ≥ 5 acres from Georgia to Vermont. Removal of the natural fire dynamic and encroachment by urban development have contributed to the loss of these barrens (Latham 1993; Tyndall and Hull 1999). The unique flora of serpentine barrens is a consequence of the origin of serpentine soil, soil chemical composition, and the fire dynamic. Serpentine soil conditions are often associated with edaphic ecotypic variation or locally adapted genotypes that have evolved under these strong selective pressures of the serpentine soils (Brooks 1987).

Project: Restoration of a serpentine barren in the Philadelphia metropolitan area, Pennsylvania.

The urban expansion within the greater Philadelphia metropolitan area has destroyed many small remnant serpentine barrens. Most of these lost sites are in such poor condition that the locals do not even know that they have a unique serpentine plant community nearby (Latham personal communication). Restoration of these urban serpentine barrens typically requires removing tree species to reopen the canopy, replanting dominant grasses that are a significant component of this community, and reintroducing to the site rare species like the barrens aster and hairy chickweed. Given that there are few remaining eastern serpentine barrens and these sites may not be in close geographic proximity to the restoration site, restoration professionals may choose to purchase the grass seed from native plant suppliers and only field collect for select species like the barrens aster (Goal #3). The warm season grasses—such as big bluestem (*Andropogon gerardii*), Indian grass (*Sorghastrum nutans*), little bluestem (*Schizachyrium scoparium*), and prairie dropseed (*Sporobolus heterolepis*)—are significant components of the serpentine barrens;

they have been shown to contribute to plant community structure through plant-soil feedback interactions, and they provide much of the biomass fuel needed to carry a fire, which is an important dynamic of the eastern serpentine barrens (Castelli and Casper 2003; Casper and Castelli 2007; Gustafson and Casper 2004; Latham 1993). Seeds of these warm season grasses can be purchased from native seed vendors in Pennsylvania and New Jersey, but to the best of our knowledge none of these vendors specifically collects and propagates serpentine barrens collections. In this case, restoration professionals would have to decide if they want to purchase grass seeds originally from those states (Pennsylvania and New Jersey), purchase seeds from Midwestern or Plains states (Illinois, Missouri, Iowa, Kansas, Nebraska, Oklahoma), or if they want to establish on-site seed increase plots using only serpentine-collected plant material.

Case Three: Coastal Habitat and Sweetgrass

Coastal South Carolina is characterized by coastal plains, expansive estuaries, barrier islands, and back barrier (hummock) islands (Porcher and Rayner 2001; SCDNR). This region is known for its heat, humidity, mosquitoes, and hurricanes; however, residents commonly focus on the appealing climate for most of the year, natural beauty, and sociocultural distinctiveness (Halfacre et al. 2007). African-Americans living in the region, descendants of enslaved Africans, have maintained cultural traditions that were forged through the forced relocation of these peoples (NPS 2001). Gullah culture includes unique speech, religious beliefs and practices, family social units, music, dance, storytelling, arts and craftsmanship, and use of coastal resources (Crook et al. 2003; Pollitzer 1999). The terms *Gullah* and *Geechee* are often both used to describe similar cultures, but in South Carolina, *Gullah* is used to a greater extent than *Geechee*. Sweetgrass basketry is one of the cultural traditions preserved along the Gullah/Geechee coastline (NPS 2001).

Basketry was first introduced to the Carolina coast in the late seventeenth century (Rosengarten 1986), and sweetgrass basket making became increasingly important during the development of the tourism industry during the early twentieth century (Coakley 2006). Basket-making skills were carried over from slaves' homelands and were quickly adapted to the raw materials available in coastal South Carolina. The signature plant material used to make sweetgrass baskets comes from the perennial grass *Muhlenbergia sericea* (synonyms: *Muhlenbergia filipes* and *Muhlenbergia capillaris* var. *filipes*), which occurs in sandy maritime habitats on barrier islands and coastal woodlands along the southeastern and gulf coasts of the United States (Gustafson and Peterson 2007; Peterson 2003; Porcher and Rayner 2001; Radford et al. 1968).

Historically, the basket was used for fanning rice on plantations; after emancipation, the basket makers produced containers for storing food and other household items (Carney 2001). Local residents used these baskets for day-to-day agricultural and household purposes; they were objects of necessity. However, around the turn of the twentieth century, a group of black Mount Pleasant families began mass-producing more intricate "show baskets" (Figure 3) made from sweetgrass and bound with strips of palmetto leaf (Rosengarten 1986). Extirpation of historical *M. sericea* populations and urban development along the coast have resulted in fewer collectable populations, forcing basket makers to purchase or travel several hundred miles to collect sufficient plant material (Burke et al. 2003). Charleston's growth as a tourist destination and the associated rapid expansion of suburban and exurban residential development have both created a wider market for these baskets and threatened the basket makers' access to resources (Allen 2002; Hurley and Halfacre in press; World Travel and Tourism Council 2001).

Historically, basket makers and their families have had tacit arrangements for collecting plant materials on private property, but changes in ownership and suburbanization have altered these arrangements and diminished once readily available supplies. These stakeholders often express an interest in plants from local "wild" populations, citing qualitative attributes. These desires call attention to the differences between sweetgrass local ecological adaptations and the cultural preferences rendered by the basket makers (Halfacre et al. draft). Persistent public attention has enhanced the artistic, cultural, and monetary values of sweetgrass baskets, and it is an important source of supplemental income for black artisans (Allen 2002; Coakley 2006; Derby 1980; Hart et al. 2004).

Project: To establish a plant material source for sweetgrass basket makers in Mount Pleasant and Charleston, South Carolina.

Sweetgrass naturally occurs along barrier islands and the mainland juxtaposed between salt marshes and maritime forest from South Carolina to Texas (Pinson 1971; Ohlant 1992; Peterson 2003). While we can often find small populations on many barrier islands along coastal South Carolina, these populations typically have fewer than 20 mature individuals (Figure 4). In addition to not having abundant numbers of individuals within a population, the sweetgrass basket makers have indicated that they are no longer able to access sufficient areas to collect material to make their baskets (Hart et al. 2004).

Muhlenbergia sericea is recognized as a distinct species and not a variety of the more widely distributed *M. capillaris*, based on anatomical, cytological, genetic, and ecological data (Gustafson and Peterson 2007; Peterson 2003). The Citadel Plant Ecology Laboratory (CPEL) has established common garden experiments in the greenhouse and on a back barrier island (Apron Island) in Charleston County, where we looked at plant performance relative to origin of the original seed source (Charleston County, South Carolina, and Kennedy County, Texas). The South Carolina plants had lower flowering rates in the greenhouse and higher survivorship rates on Apron Island than plants originally from Texas (Figure 5). Genetic research with material from these same populations indicated that the Texas plants were genetically different from the South Carolina plants, and we have thus identified ecotypic variation between plants collected from the eastern- and western-most sections of the species range. From a practical restoration and conservation perspective, it is not realistic to think that a sweetgrass restoration project in the Carolinas would use plant material from as far away as Texas, but we have shown that ecotypic variation does occur with this species.

The next step in providing collectable populations of sweetgrass for the local sweetgrass basket makers is to determine to what extent ecotypic variation occurs in native populations of *M. sericea* along the historical range of the Gullah corridor. In addition to plant ecological research, researchers at the College of Charleston and Clemson University are presently collecting data to understand better the historical and cultural resources and land use present in the Gullah/Geechee Heritage Corridor, with a focus on the Mount Pleasant, South Carolina area. By approaching the issue of collectable populations of sweetgrass for the basket makers of the Gullah Corridor, we are addressing modern, complex problems from a multidisciplinary perspective.

Contrary to what was the case in the tallgrass prairie and serpentine barrens restoration projects, there are no commercially available seed sources of a cultivated variety of *M. sericea.* There is, however, a growing ornamental container sweetgrass industry that is planting both *M. sericea* and *M. capillaris* in urban settings throughout the southeastern U.S. It is not clear what effect these ornamental landscape plants will have on the native genotypes, for example, if nonlocal genes from these landscape plantings will swamp the local native populations and what effect that will have on population persistence, or if these landscape plants will provide sufficiently high quality and quantity of plant material for sweetgrass basket makers. What is clear is that the southeastern United States is one of the fastest growing regions in the nation (U.S. Census 2000 and 2005), and urban expansion will likely continue to diminish the accessibility of historical sources to sweetgrass basket makers.

There is a growing need to establish collectable populations of *M. sericea* for Gullah communities along the Gullah/Geechee National Heritage Corridor. Attention should also be paid to reducing the destruction of existing native populations and to diminishing the impact of horticultural plantings of nonnative container plants as a result of urbanization. In this situation, establishing community gardens or planting with appropriate plant material should be the restoration/reclamation goal, however it is too early to know if the appropriate plant material is from the historical Gullah Corridor (Goal # 1), a coastal Carolina ecotype (Goal #2), or some source from Florida or the Gulf Coast (Goal #3).

Conclusion

In this paper, we have cited specific examples of ecological restoration projects that we have experienced firsthand in the Midwest, Northeast, and Southeastern U.S. We promote the use of locally collected plant material if available, but we have faced situations where matching ecologically the donor habitat with the restoration site was simply not possible. Under such circumstances, purchasing seeds of the desired species from native plant suppliers allowed us to use material from the plant adaptation regions (PAR). PAR combines USDA plant hardiness zones with the ecoregion system (epa.gov/wed/pages/ecoregions.htm) commonly used by nongovernmental organizations (NGOs) like The Nature Conservancy, and is supported by plant material test results conducted by academic and governmental agencies (Vogel et al. 2005). There are restoration situations in which historical ecotypes no longer exist—local populations having been long extirpated due to anthropogenic activities—and there are no commercially available seed sources originally from the same PAR. In such situations, we would advocate using nonlocal seed sources of the desired native species: Since a species range can be geographically and ecologically broad, it is better to plant a native plant community within the historical range of the component species rather than use nonnative species or fail to conduct any restoration activities on degraded or damaged lands.

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Table 1: IS YOUR GOAL TO…?

Clearly defining your goals or objectives is an essential first step in any successful restoration project. These questions will help organizers prioritize activities (seed collection, field work, etc.) and more effectively manage limited labor and financial resources. In addition, articulating the goals of the restoration project will better allow you to assess the progress of the restoration project. We suggest that the practitioner ask: is the GOAL IS TO ESTABLISH

1. a historical plant community that includes the historical genotypes?
2. a historical plant community that includes “local” ecotypes?
3. a historical plant community, but whose selection of seed sources is not as important as selection of plant species that historically occurred in that community?
4. a plant community with native species that are likely to thrive under current ecological conditions, but not necessarily species that historically occurred there?
5. a plant community with non-native species that is not restoration? (Such activities should not be defined as a goal for a restoration project.)

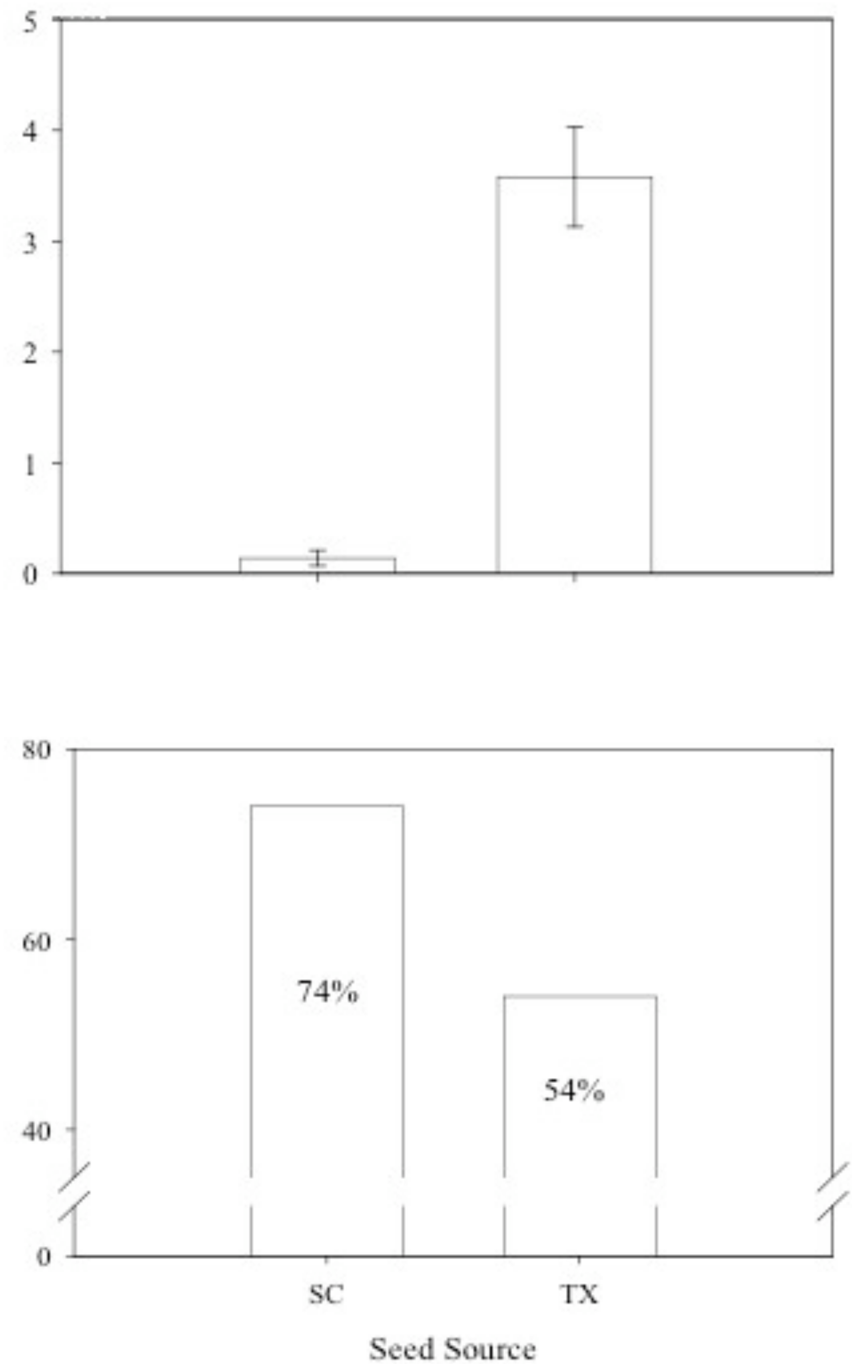
Figure 1: *Top*: Weston Cemetery Prairie, McLean County, Illinois, in the summer of 1977. This remnant five-acre pioneer cemetery prairie is surrounded on three sides by row crop agriculture and a railroad right-of-way leading to the grain elevator in the background. Fire management is used to reduce woody cover and promote species-rich forbs and grasses. ***Bottom*: Railroad prairie located 16 miles west of Madison, Wisconsin, taken in 1964.** These long, linear-shaped remnant prairies have been historically maintained by fire management of woody species by the railroad companies, however modern vegetation management uses herbicide. Removal of the natural fire regime is resulting in encroachment by fire intolerant woody species and the loss of fire dependent prairie species. **(Photos by Roger C. Anderson)**

Figure 2: The globally endangered barrens aster, *Symphyotrichum depauperatum*, at Nottingham Serpentine Barren, Chester County, Pennsylvania. Note the shallow soils and serpentinite rock typical of eastern serpentine barrens. **(Photo by Danny J. Gustafson, October 2002)**

Figure 3: Traditional Gullah basket displayed at the Sweetgrass Cultural Arts Festival in Mount Pleasant, South Carolina. (Photo by Angela C. Halfacre, June 2006)

Figure 4: Characteristic *Muhlenbergia sericea* habitat on front barrier (*Top*: Dewee's Island) and back barrier (*Bottom*: Apron Island) islands in Charleston County, South Carolina. This species can occur in the interdunal troughs between the established dune communities, areas without significant woody vegetation, and in the ecotone between the salt marsh community and maritime forest. The typical flowering period for *Muhlenbergia sericea* along the South Carolina coast is from the middle of October through November. **(Photos by Danny J. Gustafson)**

Figure 5: In a 6-month greenhouse common garden, 34 of 50 plants from Texas flowered while 1 out of 50 South Carolina plants flowered (χ^2 = 60.14, d.f. = 1, P < 0.0001) (*Top*). These same plants were transplanted to Apron Island in October 2005, and plant survival was recorded in May 2006 (*Bottom*). South Carolina plants had higher survivorship than plants from Texas.



A Call to Establish a National System of Regional Seed Banks and Seed Networks

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The success of a restoration project depends on many factors, but most critically on the selection of appropriate native plant materials from appropriate genetic sources, on utilizing proper genetic sampling, and on cultivation of propagules. More often than not, the approach to securing appropriate native plant material for restoration and management projects is outdated, flawed, and haphazard. Although large land management agencies such as the U.S. Forest Service, National Park Service, Bureau of Land Management, and some state agencies have their own in-house plant development services, their appropriateness and effectiveness vary widely. Other agencies have no such services, and options are generally much more limited for municipal, state, and regional projects and programs.

Plant procurement for many of these entities is largely dependent on a local patchwork of private-sector nurseries and seed companies in business in a given area and, in many if not most instances, on what materials they have in stock. Some projects utilize custom growing, but generally the seed source is only imprecisely specified, if at all. In New York City, for example, I have never seen a specification that required scientific methods for seed collection, which would ensure good genetic diversity of the source seeds. Furthermore, if collections are initiated only at the time of procurement, one or more additional years may be needed to collect the seed or other propagation materials before plant production can even begin. On a practical level, this means most projects are using restoration materials propagated from seeds or plants already in the hands of nurseries and seed companies, regardless of their origin or appropriateness. If the restoration material is being procured late in project development or implementation—or on an emergency basis—the source of the material may be hundreds of miles away, from very different climates and ecological zones, and may even be from horticultural stock.

In some regions of the country and in some public agencies, there are programs that follow more integrative methods, but these are individual examples that do not yet represent a national trend.[1] I base my judgment on over twenty years of experience, first as an urban land manager and for the last decade as director of a municipal native plant nursery and seed bank. My view reflects the situation that I perceive in the urban Northeast and may differ considerably from opinions of those in other regions and at other levels of government. However, I think readers from across the United States will benefit from the practical recommendations I present here. Indeed, it would be a worthwhile outcome of this paper if readers responded by citing other well-formulated programs. It is my hope that this article will open up a broader discussion of seed and plant procurement policies since there is a critical need for initiatives that will lead to improved practices.

Efforts to plan systematically for future seed needs mostly come from large federal agencies such as the U.S. Forest Service or Bureau of Land Management. For instance, the Forest Service requires "each National Forest to develop and implement a Ten-Year Procurement Plan" for tree species (USFS 1998). I am aware of only one such effort at the state or municipal level, although others probably exist but are not well documented and are difficult to survey.[2] It would seem that the role of the public sector in securing genetically appropriate source material is not as widely considered as it should be. Furthermore, the examples that I am aware of are not all well coordinated to ensure that appropriate plant material is available when needed and in the quantities required by the projects and programs served.

There is extensive scientific literature documenting the importance of protecting the amount and integrity of genetic diversity in local plant populations from the introduction of novel genes via restoration materials. The significant negative consequences of these translocations have been demonstrated in principle and are excellently summarized in Hufford and Mazer (2003), Rogers and Montalvo (2004), McKay et al. (2005), and others.

There are two concerns regarding the appropriate choice of plant materials for restoration: (1) the likelihood of success of the project, and (2) the impact on neighboring native plant populations, if any. Regarding the latter concern, if many plants are used but represent only a small amount of genetic diversity—for example, if there are cuttings from one plant or clone—they could cause over time the genetic diversity in neighboring populations to decline (this is called genetic erosion). Alternatively, if the restoration materials were not well adapted they could, nevertheless, undermine the adaptations of nearby natives over time.

Scientifically Sound Methodologies

Selecting genetically appropriate sources for restoration materials is not a simple undertaking. Furthermore, the process differs depending on the management objectives and the size and context of the restoration project (e.g., vast landscape vs. small urban area). Rogers and Montalvo (2004) suggested a methodology for selecting genetically appropriate source material for a project site, utilizing a ten-step decision tree applied to each species under consideration. Johnson and Roy (draft) have attempted to simplify this process by utilizing a Genetic Effects Rapid Assessment Matrix. Both of these methodologies attempt to tackle the pragmatic dilemma project managers face of how far they can go off-site for plant material without genetically compromising on-site and adjacent plant populations (a so-called safe seed-transfer zone). Ultimately, a practitioner applying these methods could arrive at some definition of an acceptable seed-transfer zone or seed-transfer strategy for each species in that particular project, leading to some a reasonable assuredness that they have answered their question and protected their resources. (In highly urbanized sites with no biological connectivity between populations, land managers may not need to go this far. Ensuring that proper seed collections for genetic diversity have been made from locally adapted seed sources may be enough to ensure long-term sustainability of their restoration and management efforts on genetically isolated populations. However, they will still need to be concerned with the effects of those translocations on any extant remnant populations.)

In order to construct seed-transfer zones, and in the absence of direct genetic information about species selected for restoration, both the Rogers and Montalvo (2004) and Johnson and Roy (draft) methodologies utilize ecological, life history, and other biological species data. Much of this data is not easily accessible to non-academic project planners, or may not yet exist. To apply either of these methodologies on a species-by-species basis for all of the species intended for a project would, in most instances, exceed the time constraints that virtually all projects face. Rogers and Montalvo (2004), among many others, wisely counsel that sufficient lead time for planning is crucial if these issues are to be addressed. For many projects, this is a luxury rarely enjoyed. A project planner or manager looking to apply these methods while working unaided would find it hard to assemble sufficient information to make critical seed source choices. Those with access to academic resources may find it less problematic. In any case, even if they could successfully perform these evaluations, the lead time needed to collect the proper seed may already have passed.

We need to begin to come to grips with the complexities of these issues, propose steps to take, and reach pragmatic solutions to their implementation. I believe that in building upon the foundation of existing programs we have the means of implementing the necessary policies, practices, and bureaucratic frameworks to do so.

Practical Alternative Solutions

So how might we reasonably approach these issues and take positive steps to resolve them? I submit two proposals for consideration. Both advocate for regional efforts and a strong public-sector role.

First, as a pragmatic solution that would result in immediate improvements, I propose a national system of regional active seed banks to dramatically increase availability of local seed. Second, I propose the simultaneous establishment of regional seed networks—geographically identical in scope to the regional seed banks—to address the issues of seed-transfer zones and to provide a bureaucratic framework for regional cooperation and cost sharing. These proposals are diagrammed in Figure 1.

Regional Active Seed Banks

Initially, in the absence of seed networks and local seed-transfer zones (which arguably would take time to establish and yield practical results), we can still vastly improve upon most current seed-procurement practices by investing in regional seed banking. In this scenario, practitioners would not yet employ the complete methodology of Rogers and Montalvo (2004) or Johnson and Roy (draft) to select seed source for their project. (I would still recommend that project planners familiarize themselves with them, as they are crucial next steps in evolving appropriate strategies.) They would, however, make use of sources as close to their project site as is practical and would exercise much greater control over the process. This conservative approach is perhaps the closest approximation for the moment to the Rogers and Montalvo (2004) methodology and a clear improvement over the largely random process that currently exists in many places.

Let me first distinguish between active seed banking and long-term or conservation seed banking. Conservation seed banking is what most of us think of when we hear the words “seed bank.” In this scenario, seeds—most commonly of species of conservation concern—are dried to low relative humidity, hermetically sealed, and then stored at low temperatures (typically -18°C) as a hedge against their loss in natural habitats. This has been commonly referred to as a "Noah's Ark" approach to conservation. Facilities to store this seed safely, for perhaps hundreds of years or longer, cost in the millions of dollars, and are mostly run as national or international institutions. Two examples are the Millennium Seed Bank Project (MSBP) of England's Royal Botanic Gardens, Kew, and the USDA National Center for Genetic Resource Preservation in Fort Collins, Colorado.

Active seed banking entails a shorter storage period, under similarly low relative humidity conditions, but at only moderately low temperatures (5–12°C), which guarantees seed viability for perhaps decades at a time, long enough to serve the needs of supplying local restoration and management projects. Costs for these types of facilities are much more modest than those of conservation seed banks. At the Greenbelt Native Plant Center in New York City we have established, with the aid of MSBP, an active seed bank at costs only in the tens of thousands of dollars. The purpose of establishing such a facility is much more analogous to a true bank, where seed can be withdrawn by depositors as needed. [See Cromarty et al. (1990 revision) and CPC (1994).]

What would a regional seed banking effort look like? First, collections would be made only by properly trained technicians to make certain that the maximum genetic diversity of a population is being captured in the collection and that established collecting protocols, such as those from the national Seeds of Success (SOS) program, are followed. Second, collections would be accessioned, entered into a database, and maintained individually so that the seed bank would truly be a repository of local collections that could be utilized for local projects. In this way, seed could be collected and stored in preparation for specific local projects, to be withdrawn when the time came to begin propagation. This would go a long way toward enabling the use of local seed by ensuring a ready supply. (Once the seed bank is in place, local agencies and organizations would also be better motivated to plan ahead for future needs, since a clearer pathway to implementing sound practices would be in place.) Seed would be collected only for the specific projects and programs that have partnered with the seed bank, and the banked seed would be theirs. The partners themselves would then provide their seed to commercial nurseries and seed companies to contract grow for them, and not for general sales or release. With the costs of collecting, processing, and storing shared regionally among all of the partners in the seed bank, costs could be kept relatively low, and individual agencies or organizations within the region would not need to invest in the staffing and infrastructure required if doing these tasks alone.

Such efforts must be properly managed so that these resources are neither squandered nor misused, and collection is not detrimental to source populations. Seed must also be fairly distributed to partners. Methodology must be developed to anticipate future needs far enough in advance so that the necessary seed resources are available and can be provided to the facilities, largely in the private sector, that will produce the required plant materials for the specific projects and programs. This last step must also be within a controlled and monitored framework that guarantees the verity of the plant and seed end products. The public sector should control the seed resources of a region, seeing to their conservation and effective utilization.

On a national level, I am aware of two large-scale government efforts that could form the basis of a system of regional active seed banks. The current network of 26 participants in the national Seeds of Success program, with their regional expertise and population-genetics-based approach to seed collection and conservation, could act as the national base for this program, to which more regional partners could be added as needed. Already aggregated since 2001 into a national framework of cooperation, their relationship is soon to be formalized in a memorandum of understanding and their national role expanded.

Additionally, there are the 27 regional Plant Material Centers of the USDA National Resource Conservation Service. Although their history and mission lie in plant improvement and selection, including that of native species, their substantial knowledge of plant genetics, seed collecting and banking, seed technology transfer, and the production of source-identified seed would make them invaluable partners in this effort. Their mission would need to be expanded to function as regional active seed banks, or, if not, then to assist others with their various areas of expertise. I would strongly advocate that such possibilities be explored.

As many as fifty or sixty well-trained centers drawn from these two sources, or newly formed by others, could easily form the nucleus of a nationwide network of regional active seed banks, each focusing on its individual area, but certainly drawing synergistically on the effort of the others.

The Greenbelt Native Plant Center has recently taken steps to offer our active seed bank as a regional resource. We are working with groups on Long Island and in the Catskill region to collect cooperatively and bank their local seed resources for future use. We anticipate continuing to expand on these efforts.

Regional Seed Networks

As useful as a national system of regional seed banks would be, I envision their creation as only a pragmatic first step. The concept should be expanded to include regional seed networks. These networks would serve as a bureaucratic framework for interorganizational management of the region's seed resources and seed bank and would also pool the scientific and technical expertise of the region to address seed-transfer zones and any other scientific or technical issues as they arise.

Through a regional seed network all of the interested parties could come together, discuss their needs, issues, and limitations, and give shape to a cooperative effort. In this way, the region's seed resources could be managed so as to conserve them and to provide an adequate and timely supply of local seed to meet the cooperator's needs. Ideally, all of the relevant local, state, and federal agencies together with the principal nongovernmental organizations (NGOs) that are jointly responsible for most of the restoration and management activities in the region should be encouraged to participate in the regional seed network as members. This is critical in several respects—first, in building a comprehensive picture of the region's seed needs; second, to cost share the seed banking operational expenses among as many partners as possible, lowering the cost for each participant; and third, to make sure that most of the region's seed resources, which in many regions of the country are largely on public lands or locked up within private conservancies, are accessible to the seed bank for collection and storage. Such an arrangement would be formalized as a memorandum of understanding between all of the participants. A collection MOU should include provisions for environmentally sensitive collection practices and ways of sampling seeds/propagules to ensure good genetic representation in the samples. The networks would meet periodically to set policy and discuss larger organizational issues. To streamline operational decisions, a more limited governing council would be appropriate for more regular meetings.

So that a realistic assessment of seed needs can be made, I propose that a regional registry of projects be established. This would be a definitive list of network cooperators' planned projects and/or ongoing management needs over the next five to ten years, with information about species, quantities, projected start dates, etc. This information would be critical to planning and staging seed collection operations for the region and guiding the regional seed banks about where to concentrate their efforts, and would be continually updated. The network and its governing council would also prioritize collections, setting target species and determining the most critical needs.

The seed networks would pool regional resources to establish protocols and assemble the information needed to determine local seed-transfer zones within the region. Pooling from the region's universities, colleges, botanic gardens, arboreta, natural heritage programs, NGOs, plant societies, and even interested, trained, and skilled volunteers outside of these institutions, the networks could assemble teams to work on various aspects of the scientific and technical questions that need to be answered to start to assemble seed-transfer zones. Pertinent questions involve an understanding of breeding systems, mating systems, ploidy states, and the like for each species. Some of the necessary information would be found by compiling existing literature. Some questions might entail research, such as common garden studies, which could be the basis of academic research or thesis projects. As a body of knowledge is assembled on the species found within the region, these scientific committees would be in a position to write species-specific protocols and perhaps even to make some generalized recommendations along the lines of Johnson and Roy's Rapid Assessment Matrix or as best management practices. A sidebar to this paper contains useful starting points for how to go about the process of assembling this information.

To aid these regional efforts, and because many species will be of common interest from region to region of the country, I further recommend the establishment of a national database as a repository for all of the species-specific ecological genetics data needed to make seed-transfer zone decisions, an idea already proposed by Rogers and Montalvo (2004) in chapter ten of their work. As this information is acquired, it would be added to the database for anyone to access. This would avoid time-consuming duplicated efforts and would greatly facilitate utilizing either the Rogers and Montalvo (2004) or Johnson and Roy (draft) methods. Such a database could be part of the USDA PLANTS Database website (plants.usda.gov) or could be hosted by a national organization such as the Plant Conservation Alliance. An example of the types of information that should be part of the database is found in Tables 10.1 through 10.8 of Rogers and Montalvo (2004).

As an example of a local initiative, the Greenbelt Native Plant Center has begun a collaboration with the science staff at Brooklyn Botanic Garden and plant ecologists from the NYC Parks Department in an effort we are calling the New York City Native Plant Conservation Initiative. We plan to map extant plant populations and examine the degree of biological connectivity among them (particularly which of these populations are within effective pollen and seed dispersal distances). Based on this analysis, we will determine protocols, including possible seed-transfer zone recommendations, for the management and long-term health of these populations. We will also be looking at whether opportunities exist within the urban matrix to increase the connectivity of some of these populations. Ultimately, seed-transfer zone decision making for any project or program will have to take place at a similar local level of individual restoration and management projects, but the information and groundwork done at a regional level will greatly facilitate the task.

I reiterate that, for the successful implementation of these recommendations, the products of these efforts, most concretely the regionally banked and reserved seed, must be shared and available to all interested parties, including the private nursery and seed industries, in a manner that is equitable while at the same time protective of the seed and genetic resources. Even well-documented and banked material can be deployed in an inappropriate way, and it will be important to educate seed network members to handle seed deployment appropriately.

Clearly the establishment of networks and seed banks and the support of their operations will require substantial funding. But the need is real, the payoffs are monumental, and the consequences of ignoring these issues any longer are too devastating, as our ecosystems face the cataclysmic consequences of biological invasion, climate change, habitat fragmentation, and irreversible harm to the genetic integrity of local plant populations. The time to act is now. We have considerable resources to begin the process, and we can build from this base.

Acknowledgments

I would like to acknowledge Dr. Arlee Montalvo for her tremendous help and advice in preparing this paper and for her consistent willingness over more than ten years to help this bureaucrat understand a difficult but critical subject.

Endnotes

[1] See, for example, the Los Angeles River Master Plan Landscaping Guidelines and Plant Palettes at http://ladpw.org/wmd/watershed/LA/LAR_planting_guidelines_webversion.pdf; the Native Seed Network in the Willamette Valley in Oregon; and also the Iowa Ecotype Project.

[2] The California Department of Forestry and Fire Protection collections from California tree seed zones on about a ten-year cycle. See: http://www.fire.ca.gov/ResourceManagement/PDF/Nurseries.pdf.

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Sidebar: Obtaining Genetic Information on Native Plants Useful in Restoration Decisions

Information about the ecology and genetics of many species of plants can be found in academic, applied, and government publications online as well as within existing online databases. When direct genetic information is not available, some information on a plant's characteristics (such as its breeding system, life form, and means of dispersal) can offer insights into its genetic characteristics. For example, the Fire Effects Information System (FEIS) database provides extensive reviews of the general biology, ecology, and relationship to fire of nearly a thousand plant species. Many of the reviews contain some basic information important to selecting sources of plants, including information on life form, elevation, habitat affinities, regeneration after fire, geographic distribution, taxonomic synonyms, and establishment.

There are a number of valuable search tools and databases available to the public to search for information on individual species. College, university, and botanic garden libraries are excellent resources for online and hard-copy publications. Much published literature is available online outside of libraries and can be found with the help of electronic search engines designed to find papers published in the scholarly literature. In addition, much of the literature cited in a database search is now available online.

Where to begin?

When starting a search, one place to begin is the USDA PLANTS Database. This resource provides standardized information about the vascular plants, mosses, liverworts, hornworts, and lichens of the U.S. and its territories. The database is searchable by scientific or common name and provides taxonomic synonyms, plant distributions, wetland status, and links to a variety of databases. Also consult the Integrated Taxonomic Information System to check on nomenclature. The online version of the Flora of North America, although not complete, may also be consulted for recent taxonomy and distribution information for many species. The list of synonyms generated from this exercise is important because much important ecological genetic information can be found in older publications when a search includes older plant names. The FEIS database also includes synonyms as well as basic information on botany and fire ecology.

It is also useful to consult local floras whenever they exist. Some are online—for example, the New York Flora Atlas. Some floras include information about the ecology, cytology, geographic distribution, and if there is substantial morphological or known genetic variation within species. Each state may also have information on rare plants, and native plant societies sometimes publish useful information. For example, the website for the California Native Plant Society Rare Plant Program publishes its Inventory of Rare and Endangered Plants online, as does the Colorado Natural Heritage Program.

For a detailed search, you can pair names of species with keywords or phrases for the type of information desired. Some useful keywords and key phrases include:

1. Reproductive mode, natural regeneration, soil seed bank, seed dormancy, seed longevity,

resprouting (for regeneration capability after fire, flood, or other damage)

2. Clone, rhizomes, asexual propagation, asexual reproduction

3. Seed type, seed morphology, seed dispersal mechanism

4. Life-history, parity, annual, perennial, biennial

5. Pollination, pollinators

6. Gene flow, pollen dispersal, seed dispersal

7. Breeding system, mating system, selfing, outcrossing, mixed mating

8. Ploidy, chromosome number, cytotype

9. Local adaptation, population differentiation, geographic variation, population structure

10. Inbreeding depression, outbreeding depression, inbreeding, outbreeding, heterosis

11. Hybridization

Detailed searches of species names and topics can be made using online search engines. Google Scholar is available to all, and though not thorough, can come up with some useful information. Most botanic garden, university, and college staff and students have access to a variety of journals online through their libraries and to powerful searching programs such as BIOSIS, AGRICOLA, CAB Abstracts, Digital Dissertations, and Web of Science. USDA employees have access to most of the library search programs through DigiTop on the website of the National Agricultural Library.

Once your citations are found, the text can often be found online. All volumes of over a dozen botanical journals (including the *American Journal of Botany*, *Applied Vegetation Science*, *Ecology*, *Systematic Botany*, and *Systematics and Geography of Plants*) and two dozen ecological/evolution journals (including *Conservation Biology, Ecological Monographs, Evolution*, and *American Naturalist*) are available online from JSTOR, an Internet archive for scholarly journals. JSTOR journals can be searched from the JSTOR site by typing in plant names, title words, keywords, or phrases into the search queries. A list of citations will appear, and the papers can be accessed by clicking on the citation. The JSTOR site provides lists of institutions and agencies that have subscriptions (including many public libraries, education institutions, and agencies), and individual subscriptions can be obtained easily. In addition, many professional societies and publishers of journals, including most genetic journals, have made issues available online with a subscription. Some, such as the *The Journal of Range Management*, make back issues available without a subscription. E-journals.org lists a database of online botanical journals. Botanical gardens are also great resources for information. Brooklyn Botanic Garden's website provides links to its library and herbarium resources and information about its library resources.

Online Resources

Atlas of the Vascular Plants of Utah: www.gis.usu.edu/Geography-Department/utgeog/utvatlas

Brooklyn Botanic Garden: bbg.org

California Native Plant Society Rare Plant Program: cnps.org/cnps/rareplants

Colorado Natural Heritage Program: cnhp.colostate.edu

E-Journals, Electronic Sites of Leading Botany, Plant Biology, and Science Journals: e-journals.org/botany/

Fire Effects Information System: www.fs.fed.us/database/feis

Flora of North America: hua.huh.harvard.edu/FNA/volumes.shtml

Grass Manual on the Web: herbarium.usu.edu/webmanual

Integrated Taxonomic Information System: itis.gov

JSTOR: www.jstor.org

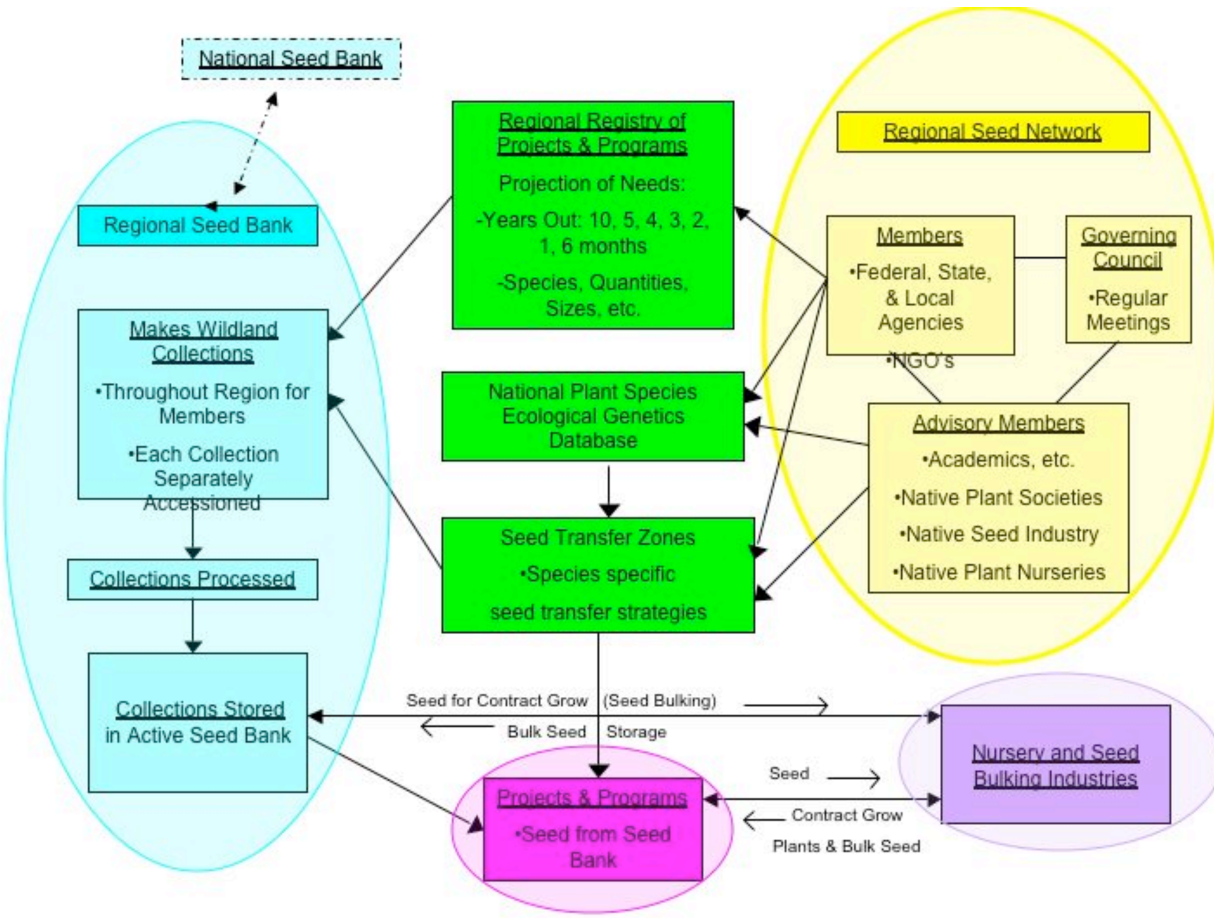
National Agricultural Library, DigiTop: nal.usda.gov/digitop

New York Flora Atlas: atlas.nyflora.org

USDA PLANTS Database: plants.usda.gov

U.S. Forest Service Native Plant Materials Policy and Authorities: www.fs.fed.us/wildflowers/nativeplantmaterials/policy.shtml

Figure 1: Flow chart showing relationships among all proposed entities and activities in the native seed procurement and production equation.



NATURAL HISTORY

Elevated Ozone Levels May Lead to Strengthened Invasive Species in Urban Forests

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Without notable advances in the protection of forests against the stresses caused by air pollution and the advance of invasive species, it is likely that dramatic diebacks and species shifts will occur in urban ecosystems (Mann et al. 1998; Fowler et al. 1999; Walther et al. 2002; Woodward et al. 2004; Carreiro and Tripler 2005; Webb et al. 2006). Yet very few scientists have concentrated on both air pollution and nonnative species and how they combine to weaken forest ecosystems. In an effort to delve deeper into this issue, the Department of Environmental Studies at the University of Virginia has undertaken an ongoing study of the effects of ground-level ozone (O_3) on mid-Atlantic urban forests. Their preliminary data suggest that some important native species (e.g., *Acer rubrum*, *Liquidambar styraciflua*, *Celtis occidentalis*, *Quercus rubra*) are less equipped to defend against oxidation reactions generated from elevated ozone levels than common invasive species (e.g., *Ailanthus altissima*, *Morus alba*, *Paulownia tomentosa*).

Ground-level, or tropospheric, ozone is the main constituent of industrial smog; the gas is formed by a chemical reaction of nitrous oxides (nitrogen dioxide, nitric acid, nitrates, nitrous oxide) and volatile organic compounds (benzene, formaldehyde, toluene) in the presence of sunlight.

When the stomata of a plant are left open for gas exchange, ozone enters the stoma cavity and oxidizes the mesophyll cell wall, creating increased permeability of the cell wall and making the cell more vulnerable to injury. Oxidation damage in a leaf results in decreased plant performance and growth after repeated exposures. The UVA study has so far focused on leaf injuries caused by ozone damage in native and invasive trees at nine forested sites within three major East Coast cities, Washington, D.C., Baltimore, Maryland, and Philadelphia, Pennsylvania.

Within each city, UVA scientists chose three separate sites on the basis of their relative annual concentration of ground-level ozone, with thresholds set at: low (0–79 ppb), medium (80–99 ppb), and high (100–125+ ppb). In determining these concentration thresholds, the scientists assumed that the low levels would not injure plants, medium levels would injure only ozone-sensitive plants, and high levels would harm all but the most ozone-tolerant plants. They also assumed that high pollution levels would selectively eliminate pollution-intolerant species while augmenting establishment of pollution-tolerant species; and that forests in low-pollution areas would not have experienced this disturbance and would therefore retain native intolerant species, thus reducing the number of invasive plants that were able to establish there (Tillman 1994).

The study's preliminary results indicate that as ground-level ozone concentrations increase in forest settings, the native flora presence decreases, while the density of invasive species actually increases. One example of this trend shows the density of the common invasive tree of heaven (*Ailanthus altissima*) increase in abundance from low sites to high sites, while the native green ash (*Fraxinus pennsylvanica*) decreases in overall abundance. Early data also suggest that the native species studied have a greater incidence of ozone-induced oxidative injury than the invasive species. Across all sites, the plants at low-ozone-level sites experienced less overall injury than those at high concentration sites.

Native species are known to be less adaptive to changes in environmental conditions, while invasive species often cope with changes by reallocating resources and out-competing native counterparts. The ruderal nature of most invasives gives them an inherent advantage over those natives that are not able to adapt to phytotoxic gas increases.

It is clear that some plant species exhibit more tolerance to oxidative damage than others (e.g., *Prunus serotina* is more tolerant than *Fraxinus americana*) (Schaub et al. 2003), which may be due to the function of leaf chemicals (Pell et al. 1999; Massman et al. 2000). The most important chemicals in determining the injury a plant will suffer are antioxidant chemicals. Some species produce large amounts of antioxidant chemicals (e.g., ascorbate, glutathione, superoxide-dismutase, peroxidase, polyamines, carotenoids, α-tocopherol), which may decrease initial injury in active oxygen species and reduce the recovery time from injury (Massman et al. 2000; Eltayeb et al. 2006). I plan to undertake a comparison of the leaf chemicals of congeneric native and invasive plants, as well as those of invasives and natives found in the same wooded locales. Developing an understanding of how leaf chemicals are involved in oxidative injury will illuminate their role in the interplay between air pollution and invasive ecology.

The EPA is currently considering amendments to national air quality standards for ground level ozone levels in order to address the issue of vegetation damage. Dating back to 1985, European agencies like the United Nations Economics Commission for Europe established the International Co-operative Programme Forests group to prevent another major forest dieback (UNECE 1988; Ashmore and Wilson 1994; UNECE 1999; EU 2002). However, the EPA's proposed work with regard to ground-level ozone and the European resolutions addressing ozone-forest interactions fail to recognize and assess the role of invasive species, as these issues are commonly dealt with separately. The UVA study clearly suggests that increased levels of air pollution lead to increased populations of invasive species—tying the two issues together quite strongly. In order to ensure that our forests persist intact into the future, scientists and regulators must begin to tackle the hazards of air pollution and invasive species in tandem.

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Figure 1: A typical edge of the nine sites in the study. The pictured site is the Baltimore, MD high ozone location. (Photo by Eric Elton)

Figure 2: Nine urban forested sites are used to compare ozone damage along an ozone level scale.

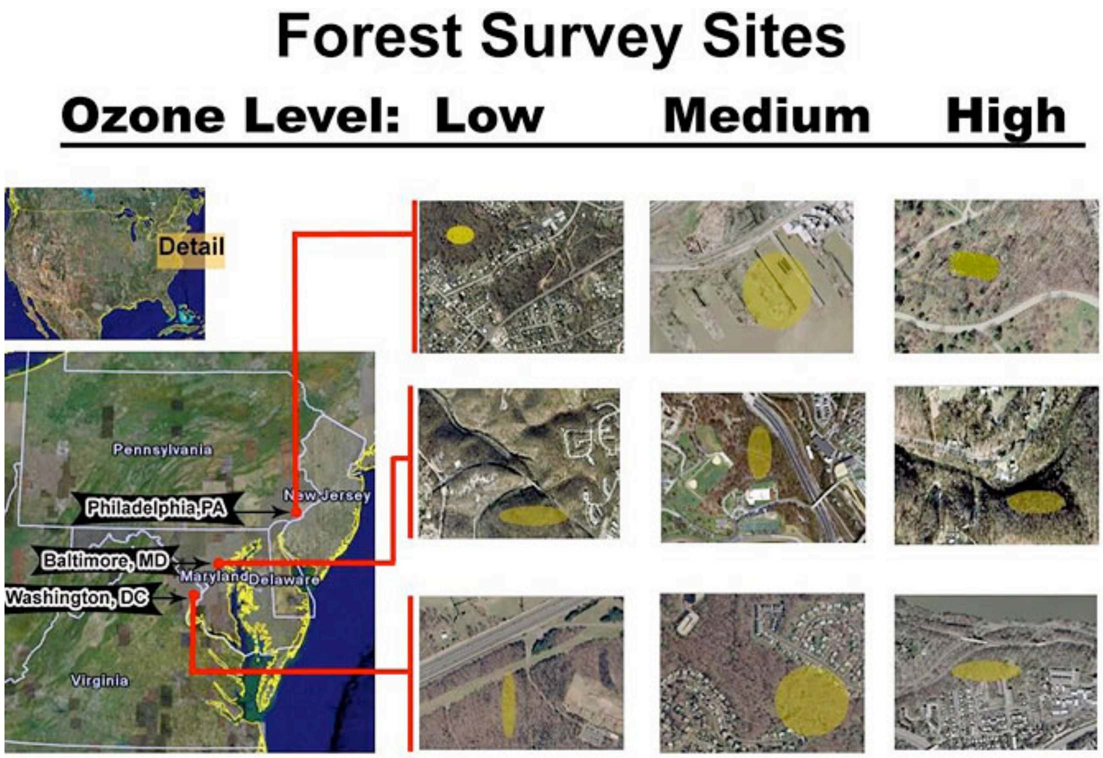

Figure 3: *Prunus serotina* showing typical stipple damage due to high levels of ozone exposure. (Photo courtesy of Schaub, M., Jakob, P., Bernhard, L., Innes, J.L., Skelly, J.M., Kräuchi, N. 2002. Ozone injury database. http://www.ozone.wsl.ch. Swiss Federal Research Institute WSL, Birmensdorf.)

Impacts of Urban Runoff on Native Woody Vegetation at Clark Reservation State Park, Jamesville, New York

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Abstract

Since 1989, Dry Lake, a unique basin located in the New York state park Clark Reservation, has experienced periodic flooding of silt-laden water from the adjacent Doubletree residential housing development. This study examines the effect of urban runoff on the native woody vegetation of the flooded zone. We selected two species of trees from both the inundated and non-flooded zones for growth studies: bitternut hickory (*Carya cordiformis*) and sugar maple (*Acer saccharum*), cored sample trees at breast height (1.37 meters) on the north and south faces, and measured the ring widths of the cores to the nearest 0.001 millimeter. Our statistical comparison of tree ring-width indices measured before and after Doubletree construction commenced showed that bitternut hickory growth has not been affected by flooding (P = 0.701). In contrast, sugar maple trees sampled from the flooded zone exhibited increased ring-width indices in the 12 years after Doubletree development began (P = 0.06). We found that a third species on the site, eastern hemlock (*Tsuga canadensis*), is intolerant of flooding, and all trees of this species in the flooded zone died.

Key words: Clark Reservation State Park, urbanization, tree ring, WinDENDRO, sugar maple (*Acer saccharum*), bitternut hickory (*Carya cordiformis*), eastern hemlock (*Tsuga canadensis*)

Introduction

Urbanization is considered one of the main causes of land-use and land-cover changes. As a consequence, urbanization has affected the structure, dynamics, and functions in a variety of ecological systems (Luck and Wu 2002). Urbanization also has significant hydrological impacts, including effects on the processes and rates of erosion and runoff (Goudie 2000; Weng 2001). The construction stage of the urbanization process is responsible for the highest erosion rates. During the construction stage, the removal of trees and vegetation leaves soil exposed, causing high rates of erosion. In addition, this stage has high levels of site disturbance (Goudie 2000). Construction sites can reach erosion rates of up to 163 metric tons per hectare per year (Krenitsky et al. 1998).

Clark Reservation State Park in central New York State contains glacier and karst topographic features and the vegetation they support at the lower end of a 146-hectare watershed. Nearly 101 hectares, or 66% of this watershed, is located outside the Park boundaries (Lendrum 1996). One of the Park's features is 4-hectare, meromictic Glacier Lake, which was formed as a plunge basin by runoff from continental glaciations (Van Diver 1980). Adjacent to Glacier Lake is a smaller closed basin known as Dry Lake, which in dry periods drains through the bottom of the basin (Figure 1).

Before the construction of Doubletree began, the Dry Lake basin usually held standing water for only short periods of time. There is evidence that water levels did not exceed approximately 3.66 meters in depth above the floor of the basin (Figure 2). However, after construction of Doubletree began, the basin was affected by siltation and excess water. As a consequence, water levels have reached as high as 8.5 meters above the floor of the basin (Figures 3 and 4). Many trees in the flooded zone have been affected by thick layers of ice that accumulate in the floodwater and damage the trees' bark, often down to the cambial layer (Figures 5 and 6). The flooding is caused primarily by siltation, which fills crevices in the ordinarily permeable surfaces of Dry Lake basin and the upstream watershed. This results in a greater volume of water flowing into the basin and remaining. The depth of silt and larger soil particles varies across the floor of the basin, with greater amounts near the inflow stream. Near the center of the basin, the rate of sedimentation over the first 13 years post-development of Doubletree has averaged 4.8 centimeters per year (Franco 2002). The surfaces of flood water freeze in the winter, forming thick layers of ice that damage the bark of flood-zone trees down to the cambial layer and result in death (Figures 5 and 6). Siltation and flooding have also affected herbaceous vegetation in the open part of the bowl, altering the community composition of the basin (Figure 7).

The recently proposed Hummel Estates Residential Subdivision would cover approximately 20% of the watershed of Glacier Lake (Figure 1) and presents a major concern for environmental impacts caused by future development. The Glacier Lake area consists of old field recovery from agriculture. A two-lane state highway runs from east to west between the Doubletree and proposed Hummel Estates development areas and the park, but culverts beneath the highway allow water into the park. A major unknown regarding flow from the proposed Hummel Estates into Clark Reservation is the ultimate destination of the water after it sinks into Dry Lake's floor of fractured bedrock and on a limestone shelf above Glacier Lake (Heisler 1996). Because of concerns about damage in the park, developers and the town planning board modified plans for Hummel Estates reducing the number of planned dwellings from over 80 to 32, placing most of the northern half of the property into a permanent no-disturbance parcel, and directing south much of the storm water from the developed portion into a through-flowing stream.

The purpose of our study was to analyze the impact of the Doubletree housing development on the vegetation of the Dry Lake ecosystem at Clark Reservation State Park in order to better inform future development decisions affecting the watershed area for Glacier Lake. The main objective of this study was to determine how sugar maple (*Acer saccharum*) and bitternut hickory (*Carya cordiformis*) in the Dry Lake ecosystem of Clark Reservation State Park respond to flooding by observing changes in annual growth rings of stem xylem from individual trees. We compared ring-width growth of specimens of both species from the flooded zone to those in the non-flooded zone before and after the construction of Doubletree. Our study updates and expands upon the research of Drew and Wink (1997; 1999) on impact to vegetation.

Background on the Park

Clark Reservation State Park is located in Onondaga County, 3.11 kilometers southeast of the city of Syracuse, New York, and 0.78 kilometers west of the village of Jamesville. It is approximately 141.65 hectares in size and its coordinates are 76° 05' longitude and 43° 00' latitude (New York State 2000).

The park contains unique limestone-bedrock geology that includes the 4-hectare, 18-meter deep, meromictic Glacier Lake created in a plunge basin 55 meters below the lip of a melt-water falls that existed in the last continental glaciation (Van Diver 1980). Other, smaller closed basins in the Park were created either by limestone solution or as plunge basins during periods of less flow or shorter duration than those of Glacier Lake. Dry Lake is a roughly circular depression about 2 hectares in size and approximately 12 meters deep. It is believed to be a karst feature created by dissolving limestone that formed a sinkhole basin. The bedrock is 300–400 million years old (Van Diver 1985) and its fissures allowed for rapid post-glacial water drainage.

Sydansk's 1936 thesis briefly describes the pre-Doubletree vegetation and ecological processes within the center of the Dry Lake basin. Sydansk describes the area as "a rather small inconspicuous bowl supporting dense herbaceous vegetation but quite free of woody plants. In the winter…when the soil freezes, drainage is reduced materially and the area" fills with water "forming a small lake, the surface of which freezes over." "The water beneath the ice slowly drains out…. The ice…sags and pulls toward the center of the bowl…breaking all vegetation encased in it." Sydansk also states that forest growth began at about 3.66 meters, allowing us to estimate the pre-Doubletree water level at about 3.66 meters above the floor of the basin. The forest community along the edge of Dry Lake basin was composed of Allegheny hardwoods (oak, hickory, tulip poplar), which predominated in early successional stages and later gave way to the northern hardwoods (sugar maple, American beech, yellow birch, Eastern hemlock, white pine) which comprised most of the area's supporting climax vegetation (Sydansk 1936). The vegetation today is very similar to the vegetation described by Sydansk.

At around 1940, the flora of Clark Reservation included 304 species of ferns and flowering plants (Egler 1943). The American hart's-tongue fern (*Phyllitis scolopendrium* (L) Newmn. var *americana* Fern.) was first sighted in North America in 1807 a few miles west of Clark Reservation (Cinquemani et al. 1986). In the United States, the American hart's-tongue fern is federally listed as threatened, and Clark Reservation State Park was founded in 1926 primarily to provide protection for the plant. About 70% of the U.S. population of hart's-tongue fern lives in Clark Reservation (Cinquemani-Kuehn and Leopold 1992).

Clark Reservation provides recreation for thousands of visitors each year. Since 1995, the New York State Open Space Plan has included Clark Reservation State Park as a high priority area for protection from outside influences. The plan states, "conservation measures are needed in areas outside and upstream from the park in order to protect these cultural environmental resources" (New York State Department of Environmental Conservation and the Office of Parks, Recreation and Historic Preservation 2006).

Methods

Drew and Wink (1997 and 1999, respectively) cored 12 sugar maple and 14 eastern hemlock trees, equally sampling the seasonally inundated and non-flooding zones. Trees were cored on the east face with an increment borer at breast height and cores were air-dried, mounted on strips of wood, and surfaced. Ring widths were measured to the nearest 0.001 millimeter and the cores cross-dated to determine the year of each tree's death.

On November 29 and December 9, 2001, we collected increment corings from one additional tree species and updated cores collected previously from sugar maple. These were chosen from the east side of Dry Lake. Twelve sugar maple trees were chosen, six from the seasonally inundated zone below the 7.9 meter high water line, which were the same individuals cored by Drew and Wink (1997), and six trees arbitrarily selected out of twelve trees from outside the flooded zone, up the slope above the known high water level. We measured only the undamaged circumference of sugar maple growth rings. We randomly selected twelve bitternut hickory trees, six out of the nine individuals in the flooded zone and six out of the ten in the non-flooded zone. All trees were of dominant or co-dominant crown classes.

We cored trees at breast height on the north and south faces, with care taken not to core directly above or below points of bark damage where a wound response effect could alter growth and bias diameter increment estimates (Drew and Wink 1997). We then measured the ring widths of the cores to the nearest 0.001 millimeter and also recorded the trees' diameter at breast height (BH = 1.37 meters).

We used WinDENDRO, an image analysis system specifically designed for tree-ring measurement and analysis, to measure individual annual ring widths. For greater precision, the analysis was revised by browsing the image for missing or false rings. The extraction of two cores per tree over six trees allowed for analysis of variation within and among trees on a site. Ring widths were averaged for each tree and converted to a ring width index for each year (Fritts 1976). Ring-width indices were derived and calculated as a ratio of actual ring widths and divided by the estimated ring width from the overall growth curve. One bitternut hickory was excluded from analysis because the growth rings were not prominent enough to measure. Paired t-tests with a 90% confidence level (0.1 alpha level), due to the small sample size, were used to make the following comparisons: average difference in ring-width indices from the two tree species between flooded and non-flooded zone, and comparison and average comparison before and after the construction of Doubletree (before 1989 vs. 1990–2002).

Results and Discussion

The previous work of Drew and Wink (1999) showed that six out of seven of the dead eastern hemlock trees from the inundated zone had died in the 1990s (Figure 4), post-Doubletree. The trees from the non-flooded zone appeared healthy, without any indication of abnormal or reduced growth. These findings are consistent with the ecology and habitat of the species. According to White (1973) and Whitlow and Harris (1979), eastern hemlock is considered intolerant of short-term flooding during the growing season.

Growth rate index of bitternut hickory increased from 0.997 ± 0.004 to 1.165 ± 0.197 in the flooded zone and from 1.002 ± 0.004 to 1.119 ± 0.233 in the non-flooded zone, although the difference was not significant (P = 0.701). Our results show that the construction of Doubletree has not yet affected the bitternut hickory trees in the flooded area of the basin; these results appear to be inconsistent with the ecology and habitat of this tree species. Whitlow and Harris (1979) and Loucks (1987) consider bitternut hickory an intolerant species—one that would suffer considerable injury if its soils were saturated for more than 30 days during the growing season.

In the case of sugar maple, growth rate index increased from 0.990 ± 0.004 to 1.275 ± 0.388 in the flooded zone, and growth rate index decreased from 1.001 ± 0.008 to 0.883 ± 0.212 in the non-flooded zone. The growth index of the trees in the flooded zone is significantly different from that of the trees in the non-flooded zone (P = 0.06). The increase in average ring width of the sugar maple trees in the flooded zone may be due the presence of fertilizers in the run-off water from Doubletree, acclimation of the sugar maple to the new wetter environment, or other unknown factors that may promote increased tree growth. Many sugar maples in the flooded zone have suffered ice damage to their cambia, and ring widths may have increased as a result of concentrated growth in the undamaged sections of the boles. Just as with bitternut hickory, the results obtained for sugar maple seem to be inconsistent with the ecology of this tree species. According to White (1973), sugar maple trees are intolerant of flooding and will not stand flooding for more than 10 days. Hall et al. (1946), Broadfoot and Williston (1973), and Whitlow and Harris (1979) also consider sugar maple a flood-intolerant species—defined as one that will not survive continuous flooding (of at least 0.3 meters deep of standing water) during significant portions of its growing season.

Conclusions and Recommendations

The construction of the Doubletree residential project on the watershed of Clark Reservation State Park has had a variety of effects on the park's ecosystem; thus far, effects are evident primarily in the Dry Lake basin. Late winter and spring flooding in certain areas caused the death of hemlocks present there. Hemlocks did not appear to suffer from ice damage, but trees of other species were killed in the flooded zone, with ice damage appearing to be a primary cause of mortality.

Bitternut hickory and sugar maple were the two tree species chosen for growth studies. Only sugar maple responded positively to flooding by exhibiting an increase in lower-stem ring growth. These findings support the results of Drew and Wink (1997), who reported an increase in ring widths of the sugar maple trees cored in a seasonally flooded zone. Bitternut hickory has yet to show any significant negative flooding effects on diameter growth rate, and bark damage was minimal.

Differing degrees of flood intolerance were exhibited by the three species tested. Our results show that eastern hemlock is the most flood-intolerant of the three. Hemlock trees in the flooded zone died shortly after construction began on Doubletree. Bitternut hickory appeared unaffected by the flooding. Sugar maple exhibited an increase in ring widths, an effect in direct opposite to our hypothesis. However, ice damage to bark of sugar maples has been severe. Therefore, we recommend continued monitoring of ring growth on bitternut hickory and sugar maple.

The post-development watershed impacts on the park ecosystem exceeded the effects of agricultural activities on the watershed, which date back more that 100 years. Thus, further development within the watershed, even if external to Clark Reservation State Park, will likely increase the impacts on the park's ecosystem. Eastern hemlocks have already died, and continued flooding has potential to negatively affect other species in the Dry Lake community, in addition to those addressed in this study.

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Table 1: Average of growth indices for individual bitternut hickory trees in the flooded and non-flooded zones of Dry Lake before (GIpre) and after (GIpost) the start of Doubletree housing development in 1989, and average difference in the observed growth indices (DIFF in GI).

	GIpre	GIpost	DIFF in GI
Bh-1	0.997	1.353	0.356
Bh-2	1.001	1.002	0.001
Bh-3	1.000	1.279	0.279
Bh-4	1.000	1.391	0.391
Bh-5	0.991	0.957	-0.034
Bh-6	0.993	1.007	0.014
Average in flooded zone	0.997	1.165	0.168
Bh-7	1.003	0.856	-0.147
Bh-8	1.002	1.195	0.193
Bh-9	0.998	1.148	0.150
Bh-10	0.998	1.148	0.150
Bh-11	0.998	1.148	0.150
Bh-12	0.998	1.148	0.150
Average in non-flooded zone	1.002	1.119	0.117

Table 2: A matched pair of t-tests for bitternut hickory using a risk level of 0.1.

Treatment	Average Diff in GI	STDEV	n-1
flooded	0.168	0.195	5
non-flooded	0.117	0.235	4

Tc		Tt (9, 0.1)
0.349	<	1.383

Table 3: Average of growth indices for individual sugar maple trees in the flooded and non-flooded zones of Dry Lake before (GIpre) and after (GIpost) the start of Doubletree housing development in 1989, and average difference in the observed growth indices (DIFF in GI).

	GIpre	GIpost	DIFF in GI
Sm-1	0.993	1.744	0.751
Sm-2	1.002	1.044	0.042
Sm-3	0.995	0.886	-0.109
Sm-4	1.001	1.027	0.026
Sm-5	1.004	1.171	0.167
Sm-6	1.001	1.781	0.780
Average in flooded zone	0.999	1.275	0.276
Sm-7	0.991	0.704	-0.287
Sm-8	0.999	1.235	0.236
Sm-9	1.003	0.811	-0.192
Sm-10	1.008	0.682	-0.326
Sm-11	1.011	1.027	0.016
Sm-12	0.993	0.837	-0.157
Average in non-flooded zone	1.001	0.882	-0.119

Table 4: A matched pair of t-tests for sugar maple using a risk level of 0.1.

Treatment	Average Diff in GI	STDEV	n-1
flooded	0.276	0.389	5
non-flooded	-0.119	0.211	5

Tc		Tt (10, 0.1)
1.993	>	1.372

Figure 1: Recent Google Earth view of Dry Lake and Glacier Lake, their watershed areas (dotted light blue), the nearly completed Doubletree residential development (solid red border), and the proposed development Hummel Estates (dotted red border). The detention basin for Doubletree and the intermittent outflow stream that connects to Dry Lake are shown in medium blue.

Figure 2: An aerial photo looking approximately southeast of Glacier Lake and Dry Lake in the mid-1930s (Sydansk 1936). Hemlock trees identifiable on this photo were found to have died, apparently owing to high water levels during the 1990s. The Dry Lake basin is marked "B." Note the agricultural use on the watershed, part of which extends off the picture to the right.

Figure 3: Flooding effects at Dry Lake basin; photo taken from the non-flooded area (April 8, 2001). The main source of inflow water is the small stream, Dry Lake Stream, on the left. It originates in the Doubletree detention basin.

Figure 4: Flooding effects at Dry Lake basin. Photo taken from the non-flooded area (April 8, 2001). The hemlock trees in the foreground and on the right have been killed by high water.

Figure 5: Some species including sugar maple in the flooded zone of Dry Lake were heavily damaged by ice, although they continued to live. Here we see severe damage by ice on a sugar maple tree in the zone that is flooded at high water, January 1, 1997.

Figure 6: Dry Lake in late winter 1995 at high water, 7.9 meters (26 feet) above the floor of the basin, and with ice cover. Brown needles of a hemlock tree recently killed by the flooding are visible near the center of the photo.

Figure 7: By midsummer, the Dry Lake basin is typically dry. The recent silt left by runoff, primarily from the Doubletree development, is visible. Trees with brown foliage were killed by high water or by bark damage caused by ice.

Increasing Interactions with Nature: A Survey of Expectations on a University Campus

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Abstract

In the last century, urban biodiversity has come under increasing pressure due to urbanization and consequent habitat destruction. Land-use patterns in the Cape Floristic Region of South Africa provide a strong example of this, and research has shown alarming decreases in natural vegetation cover there. Urban greening projects can play a vital role in conservation of biodiversity in the Cape Floristic region while simultaneously providing local people with an improved living and working environment (Cornelis and Hermy 2004). This article investigates the general attitudes of a sampled demographic in the city of Cape Town on the value of urban nature. I conducted a survey of personnel and students at the Faculty of Health Sciences of the University of Stellenbosch covering issues such as spare time utilization on campus and opinions and expectations regarding their study and work environments. Results showed that the overwhelming majority of respondents believe that their study and work environments need improvement, specifically as regards gardens and the natural environment. Furthermore, respondents indicated that should the school's gardens and natural environment be improved, their own attitudes toward their work and studies would improve. From the results of this study it is plausible to assume that the general urban public is in favor of urban greening projects, and this can, together with the input of conservation biologists, promote biodiversity conservation in densely populated areas.

Key words: biodiversity enhancement, fynbos, restoration ecology, urban biodiversity, urban greening, urbanization

Introduction

Abram (1997) is of the opinion that "nature…has become simply a stock of resources for human civilisation." It does indeed appear that for many potential key role players, from the individual landowner to highly structured government departments, conservation is of less importance than economic growth and development (Carlson 2005). This seems to be especially true in the urban setting, where development and urban growth take place at an increasing rate and the productive, cultural, recreational, educational, and conservation value of pristine land (cf. Jacobs 1999) may no longer be reason enough to conserve the associated biodiversity. Conservationists must constantly introduce new ideas and concepts in order to convince decision-makers to take the preservation of biodiversity into consideration when new areas are developed (Primack 2000).

One potentially fruitful method of convincing decision-makers of the importance of conservation within the urban setting would be to focus more attention on the positive effects that well protected and managed elements of a natural environment (e.g., trees, birds, insects, and plants) can have on the moods of their employees, students, or colleagues (Schoeman 1955; Abram 1997). Many public and private industries worldwide have accepted the importance of ergonomics—the study that aims to find the optimum conditions under which to achieve maximum productivity and work satisfaction (Bridger 2003)—for securing satisfying work and life conditions for their employees. However, until recently, the main factors considered by ergonomics were limited to the immediate environment, for example optimum temperature, light conditions, and noise levels within the office environment. Whether the natural environment (i.e., vegetation and associated biodiversity) influences an individual's work efficiency and performance in the same way that established ergonomic factors do needs to be investigated, as literature in this field is very scarce. Studies that partially relate to this line of research include Fredrickson and Anderson (1999), Hartig et al. (1999), Herzog and Barnes (1999), and Kerr and Tacon (1999).

The sensory experience (i.e., sound, sight, smell, and tactility) of one's immediate environment may take place unconsciously or deliberately, and it plays an important part in shaping a person's being and future (Hiss 1990). Even the way in which a person performs normal tasks is influenced by his or her physical and sensory surroundings (Schoeman 1955; Hiss 1990). This is exemplified by cases where poor working conditions cause low morale among workers, in contrast to cases where optimum working conditions result in higher productivity and a more positive attitude toward the work (cf. Edwards and Torcellini 2002; Heschong et al. 2002). Factors such as temperature, light, smell, noise, and the natural environment all help determine whether work conditions are valued as poor or good. In the same way that ergonomics can improve human effectiveness and enhance the quality of life in the work or home environment (Sanders and McCormick 1987), so unspoiled, well-managed natural environments can contribute to one's positive perception of a place and to an overall positive attitude (Hartig et al. 1991; Hartig et al. 1996; Hartig et al. 1999). A positive attitude toward the work environment could then directly influence the level of effectiveness in the work place (Norsworthy and Zabala 1985; Ries et al. 2006) (Figure 1).

Urban greening and nature conservation within urban areas has grown into an important consideration for ecologists, naturalists, and landscape designers since the early 1980s (Goode 1989), and urban green areas are now increasingly seen as an integral and important aspect of the urban ecosystem (Goode 1989; Li et al. 2005; Nilsson et al. 2007). The motives for implementing urban greening projects vary to a great degree, but mainly aim to meet social needs by allowing more frequent and readily available interaction with the natural environment and to balance infrastructure development with available urban green space (Geist and Galatowitsch 1999; Skärbäck 2007).

Urban greening has not been undertaken as widely in southern Africa as it has been in other parts of the world. Few attempts have been made to investigate and implement urban greening projects in South Africa (cf. Addo et al. 2000, Donaldson-Selby et al. 2007). In this study, I investigated the expectations of local people regarding conservation-based urban greening projects in an urban setting in South Africa by conducting a survey of students and employees of the Faculty of Health Sciences of the University of Stellenbosch, which is located on the Tygerberg Medical Campus (TMC) in the city of Cape Town. The aim of the survey was to gather information about respondents' opinions and expectations regarding their immediate natural environment surrounding the buildings on the campus. The results of the survey indicate that an attempt to promote biodiversity enhancement through urban greening on campus would be met by support from major stakeholders on the campus.

As a follow-up to the current study, I will use the data gathered to develop a rehabilitation and biodiversity-enhancement project as well as a functional management plan for the TMC. Thus, the project's developers can aim to achieve and maintain higher levels of wildlife biodiversity while taking into account the expectations of the campus populace.

The Study Site

The Tygerberg Medical Campus in the Faculty of Health Sciences of the University of Stellenbosch is situated in the city of Cape Town, which lies in the Cape Floral Kingdom, one of the richest floral kingdoms in the world (Bond and Goldblatt 1984). With just under 1000 administrative and academic staff members and more than 2000 students, the TMC's population represents a broad demographic spectrum. Members of this population have different needs and expectations of their environment, and are active in their interpretations and evaluation of the environment (cf. Churchman 2002). Thus, given the different backgrounds and roles of the various members of the TMC, it is highly likely that they hold a wide range of ideas, opinions, and expectations about global conservation issues, their immediate environment, and the appearance of the campus.

The site was selected following an expression of interest by the management of the faculty to improve the vegetation and overall biodiversity of the campus. The campus of approximately 26 hectares is situated next to the Tygerberg State Hospital, and together they cover a large area in the form of concrete buildings, parking lots, and tarred roads (Figure 2). The Tygerberg State Hospital works closely with the Faculty of Health Sciences, but they are under different management. The main vegetation on the campus consists of lawns and trees, of which a large proportion of species is nonnative. Very little other vegetation occurs on the campus, and where remnants of vegetation do occur, they are controlled through regular mowing. In addition to the above-mentioned facilities, sports grounds cover a significant area of the campus (Figure 3).

Methods and Materials

Although the campus is rich in vegetation when compared to nearby industries and office complexes, it seems that students and personnel perceive the Tygerberg Medical Campus as dull in comparison to the main campus of the University, also situated in Stellenbosch. This study aims to establish (1) whether this perception represents the general attitude of students and personnel, and (2) whether the need exists to see an improvement in this respect. Information will also be gathered on what the students and personnel expect from improvements of the premises. Furthermore, respondents will be given an opportunity to state whether they think that an improved work environment will have a positive effect on their efficiency and attitude.

To establish the current opinions and expectations of personnel and students on the campus, a survey was conducted. Two questionnaires were designed: one for personnel and one for students. The questionnaire was printed in both English and Afrikaans, the two main languages spoken on the campus. The questionnaire designed for personnel (Appendix A) was handed out to 750 administrative and academic staff members on 15 October 2003 with the request that completed questionnaires be returned to a specified office.

The questionnaire designed for students (Appendix B) was handed out to 600 students in all the different disciplines (medicine, physiotherapy, dentistry, etc.) of the faculty, ranging from the second to the final (sixth) academic year, during the registration period on 16 and 19 January 2004. By combining the completion of the questionnaires with the registration process, I reached a significant proportion of the student population and anticipated a large return percentage.

Contents of the Questionnaires

The two questionnaires contained mostly the same questions, but in certain categories questions were tailored to be relevant to the respective groups (e.g., student questionnaires included an extra section regarding their residency).

Analysis

I assigned coded values to all questionnaire answers (see Appendix A and B for coding) and entered these values into a Microsoft Excel spreadsheet. The following sections were specifically coded to get a collective value indicating the respondent's opinion on that section.

Opinion of Nature

I coded answers in this section in such a way that a negative answer was given the lowest value (i.e., 1), while the most positive answer was assigned the highest value (i.e., 3 or higher, depending on the range of possible answers). Then, I added together the coded values allocated to each of the selected complements to the four half statements, resulting in a value ranging from four to thirteen. I then adjusted this value to a final score out of ten. The final score represents an indication of each respondent's opinion value of nature, where 1 represents the lowest possible opinion of nature and 10 the highest possible opinion of nature. In this valuation, the secondary question in question 1 of this section, in which respondents had to respond whether they viewed nature as important or crucial, was not taken into account as too few respondents answered the question.

Campus Appearance

The same method was applied to the first three questions of this section. The answers were coded so that the most negative answer was allocated the lowest value and the most positive answer the highest. The coded values were then added to get an impression value ranging from 3 to 14, which was consequently adjusted to range from 1 to 12, with 1 showing a very negative impression value of the campus and 12 representing the most positive impression value.

I tested all the questions in this section for statistically significant differences between the possible answers by performing a chi-square test in Microsoft Excel. Furthermore, specific questions directed to both students and personnel were also tested for any significant differences between these two groups. In cases where a respondent did not answer the question at all, resulting in a zero value in the chi-square test (i.e., a divided by 0 error occurred in the analysis), the 0 was replaced by 0.5.

Results

A very good response by personnel was achieved, with a return of 196 (28%) completed questionnaires out of 750 questionnaires issued. Of these, 55 (28%) were completed by male respondents and 141 (72%) by female respondents. Distributing questionnaires among students during registration resulted in the exceptionally high return of 568 (97%) out of the 600 questionnaires issued. Of these, 158 (28%) were completed by male respondents and 410 (72%) by female respondents. The following results for students and personnel and comparisons between students and personnel were generated:

Opinion of Nature

The opinion value of students and personnel regarding nature reveals that both groups places very high value on nature (students: $X^2 = 1118.55$, df =9, $p < 0.05$; personnel: $X^2 = 549$, df = 9, $p < 0.05$) (Figure 4). The difference in values placed on nature is statistically significant between students and personnel ($X^2 = 20.75$, df = 9, $p < 0.05$), with personnel placing a higher value on nature than students.

Spare Time Utilization

During a normal weekday, 82% of students prefer to spend spare time off campus ($X^2 = 233.09$, df = 1, $p < 0.05$). When they do spend spare time on campus, 22.7% remain indoors or visit a residence; the majority (38.8%) visit the Tygerberg Student Centre and the second largest number (25.6%) partake in some form of sport. Only 12.7% indicated that they choose to go outside to enjoy nature on the campus ($X^2 = 54.41$, df = 3, $p < 0.05$) (Figure 5a).

In Figure 5b it is clear that during lunch time, the majority of personnel on campus (66%) prefer spending their time indoors ($X^2 = 167.11$, df = 3, $p < 0.05$), with the remaining 34% either leaving campus, staying outdoors (i.e. utilizing the natural environment to some extent), or engaging in other activities. Only 10.4% of personnel indicated that when they have spare time in addition to their lunch break on campus, they regularly take walks, while 48.4% indicated that they seldom take walks on campus and 41% indicated that they never take walks ($X^2 = 46.91$, df = 2, $p < 0.05$).

Figure 5c shows that a significant difference exists between the reasons given by students and those given by personnel for taking walks on campus ($X^2 = 92.81$, df = 4, $p < 0.05$). Apart from walking to and from class, students walk on campus mainly while in conversation with friends or other students, while the smallest group of respondents walk to enjoy nature ($X^2 = 213.27$, df = 4, $p < 0.05$). Personnel, on the other hand, walk on campus mainly in order to enjoy nature and are least likely to walk to undertake private contemplation ($X^2 = 14.70$, df = 4, $p < 0.05$).

The main reasons given by students and personnel for not walking on campus are the lack of features to enjoy while walking and the lack of time to take walks (students: $X^2 = 188.90$, df = 3, $p < 0.05$; personnel: $X^2 = 73.91$, df = 3, $p < 0.05$) (Figure 5d).

Campus Appearance

The impression value of students and personnel is a representation of their thoughts and attitudes regarding the physical appearance of the campus. Data for both students and personnel suggest an average impression value for both groups (students: $X^2 = 399.4$, df = 11, $p < 0.05$; personnel: $X^2 = 205.13$, df = 11, $p < 0.05$) (Figure 6a). There is also a significant difference between the impression values of students and those of personnel ($X^2 = 25.37$, df = 11, $p < 0.05$), with students showing a slightly lower impression value than personnel.

Students and personnel each indicated specific areas or features on campus that they would like to see receive an improved appearance. Gardens and natural vegetation are the two areas that both groups feel need the most improvement (students: $X^2 = 380.24$, df = 3, $p < 0.05$; personnel: $X^2 = 123.65$, df = 3, $p < 0.05$) (Figure 6b). There was no significant difference between student and personnel data for this question.

Campus Improvements

Both students and personnel agreed that if natural vegetation and bird and animal life were improved on campus, they would spend more spare time outdoors on campus than in the past (students: $X^2 = 446.76$, df = 3, $p < 0.05$; personnel: $X^2 = 173.84$, df = 3, $p < 0.05$) (see Figure 7a). Both groups believed that their attitude toward the campus and their work would improve, if natural vegetation and bird and animal life were improved (students: $X^2 = 536.45$, df = 3, $p < 0.05$; personnel: $X^2 = 101.05$, df = 3, $p < 0.05$) (Figure 7b). Furthermore, a larger proportion of students felt this way than personnel ($X^2 = 20.58$, df = 3, $p < 0.05$).

The majority of students (94%) and personnel (97%) support the creation of natural vegetation corridors linking the campus with other natural vegetation areas (students: $X^2 = 432.99$, df = 3, $p < 0.05$; personnel: $X^2 = 173.52$, df = 3, $p < 0.05$). Figure 7c shows the extent of support students and personnel give to suggested improvements on campus. There are statistically significant differences between the options they support (students: $X^2 = 331.19$, df = 5, $p < 0.05$; personnel: $X^2 = 92.39$, df = 5, $p < 0.05$) and significant differences between the options supported by students and those supported by personnel ($X^2 = 55.42$, df = 3, $p < 0.05$).

Additional Commentary

In addition to answering the survey's questions, some respondents wrote supplemental commentary on the questionnaires. A total of 123 students and 65 staff members gave additional commentary. The most frequent suggestions are listed as follows, with the numbers in brackets indicating the number of respondents who made these suggestions:

* Plant more trees (31)
* Add benches and tables (18)
* Plant more indigenous vegetation (14)
* Create a water feature (11)
* Create animal and bird refuges (10)

Questionnaire Return

The high return of completed questionnaires by personnel and students could be an indication of the level of priority with which they regard the issue at hand. According to the Faculty administration, the return rate of 28% by personnel was much higher than their usual return for responses about financial and administrative matters at the University. The very high return by students was in part a result of the timing of the distribution of the questionnaires. As this took place at the beginning of the academic year, students were subjected to fewer time constraints and as they were asked to complete the questionnaires during the registration process, all students were easily targeted. Students were given the option of completing the questionnaires at a later time; nevertheless, most of them, when informed of the nature of the questionnaire, were more than willing to complete it immediately. The significantly higher return by female respondents could most likely be related to the proportion of female and male personnel and students employed by and enrolled at the University.

Questionnaire Results

Opinion of Nature

The general opinion of respondents regarding the environment was very positive. They see it as an important, if not integral, part of human life that has to be protected. It follows that the students would generally be in favor of environmentally positive propositions on campus. It is possible that, given the academic nature of the institution, the importance respondents attach to nature is related to their level of academic development. If this is true, further studies should be conducted to distinguish between the diverse views respondents with different academic backgrounds will express regarding nature. This will also determine the approach to be used when dealing with other sectors of society about conservation issues.

A higher opinion value among personnel, when compared to that of students, may be an indication of the level of responsibility exhibited by each of the groups. In general, it is assumed that students take less responsibility for external concerns than people who have responsibilities to answer to, e.g., careers and families. It may therefore follow that students in general feel less responsible toward issues regarding nature. Kaiser and Shimoda (1999) have shown that moral and conventional responsibility play a role in a person's ecological behavior.

Spare Time Utilization

The high number of students who prefer spending spare time either off campus or indoors, coupled with the small percentage that spend time outdoors, is an indication of the impression that they seem to have of the campus environment. Individuals will be less likely to spend time outdoors if no stimulating experience occurs there. Personnel also spend most of their spare time indoors during a normal working day, with only 10% taking regular walks on the campus. This tendency to stay indoors or to leave the campus whenever respondents have spare time is consistent with a lack of stimulation (in the form of activities, scenery, or recreation) on campus. This is supported by the respondents' impression values regarding the appearance of the campus (Figure 6a).

In cases where students do walk on campus, they do so mainly while in conversation with friends or other students. They are least likely to walk while enjoying the natural aspects of the campus. Personnel, on the other hand, walk mainly to enjoy the natural aspects of the campus. As the respondents are from an academic institution where work requires a lot of their time, it is not surprising that the main reason both students and personnel give for not taking walks on campus is a lack of spare time. Furthermore, there are more spare time activities for students than for personnel. On campus, students can participate in sports, visit friends, stay in their residences, or go to the Tygerberg Student Centre. Personnel, on the other hand, can only stay in their offices, leave campus (time permitting) or take walks on campus. This explains why personnel are more likely to walk on the campus in their spare time. However, this does not suggest that personnel find the natural aspects of the campus to be adequately entertaining while walking.

Campus Appearance

The physical conditions of any work environment play an important role in employee happiness and work satisfaction. Physical conditions can include aspects such as office ergonomics and physical and natural appearances. Respondents at the TMC rate the impression value of the campus as slightly below average, with students generally rating it lower than personnel. A possible reason for this phenomenon could be that 59% of the student respondents reside on the campus, coupled with the assumption that individuals seem to place a higher premium on and therefore show higher expectations of their living environment than their working environment. Both students and personnel feel that gardens and natural vegetation are the two areas that need most improvement. Should these areas be improved, it is expected that the impression value of both students and personnel will increase significantly.

Campus Improvements

Respondents agree that they would spend more of their spare time on campus if the natural vegetation and bird and animal life of the campus were improved. They also feel that should this happen, their attitude toward the campus and their work would improve. This opinion is stronger among students than among personnel, which could again be explained by the fact that students reside on campus. The general opinion of respondents (i.e., that their attitude toward the campus and their work would improve and that they would spend more time on the campus should the campus environment be improved) supports findings that one's mood is affected by the qualities of one's surroundings and that entering different environments can alter one's mental state or mood (Apter 1982, 1989; Russell and Snodgrass 1987). This was shown by experiments using photographic environmental simulations, in which natural settings have been found to alter emotions positively, while urban settings seem to create negative emotions (Hartig et al. 1991, 1996, 1999). These findings could be used as leverage in attempts to convince other institutions to improve and manage their natural surroundings. The argument would be that in return for their investment, the institutions are likely to witness higher levels of work satisfaction and higher efficiency in staff.

An overwhelming proportion of respondents (94% of students and 97% of personnel) support the proposal to establish natural vegetation corridors between the TMC and other natural vegetation areas. Their support for the proposed enhancements on campus (e.g. the establishment of footpaths, benches, rest areas, water features, and the reintroduction of fynbos) can be utilized in the development of a management plan for the enhancement of biodiversity on the campus.

Discussion

It is clear from this study that the individuals and groups working and studying at the TMC have specific expectations regarding their living and working environment. They also seem to be very particular about what they would like to see improved there. The general sentiment among respondents is that improvements in the gardens, followed closely by improvement in the general natural surroundings, would improve their attitude toward the campus and their work/study environment. This is supported by previous studies on urban greening and urban nature conservation (cf. Goode 1989; Geist and Galatowitsch 1999). One can therefore assume that urban greening is very likely to improve the respondents' work efficiency and willingness to perform tasks to the best of their ability. One would expect that these findings at the TMC might also be indicative of respondent reactions in other sectors of society. To verify this, similar studies should be performed in the industrial and business sectors. Should these sectors show similar responses, conservation biologists would gain an argument to use in support of well-designed biodiversity enhancement projects. By including the participation of local people in the design and execution of such projects, they will also increase the success achieved, as indicated by Goode (1989).

Arrow et al. (1993) suggest that when the value a person places on nature is determined, one should ask what monetary value he or she is willing to attach to access to a natural environment, and not merely what he or she is willing to accept without committing his or her own financial or material resources. This would reflect a more accurate expression of a person's willingness to use his or her own resources to conserve and protect nature. Within this framework, the results of the survey in this study cannot necessarily be regarded as a true reflection of the willingness of respondents to contribute in full to environmental conservation. Respondents were seemingly eager to suggest improvements to their immediate environment, but whether they would actively participate in a project to bring about the suggested improvements remains to be seen.

It is clear from the outcome of this survey that people seek increased interaction with nature in their everyday environment. Conservationists should utilize this need in order to convince authorities to spend more time and effort on ensuring the natural well-being of the urban environment, even within the most densely developed cities. However, further investigations need to be carried out in order to truly understand the influence that urban nature has on society. The effect of nature on the productivity and general well-being of employees in the work environment needs to be tested and the influence of restored environments on humans determined. Furthermore, this study also provides evidence of the need to involve local people in urban greening projects.

It has now become the task of conservationists to convince society to make investments in the protection of earth's remaining natural habitats and to promote their restoration and management. One way to achieve this would be to promote the human benefits that result from interacting with a richly biodiverse area in one's immediate surroundings, which has been shown to strongly affect emotions, attitude, and mental abilities (Fredrickson and Anderson 1999; Hartig et al. 1999; Herzog and Barnes 1999; Kerr and Tacon 1999). By capitalizing on these benefits, society can be motivated to put more effort into habitat restoration and management.

Acknowledgments

I am grateful to Dave Pepler, the Department of Conservation Ecology, and the Faculty of Health Sciences of the University of Stellenbosch for their support and provision of resources to conduct the research for this study. I would also like to thank the following institutions: the Namibian Government, the Africa-America Institute, and the University of Namibia for their financial support for the duration of the study.

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Figure 1: The effect of an individual's surroundings on his or her attitude and the consequences for work efficiency. The diagram shows that negative stimuli result in a negative attitude or lack of spirituality, and positive stimuli result in a positive attitude or heightened spirituality.

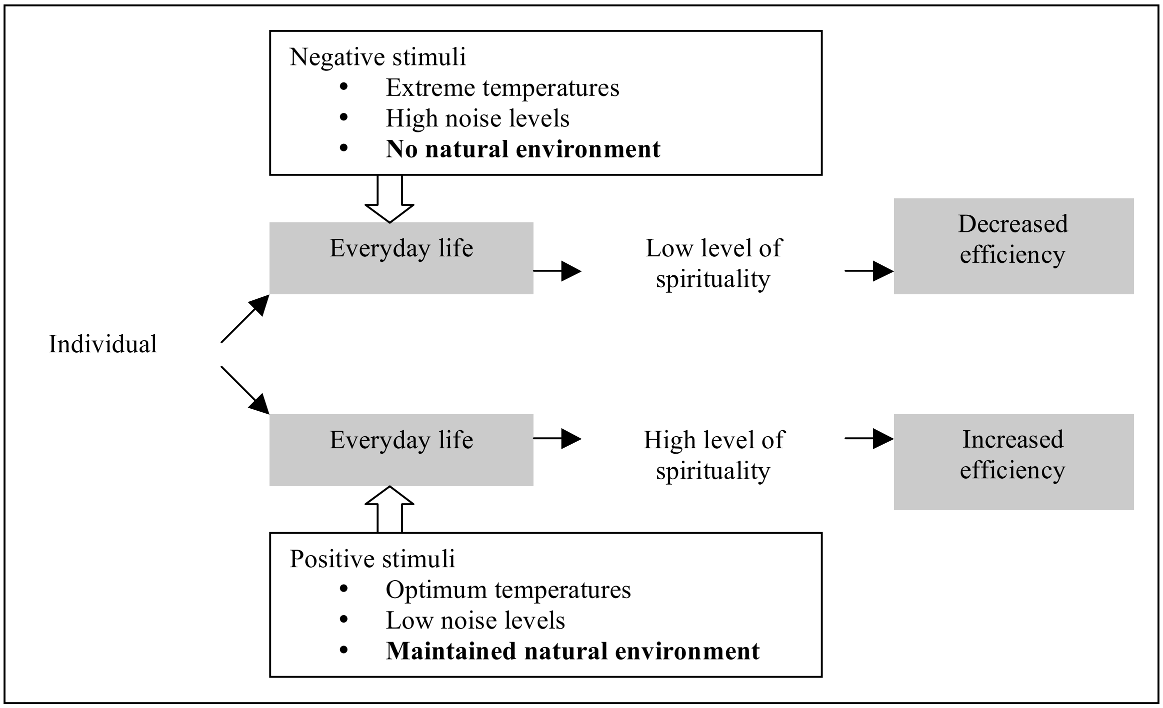


Figure 2: The Tygerberg Medical Campus (TMC) garden areas with the buildings in the background.

Figure 3: The open areas and sports grounds on the western side of the Tygerberg Medical Campus.

Image 1: This image depicts the difference between the mowed areas on campus and the neighboring railway grounds.

Image 2: This image depicts the sports grounds and other open areas on campus that are kept neat by mowing.

Figure 4: Graph representing, as a percentage, the opinion value of respondents regarding nature. Both students and personnel show a significant positive that averages 9 (students: X^2 = 1118.55, df = 9, p < 0.05), personnel: X^2 = 549, df = 9, p < 0.05). Personnel places a significantly higher value on nature than students (X^2 = 20.75, df = 9, p < 0.05).

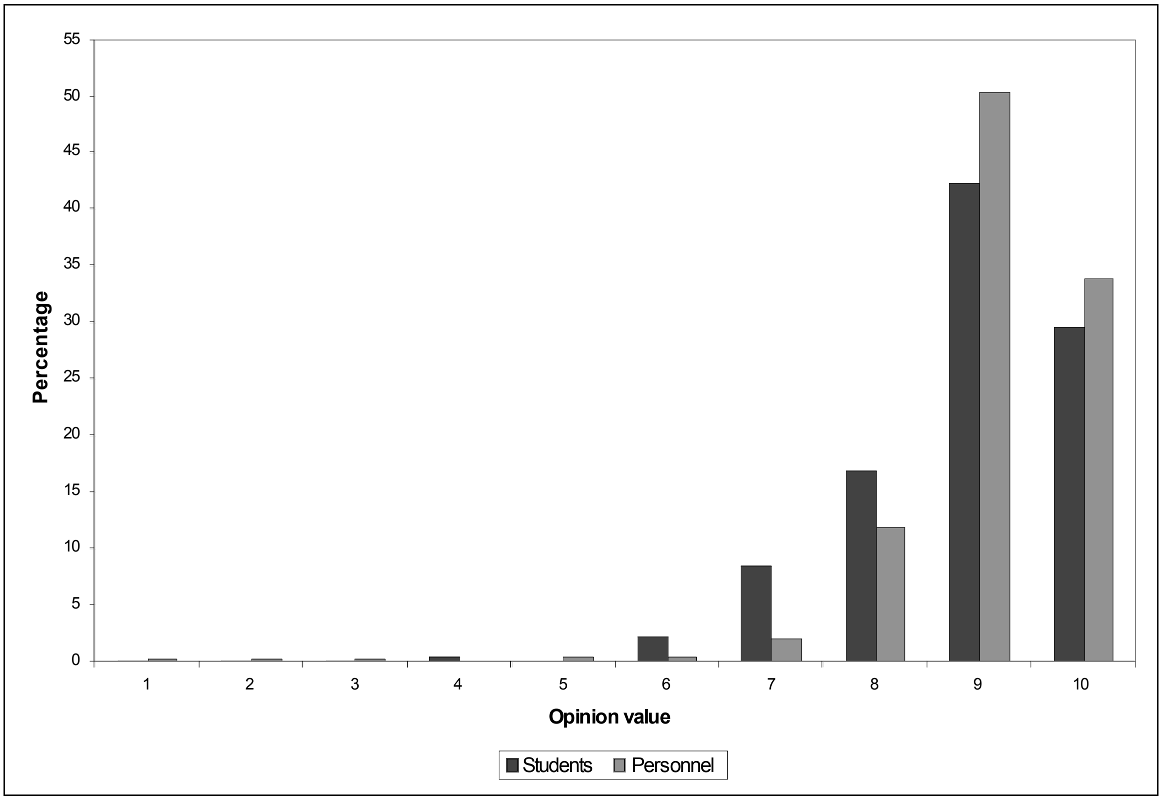


Figure 5a: Figure representing, as a percentage, student choice of place for spending any available spare time while on campus. The most significant proportion prefer spending time in the Student Centre (X^2 = 54.41, df= 3, $p < 0.05$).

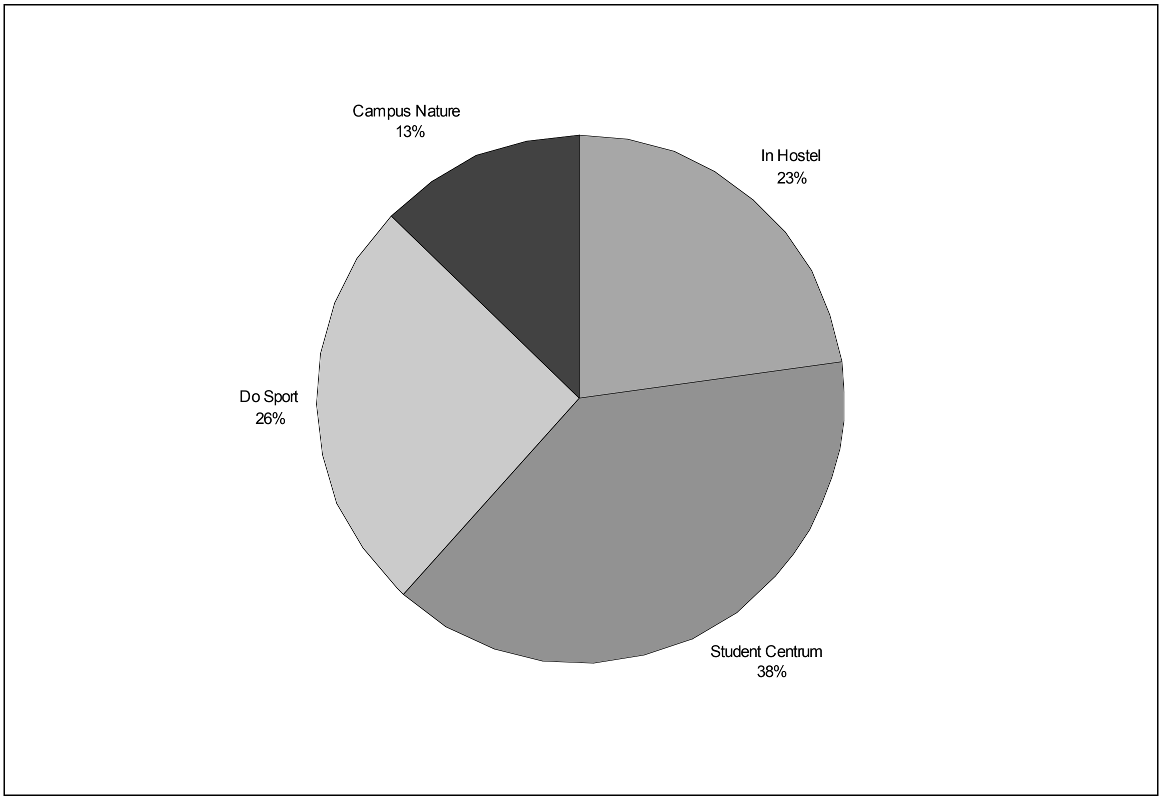


Figure 5b: Figure representing, as a percentage, personnel choice of place for spending lunch time. The majority of personnel (66%) prefer spending their time indoors on campus ($X^2 = 167.11$, df = 3, $p < 0.05$), while the remaining 34% either go off campus, stay outdoors, or do something else.

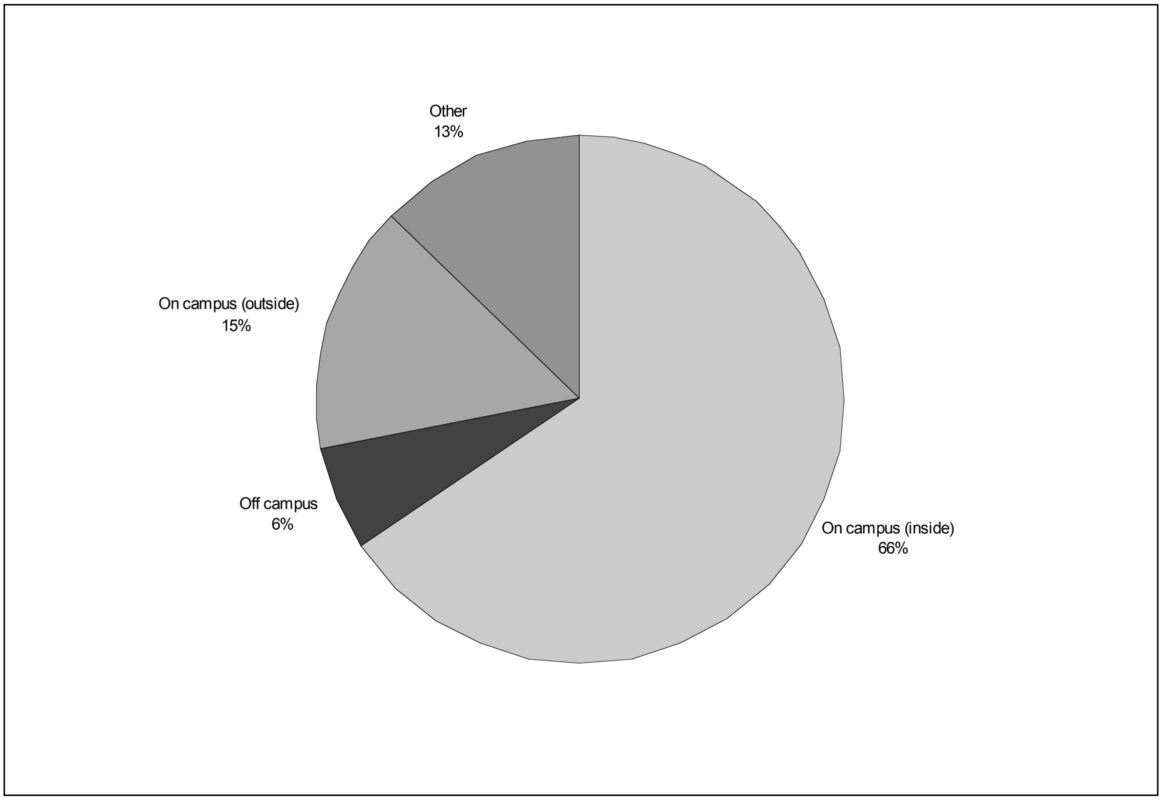


Figure 5c: Graph representing as a percentage the reasons why respondents take walks on campus. A significant difference exists between the reasons of students and those of personnel for walking on campus (X^2 = 92.81, df = 4, $p < 0.05$). Apart from walking to class and back, students walk on campus mainly while having discussions with friends or other students, while the smallest group walk to enjoy nature (X^2 = 213.27, df = 4, $p < 0.05$). Personnel, on the other hand, mainly walk on campus in order to enjoy nature and are the least likely to walk while taking time for private contemplation (X^2 = 14.70, df = 4, $p < 0.05$).

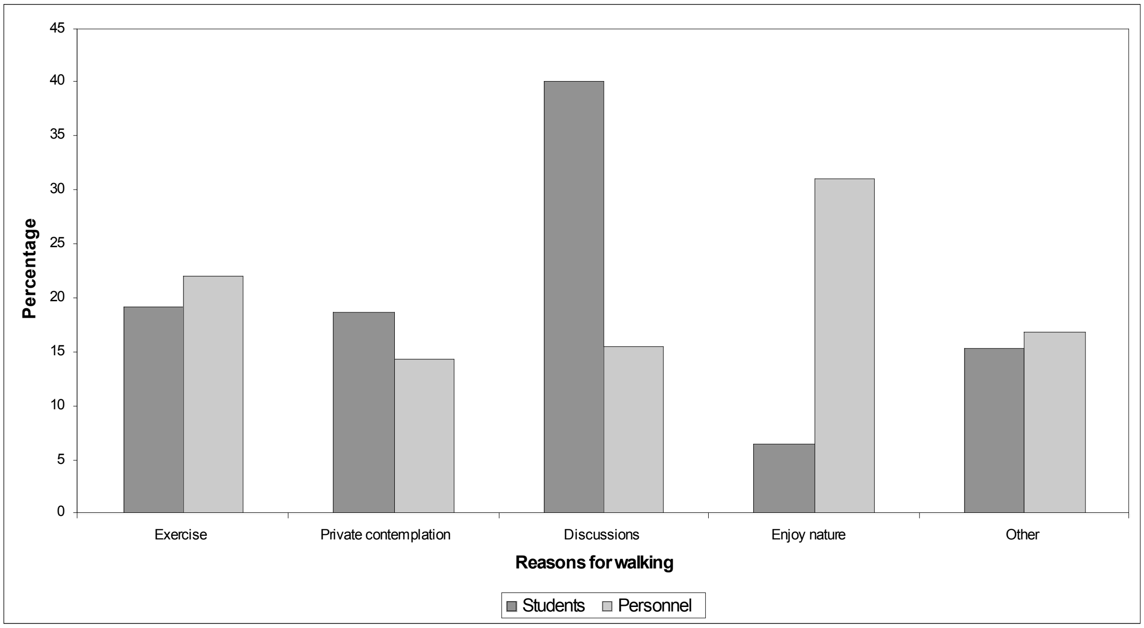


Figure 5d: Graph representing as a percentage the reasons why respondents are reluctant to take walks on campus. The main reasons why students and personnel don't walk on campus are the lack of things to enjoy while walking and the lack of time to take walks (students: $X^2 = 188.90$, df = 3, $p < 0.05$, personnel: $X^2 = 73.91$, df = 3, $p < 0.05$).

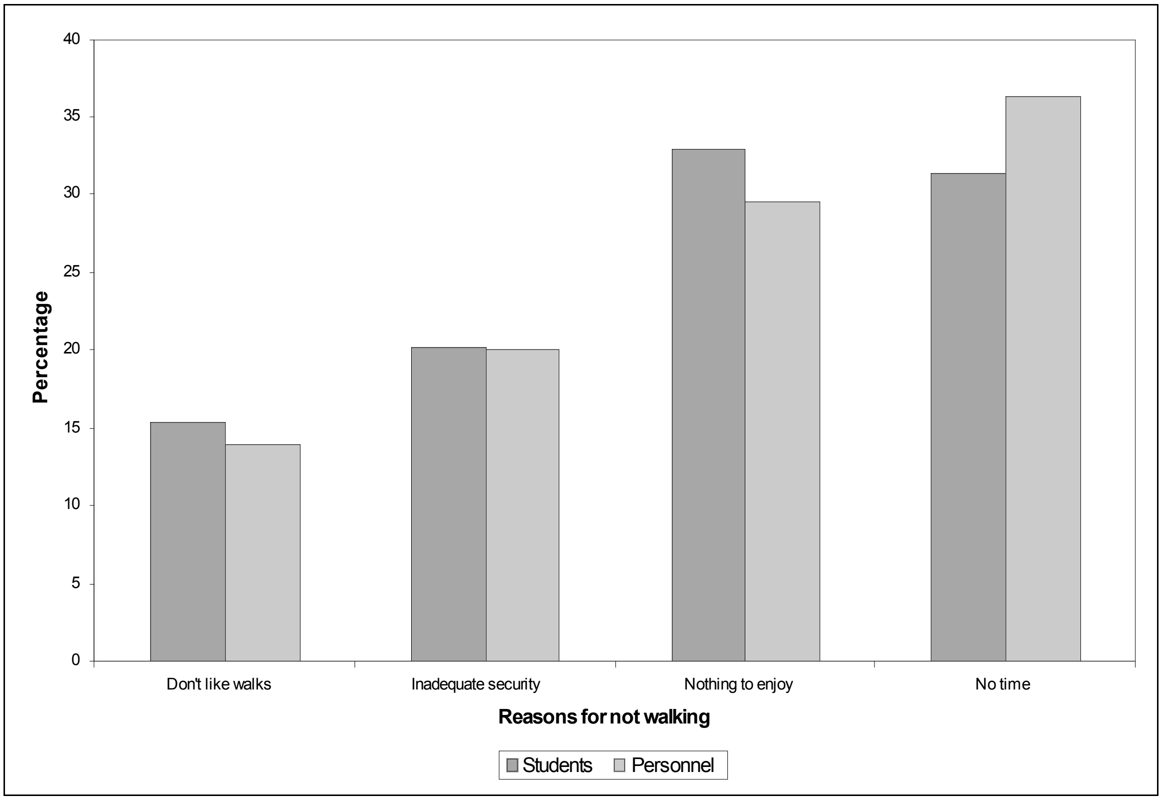


Figure 6a: The impression value of students and personnel regarding the Tygerberg Medical Campus appearance. Data for both students and personnel suggest a statistically significant tendency for respondents to have an average impression value (students: $X^2 = 399.4$, df = 11, $p < 0.05$; personnel $X^2 = 205.13$, df = 11, $p < 0.05$). There is also a significant difference between the impression values of students and personnel ($X^2 = 25.37$, df = 11, $p < 0.05$), with students having a slightly lower impression value than personnel.

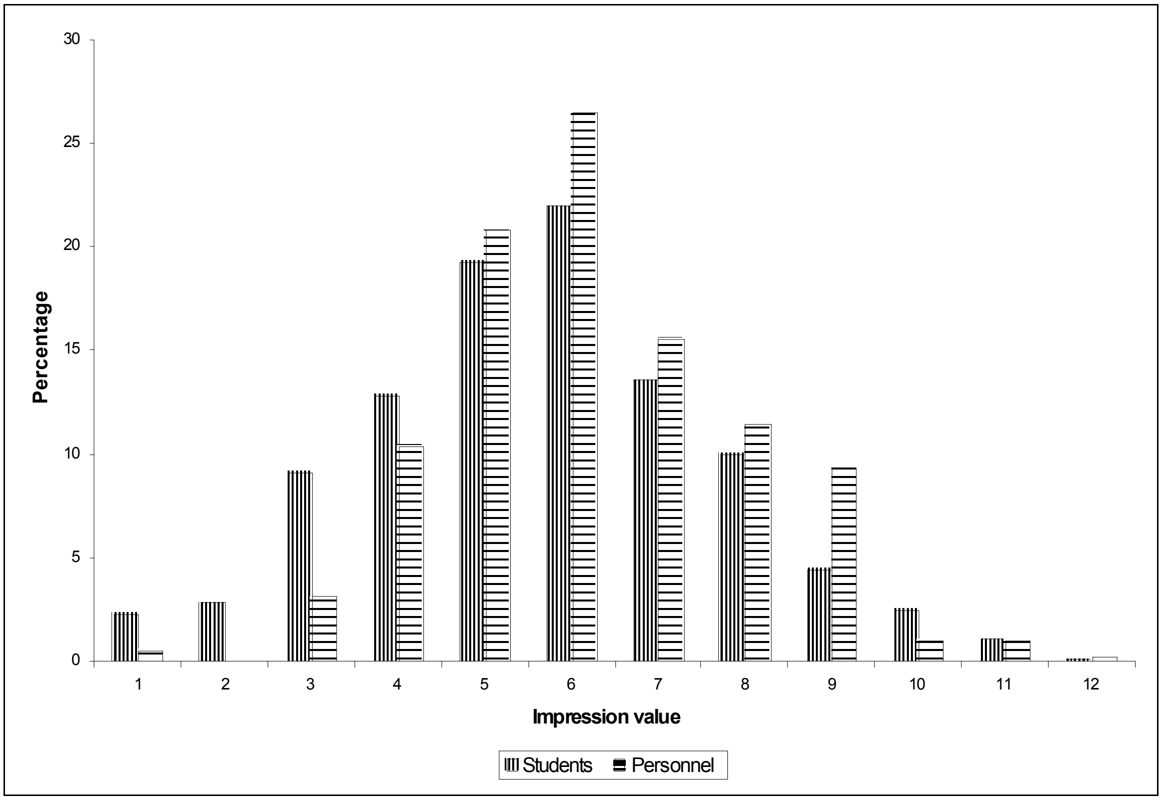


Figure 6b: The areas or items on campus that respondents would like to see improved in appearance. Gardens and natural vegetation are the two areas that they feel need the most improvement (students: $X^2 = 380.24$, df = 3, $p < 0.05$; personnel: $X^2 = 123.65$, df = 3, $p < 0.05$). There was no significant difference between student and personnel data.

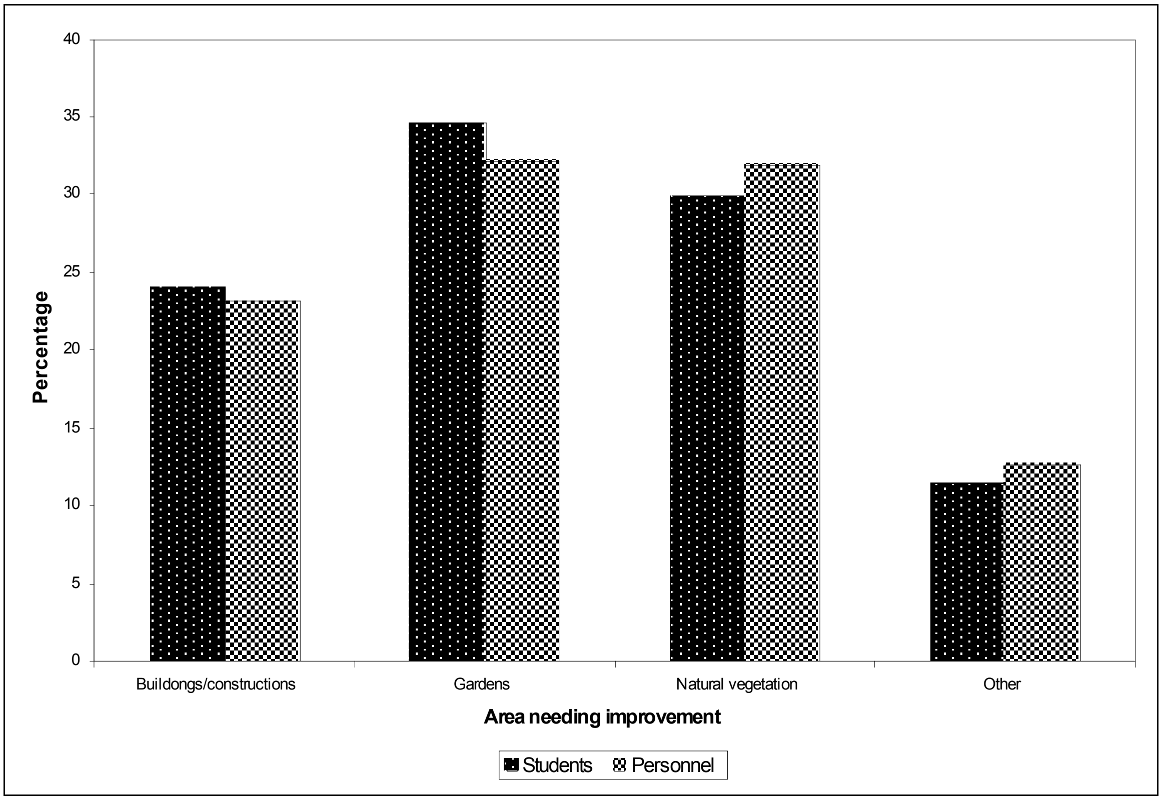


Figure 7a: Respondents' level of agreement with the statement that if natural vegetation and bird and animal life were improved on campus, they would spend more of their spare time on campus than in the past. A significant proportion agreed to this (students: $X^2 = 446.76$, df = 3, $p < 0.05$; personnel: $X^2 = 173.84$, df = 3, $p < 0.05$).

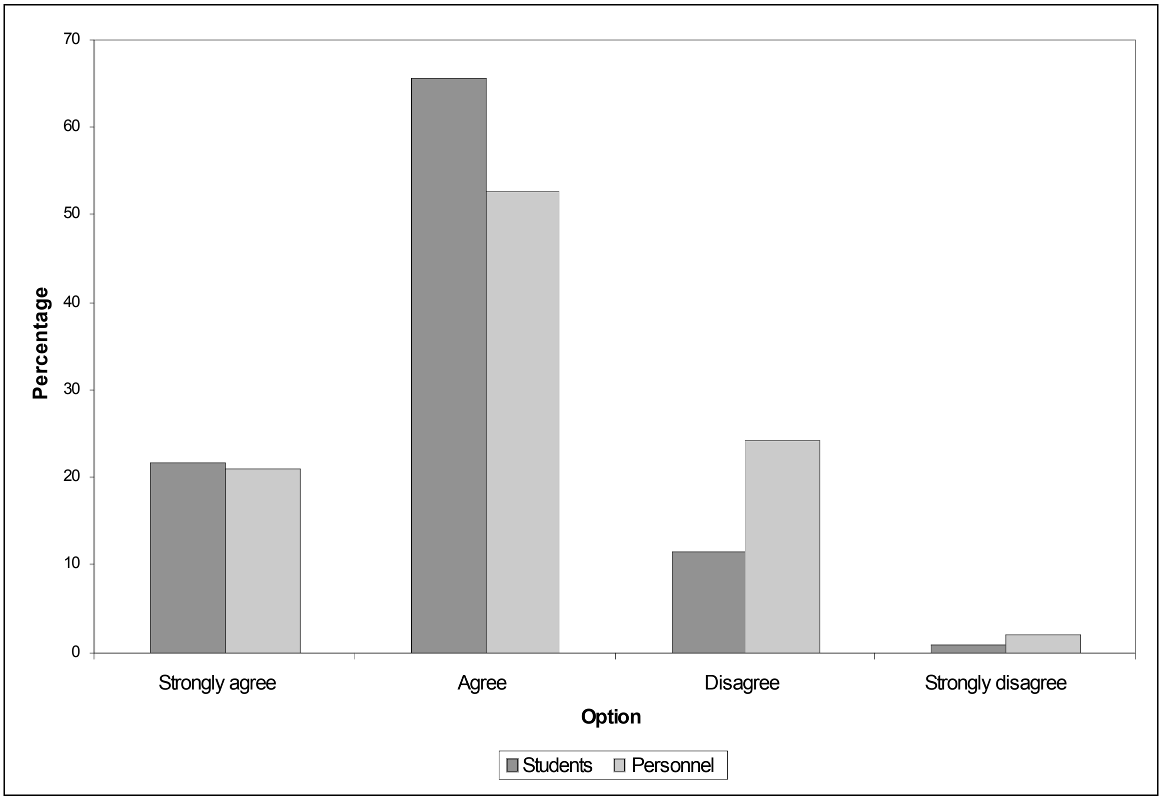


Figure 7b: Student and personnel attitudes toward the campus. A larger proportion of students felt that their attitude toward the campus and their work would improve if the natural vegetation and bird and animal life were improved ($X^2 = 20.58$, df = 3, $p < 0.05$).

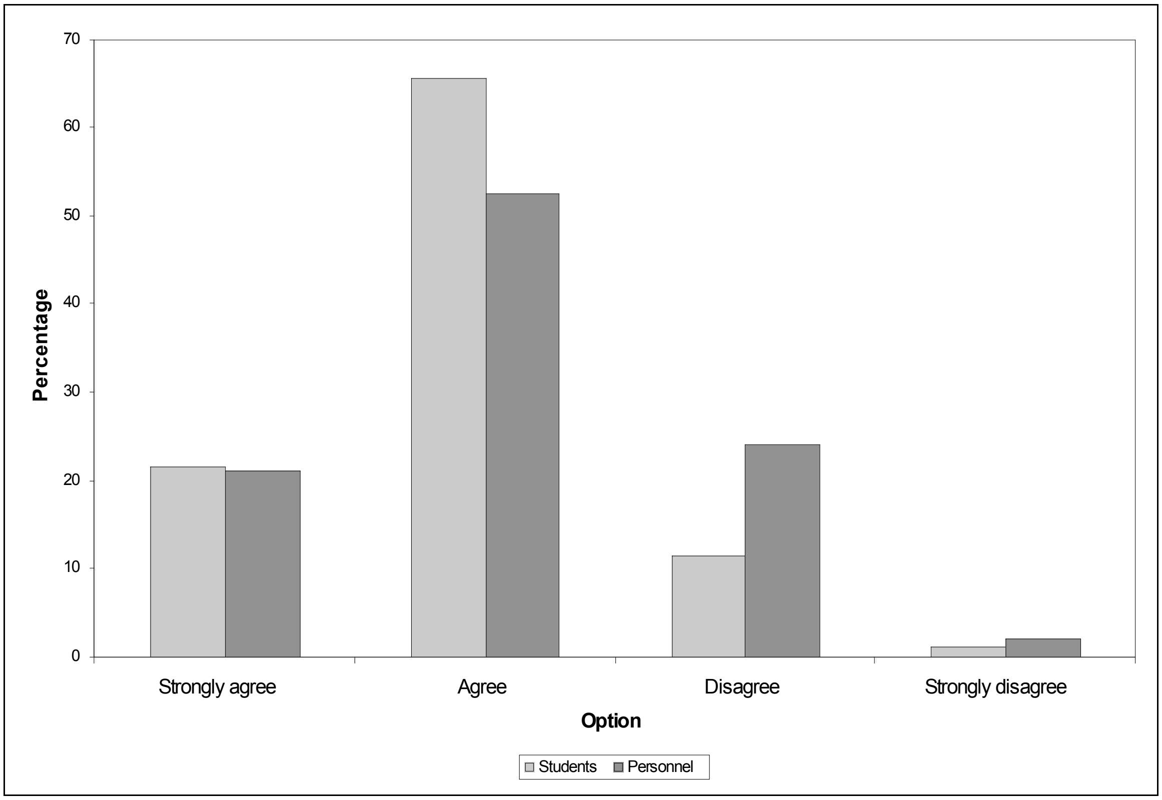
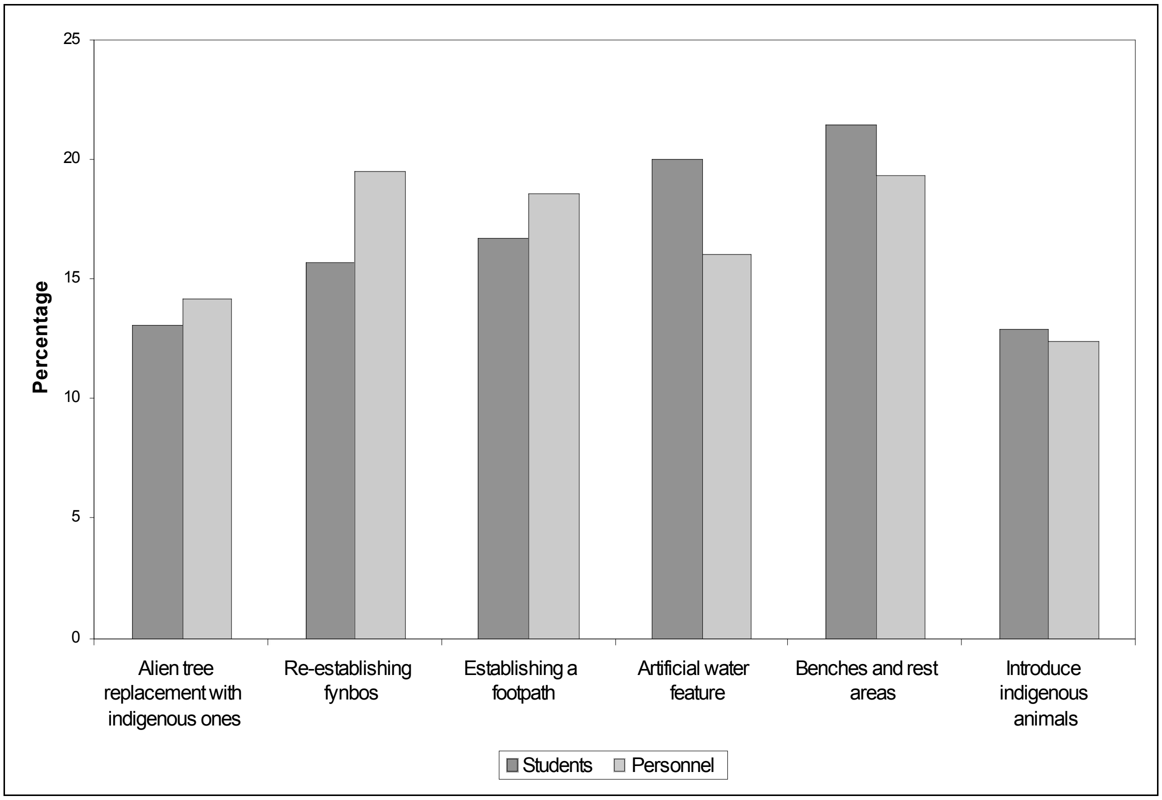


Figure 7c: The percentage of support students and personnel give to suggested improvements on campus. There are statistically significant differences between the options they support (Students: $X^2 = 331.19$, df = 5, $p < 0.05$; Personnel: $X^2 = 92.39$, df = 5, $p < 0.05$) and significant differences between the options supported by students and the options supported by personnel ($X^2 = 55.42$, df = 3, $p < 0.05$).



Butterfly Activity in a Residential Garden

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Abstract

Butterflies are a highly visible, well-loved, and well-studied part of Britain's native fauna, yet there is still very little known about how butterflies use one of the country's most commonly available habitats, the residential garden. Studies in a Wolverhampton (UK) garden demonstrate that the majority of individuals use these spaces as movement routes through the urban matrix. Of 516 observed individual visits by butterflies over three recording seasons (2000–2002), only 13.8% involved a stop for some purpose. The duration of these visits was characteristically short, with a mean visit time of nine seconds. Individuals tended to fly through the study garden using distinct entry and exit points largely dictated by variations in structure within the study garden and in the immediately surrounding gardens. Individual garden use by butterflies would therefore seem to be defined as much by structural imperatives as by availability of nectar- or food-plant species. When considered as systems of interconnected green spaces on the level of the housing block (defined as a continuous area of residential land use bounded by infrastructure or contrasting land uses) and of the urban area as a whole, residential gardens represent an extraordinarily valuable and dynamic component of the urban habitat.

Key words: residential garden, housing block, butterfly, flight paths, vegetation structure, corridor, urban green space

Background: Butterflies and Gardens

Butterflies are one of the best-known and most charismatic groups of fauna (Asher et al. 2001; Dover and Sparks 2000; Vickery 1995). Encounters with these easily identifiable creatures, along with garden birds, are among the first real contacts many people have with wildlife, and the careers of many keen professional and amateur naturalists have been inspired by such sightings. There is, as a result, a thriving network of ecologists studying these creatures, including many who contribute to a wide range of research studies into the spatial distributions and population dynamics of both rare and common butterfly species. This research encompasses a variety of survey activities, from national schemes—for example, the Millennium Atlas of Butterflies in Britain and Ireland (Asher et al. 2001) and long-term monitoring programs (UK Butterfly Monitoring Scheme 2006)—to studies of characteristic species assemblages of particular habitat types (Croxton et al. 2005; Van Swaay 2002; Dover and Sparks 2000) and metapopulation studies of spatially restricted or rare species (Heikkinen et al. in press).

Urban areas have come under closer investigation in recent times due to the recognition that conservation and management of urban habitats and species pose particular challenges (Angold et al. 2006; Young and Jarvis 2003; McDonnell et al. 1997). Because of their intrinsic appeal, generally known life-history requirements, and sensitivity to small changes in site conditions, butterflies have often been a significant part of these studies (e.g., Wood and Pullin 2002). Indeed, some larger-scale studies have focused wholly or partially on butterflies as indicators of urbanization and/or the effects of urbanization (Bock et al. 2007; Vanreusel and Van Dyck 2007; Blair and Launer 1997; Ruszczyk and De Araujo 1992). Despite this large research community, there appears to be relatively little research—with the exception of the recent BUGS (Biodiversity in Urban Gardens in Sheffield) Urban Regeneration (URGENT) project (Thompson et al. 2003 et seq.)—into the most common, everyday, urban garden habitat, even though it is recognized in many local conservation strategies (for example, Wildlife Trust 2000).

Gardens are a significant proportion of the fabric of urban areas in the UK, comprising 19–27% of land use (Smith et al. 2005), and constitute a significant area of extensive interconnected green space (Mathieu et al. 2007). Yet we still know surprisingly little about their landscape or ecological roles (Smith et al. 2006; Chamberlain et al. 2004; Owen 1991). Although schemes such as the UK annual Butterfly Conservation Garden Butterfly Survey use data from amateur recorders to monitor gross nationwide trends in species, no survey has explicitly investigated "the mobility or duration of stay of butterflies in gardens. There appears to be a dearth of published work on this aspect." (Vickery 1995).

When undertaken, movement studies on butterflies have tended to look at the regional movements of certain species (Binzenhofer et al. 2005; Schneider and Fry 2005; Pryke and Samways 2001) as well as landscape-scale interpopulation movements (Sutcliffe et al. 2003) and the resultant effect on genetic diversity (Wood and Pullin 2002). Although there have been some smaller-scale movement studies that have investigated the influence of minor landscape features, either experimentally (Haddad 2000) or in the wider rural landscape (Cant et al. 2005), very few have investigated the microscale—the features that determine whether an individual butterfly will move through the structural complexity at the scale encountered within an individual patch (Dover and Fry 2001; Sutcliffe and Thomas 1996; Loertscher et al. 1995; Dennis 1986). No studies have looked at this in relation to the garden habitat.

This study aims to rectify this omission by quantifying indicative residence times and overall garden use by butterflies in a residential British garden. The data set was collected over three recording seasons in a single garden, rather than via a spatially complete study, and as such it provides baseline data on garden use rather than on the distributions of species in gardens.

Study Site

The study site is a residential garden in suburban northwest Wolverhampton in the West Midlands area of the UK (Figure 1). The 20-meter-long and 10-meter-wide garden is bounded on the north, south, and east sides by a 1.5-meter-high wooden fence and on the west side by a semi-detached house. The garden is adjacent to other gardens to the north, south, and east. Immediately bounding the garden is a variety of small garden buildings, trees, and shrubs (Figure 2a and Figure 2b). This includes sheds and garages that provide breaks of between 1 and 3 meters in the otherwise largely solid, shrubby boundary edge where it rises above the fence. There are also some larger and denser plantings with small gaps between adjacent canopies that act as barriers to the movement of wildlife. Within the garden, there is a similar mix of open areas and denser planting/solid structures (especially at the eastern end). The interplay of taller trees (> 5 meters), medium height shrubs (up to 5 meters), and the gaps between them resulting from garden management and garden buildings provides an intricate, though limited, network of potential routes for butterflies to use when moving in and out of the garden.

The study garden is one of approximately 100 such plots that form a continuous block of garden green space—which is itself one block among many thousands in the Wolverhampton urban area. In the context of this study, a "block" is taken to describe a continuous area of residential land use bounded by infrastructure or contrasting land uses and is used as a purely descriptive term. The garden itself is broadly wildlife-friendly, with a low input of artificial fertilizers, pesticides, and herbicides; apart from this, the garden is rather typical of urban UK gardens, being approximately rectangular in shape and consisting of a mixture of lawn, border flowers, herbs, shrubs, and trees.

Recording Butterfly Activity

To record butterfly activity, I used a hybrid of several established butterfly recording approaches. Recording time frames and environmental limitations were adopted from Pollard and Yates's (1993) standard Butterfly Monitoring Scheme (BMS) transect method. The practicalities of recording the movements of an individual visitor were adapted from the botanic garden method used by Wood and Samways (1991), and the method for recording and transcribing observations of butterfly behavior was adapted from Dover (1989).

To observe visitors, I undertook steady-paced, repetitive walks around the study garden. Flying butterflies were spotted prior to entering the garden and then were tracked over the course of their activity. The total range of each individual's activities was noted, and the timing of each activity, its flight track, and any stopping places were recorded as accurately as possible on a field recording sheet plan of the garden (Figure 2a). I did not actively hunt out butterfly visitors in the study garden's vegetation except when I tracked them there from flight. Each butterfly visit was recorded using a hand-held stopwatch, and times were rounded to the nearest second. Activities recorded were: flying, feeding, resting/perching, basking, and "other" (e.g., territorial displays).

The first individual observed was followed until it left the garden confines and airspace, even if it was still in sight (for example, moving into the neighboring garden), and any other individuals entering the garden during this time period were not recorded. If a butterfly flight was close to the garden edge and there was uncertainty as to whether it was just inside or just outside of the study garden, it was recorded as a visit. If a recorded individual left the garden, was kept easily in sight, and returned without alighting elsewhere—for example, took a simple flight path detour—its return was counted as part of the same visit to the garden (as in Dover 1989). Otherwise, return trips were counted as different visits because residence times, utilization of the garden, and activity during each visit were under examination, and not complete individual life histories. Butterflies were not recorded if their appearance was registered before and at the beginning of the recording period. However, if their ongoing activity coincided with the end of the scheduled recording period, butterfly visitors were tracked and recorded until their visit was completed.

The recording season extended from April 1 to September 29 (26 weeks), with one recording day attempted per week. Recording times were standardized to four hours each recording day with two hours during the morning (between 10:00 and 13:00) and two hours during the afternoon (13:00–16:00). Due to the sheltering effects of adjacent housing and the unique microclimates created by garden vegetation and artificial structures, and in an attempt to maximize the number of recording weeks, recording was undertaken in more marginal climatic conditions, i.e., when temperatures were slightly cooler than indicated in the BMS method, in the study garden than may have been attempted in "natural" habitats. As the recording period followed recommended BMS seasons, visits outside these recording months were not taken into consideration—although it is likely that urban gardens may have particular importance for butterfly movement and activity both early and late in the year, when nectar sources and fruit may be more available.

Laboratory and Statistical Analysis

The flight path of each butterfly was transcribed as closely as possible onto the recording sheet in the field, then transferred in the laboratory to the ArcView Geographical Information System to allow the investigation of trends and spatial use of the garden. The associated database was then exported to Microsoft Excel and the data analyzed for relationships using analysis of variance (ANOVA).

General Results

Recording was undertaken in a total of 28 weeks out of a possible 78 over the 3 recording seasons of 2000–2002, 5 weeks of which produced no records. Twenty-five recording weeks were lost to bad weather and a further 25 to external commitments. Over the whole period, I recorded 13 species, out of a possible total of 59 native UK species. I also recorded a separate category of undetermined Small White species that flew through the garden quickly or at the farthest recordable distance and so were difficult to identify with certainty. Where these were recorded, I could not confidently identify these as Small Whites (*Pieris rapae*), Green-Veined Whites (*Pieris napi*), or female Orange Tips (*Anthocharis cardamines*), the "white" species recorded at other times in the study. In total, I recorded 516 individual butterfly visits in 112 hours of observation, with 278 visits in 2000, 128 visits in 2001, and 110 visits in 2002.

Two species dominated throughout: Large Whites (*Pieris brassicae*) and Small Whites (*P. rapae*) logged 169 and 138 visits, respectively. The least common visitors were Meadow Browns (*Maniola jurtina*) and Painted Ladies (*Vanessa cardui*), which registered only single visits. Commas (*Polygonia c-album*), Orange Tips (*Anthocharis cardamines*), Holly Blues (*Celastrina argiolus*), and European Peacocks (*Inachis io*) all also logged visit numbers in single figures, despite their traditional association with gardens.

The three recording years show a variety of trends in both variability of recording time and butterfly numbers (Figure 3). The year 2000 had 15 recording weeks with 278 individual garden visits and a mean of 18.5 visits per recording session. Large and Small Whites dominated with 187 visits (> 67%), while

Meadow Browns, as noted above, showed only one record for all three seasons. The 2001 observation season saw 128 visits in 8 recording weeks (16.1 visits per session), with Large and Small Whites again dominant at 73 visits (> 57%). The 2002 season had the fewest recording weeks at 5, but claimed the highest mean per session (22), with Large Whites comprising the majority (31.8%) of all observed butterfly species.

Garden Activity: Flight Times

Flight times of individual visits ranged between 2 and 128 seconds, with a mean visit time of 9 seconds. There was considerable variation among species, with Orange Tips having the shortest visit times, averaging 2.5 seconds, and Gatekeepers (*Pyronia tithonus*) having the longest mean visit times, at 16.7 seconds (Figure 4). Six species had mean flight times of between 5 and 10 seconds, and three further species had mean flight times between 10 and 11 seconds. The undetermined Small White species group had a mean flight time of 5.5 seconds, with a minimum of 3 seconds and maximum of 13 seconds.

Despite the variety of timings recorded, the large variability within each species resulted in no significant differences in overall flight times between species (P = 0.3154) or between years (P = 0.2143). For species with more than 40 visits, there was some variability in flight times between years; both Small White and Speckled Wood (*Pararge aegeria*) showed no differences, while Green-Veined White (P = 0.00005) and Large White (P = 0.009) demonstrated significant year-to-year variability.

Garden Activity: Feeding, Basking, and Perching

Of the 516 individual visits, 71 involved stops for some purpose (13.8%). Fifty-two visits involved single stops, 14 showed 2 stops, and 5 visits had between 3 and 14 stops. Individual stops varied between 1 and 951 seconds, with feeding stops averaging 78.4 seconds and basking stops 99.2 seconds.

All individual visits to the garden were for a single purpose, mainly feeding or basking. The single exceptional visit involved five stops: three for nectaring, one for basking, and one for perching. Nine species stopped in the garden for some purpose, but only five species had two stops or more (Speckled Wood, Large White, Small White, Green-Veined White, and Red Admiral). The 27 recorded basking visits involved activity on such diverse substrates as a child's paddling pool, windowsill, and vegetation, while perching behavior was noted only on vegetation (no artificial perches) and during only eight individual visits.

Garden Activity: Flight Paths and Routes

Individual butterfly visitors exploited the garden landscape in a variety of ways, moving at a range of heights and responding both to the structural complexity of the garden surroundings and the internal heterogeneity of the garden (see Young 2005 for a fuller discussion of individual species responses). Despite this variety, there were significant uses of particular routes, with butterflies using common entry and exit points in response to a range of primarily structural modifiers of their behavior.

Gaps in the surrounding vegetation and hard structures both inside and outside the garden channeled flight paths. For example, individual visitors usually avoided any trees and shrubs that extended above the fence line, but used as entry or exit points the gaps created where neighboring sheds and garages extended above the fence line and as a result formed a break in the tree barrier (Figure 5). The trees in the garden to the north had a particularly noticeable influence on butterfly movement. They were substantially taller than those in the study garden and therefore created a channeling effect, as butterflies had to fly around a tree to exploit gaps between it and adjacent tree canopies and therefore deviated from otherwise straight flight paths both to and from the neighboring garden.

Missed Individuals

As this study was not designed to establish population sizes but rather to give an insight into activities during visits, it is certain that the results underestimate the number of visits made during the recording sessions and therefore overall. Ad hoc observations in 2000 and 2001 noted several individuals that were not recorded as they flew through while other visitors were being recorded. During 2002, an attempt was made to actively record numbers of these known missed individuals.

During week 7, when 39 visits were recorded, 9 further individuals were noted (> 23% of total visits); during week 19, 39 were again recorded with 12 noted as missed (> 30%); and finally, in week 24, 30 were recorded and 6 missed (20%). This indicates that the results underestimate the number of visits by 20 to 30%, especially considering that there were undoubtedly individuals missed but not noted.

As there were fewer individuals in total on the wing in spring and early autumn, it is likely that a greater proportion of the total number of butterflies using the garden was recorded in those seasons. In the summer months, several individuals were observed in the garden at the same time, with the inevitable result that a lower proportion of the total number was probably studied.

Discussion

As with any recording activity, the results of this study were heavily dependent upon when recording was possible. For example, the exceptionally poor early-season and late-season weather of 2001 resulted in low numbers of butterflies recorded in spring and autumn and restricted recording opportunities. None of the recording years had exceptional numbers of migrating vanessids except for late-season sightings of Red Admirals in 2000 and 2002 on warm, windless autumn days. Therefore, apart from the relatively low numbers of traditional garden species of butterflies, the species presences shown here are not especially noteworthy.

Even allowing for both the underestimation of individual visits due to recording bias and the lack of studies with which to compare data, there were surprisingly small numbers of individual visits. Annual visit totals were usually boosted by a couple of busy recording weeks in midsummer. For example, in 2000, 87 individuals visited during one recording session (week 21), while one session had two visits (week 1), one had one visit (week 3), and three sessions had no visits (weeks 5, 7, and 22). There is also likely to have been a recording bias toward feeding and flying individuals because these are the most prominent (Dennis et al. 2006; Loertscher et al. 1995); therefore some resting, perching, and basking individuals could well have been overlooked.

The length of time that many butterflies spent in the garden was also unexpectedly low, with significant numbers, 47% of all visits, flying through in less than five seconds. It is evident that butterflies are using individual gardens as part of a wider meta-habitat, identifying available resources rapidly and then moving on if their requirements are not met at that particular time. In terms of determining nectar sources, for example, butterflies may be scrutinizing flower heads very rapidly (< 1 second) (Dennis 1986; Goulson 2000), and this may have a commensurate impact on residence times.

The role of structural determinants of activity identified here reinforces the importance of 3D features in the landscape, especially the blocking and diversion effect of trees (Pryke and Samways 2001; Sutcliffe and Thomas 1996). Cant et al. (2005) suggest that butterflies adjust flight paths to avoid such features at distances of 100 to 200 meters, while Smith et al. (2005) emphasize how important boundary permeability is to the accessibility of gardens to wildlife. This suggests that butterflies are likely to avoid entering a residential block if the outside boundary is dense, or else the butterfly will have to make rapid internal responses to the structural variability that dictates its passage, which thus will influence its residence times and garden activity.

Conclusion

This study is based on data from one garden only, and as the layout, structure, and composition of any given garden are all very different, general conclusions may be difficult. However, it is clear that the variety of use patterns and timings noted here warrants further investigation across a wider range of gardens. Dover and Fry (2001) modified the behavior of species moving through agricultural landscapes by manipulating simple landscape structures; the data presented here suggest that it is likely that individual gardens can be similarly, and easily, managed to improve opportunities for butterflies to move through them and also to encourage them to stay longer. Manipulation could be as simple as ensuring gaps between adjacent gardens or planting relevant butterfly-friendly nectar and larval food plants. Such straightforward structural modification has been identified elsewhere as successful for bird species (Daniels and Kirkpatrick 2006) and is an inherent component of successful domestic gardening.

The data also suggest that individual visitors use the garden as a throughway rather than a stopping-off point. The short time spans recorded for individuals flying through indicate that, for butterflies, the garden functions as part of a route through the urban area within a wider garden landscape habitat, rather than as an isolated oasis in the local area (however “local” is defined). There were a number of distinct features that appeared to have an effect on the directionality of flight path, including garden orientation and the presence of shrubs, trees, and hard structures. Individual garden use would therefore seem to be defined by structural imperatives as much as by nectar or food plant species content, the advice characteristically given by wildlife gardeners to ensure that butterflies use a garden.

The consistently low numbers of individual visits recorded for much of the time were surprising. Single visitors dispersed throughout the day were either very visible or else showed small bursts of activity in response to rapid changes in weather conditions (e.g., periods of sunshine), and therefore contributed to the perception of a highly active butterfly garden, even if this was not the case most of the time. (Incidentally, a few very butterfly-active days in summer that coincide with times when the garden is well-used by people may well give a similar impression.)

However, when these relatively low visitation numbers are magnified to the level of gardens within the contiguous residential block (approximately 100 gardens in this instance), and then again to the level of the urban area as a whole, and finally across the UK, the importance of urban gardens to butterflies can be clearly demonstrated.

Acknowledgments

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Figure 1: Location of the study garden within Wolverhampton, West Midlands, UK (© Crown Copyright/Database Right 2007. An Ordnance Survey/EDINA supplied service).

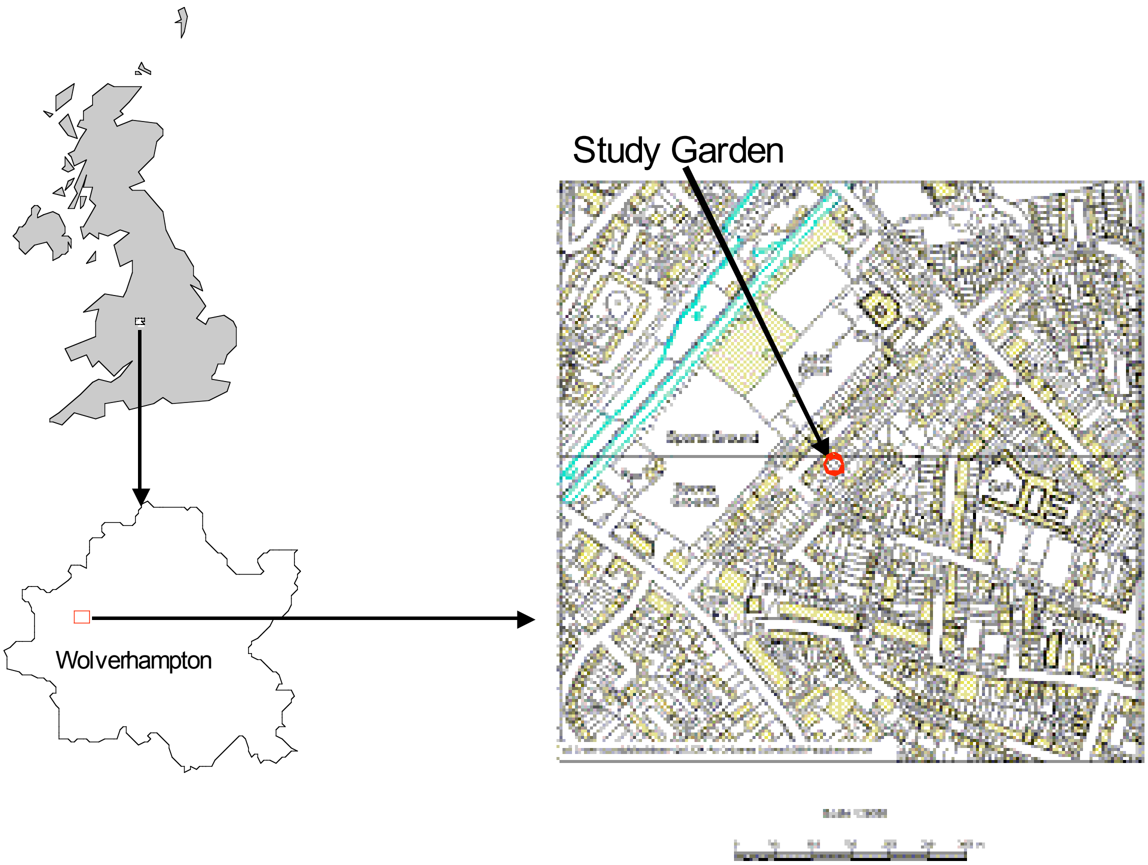

Figure 2a: Hatched areas show residential and garden buildings. Circular areas are significant garden shrubs or trees either rising above the surrounding fence or else with a strong attraction for butterflies, e.g., a series of buddleia bushes midway along southern boundary fence. Letters correspond to different species: Apple (A), Buddleia (B), Damson (D), Holly (H), Lilac (L) Plum (P), Quince (Q), Rowan (R) and Silver Birch (SB).

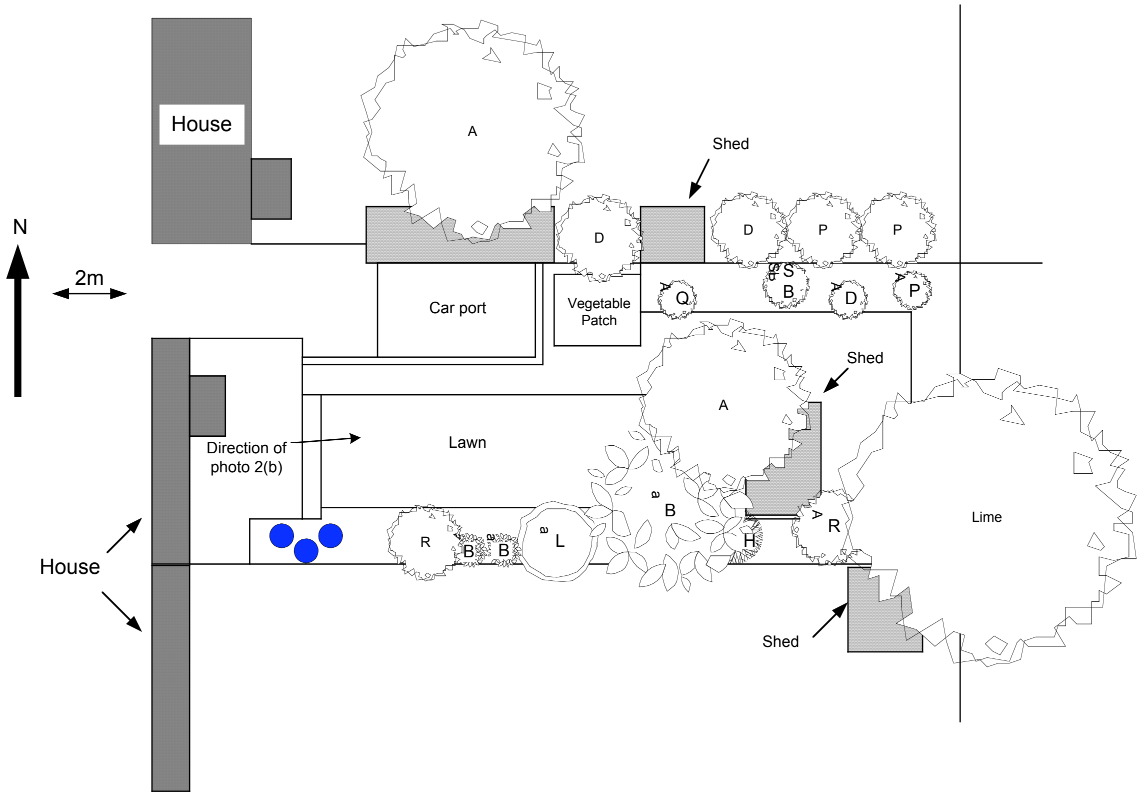

Figure 2b: Looking east along the study garden (as marked on Figure 2a). Note the dense shrubs and trees at the far end and the high, dense barrier in the garden to the north and along parts of the southern margins.

Figure 3: Number of visits to garden over all recording weeks by year, 2000–2002.

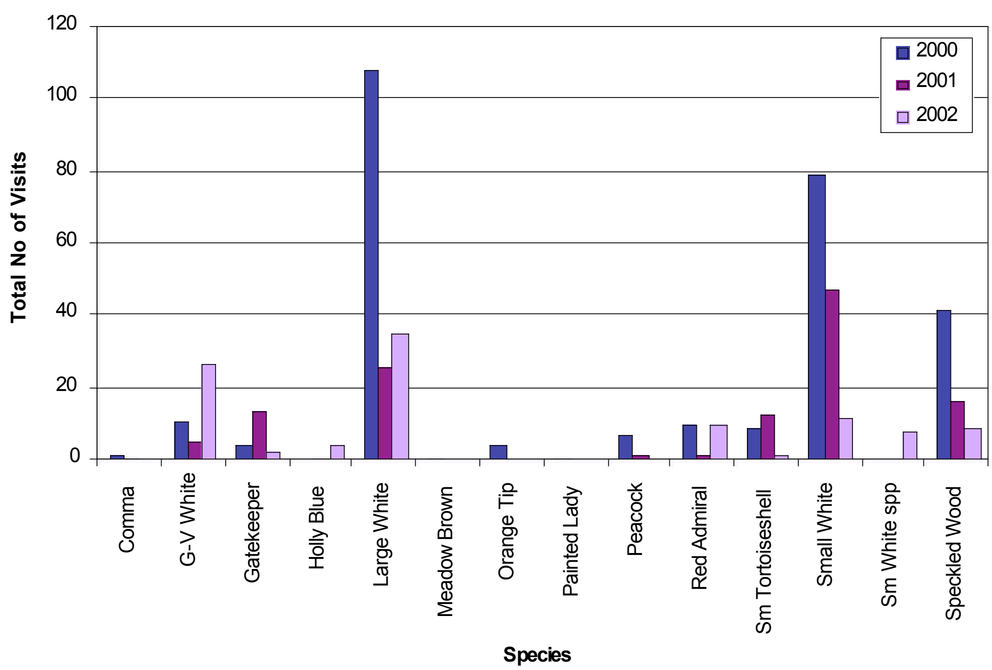

Figure 4: Mean flight times by species (±SE).

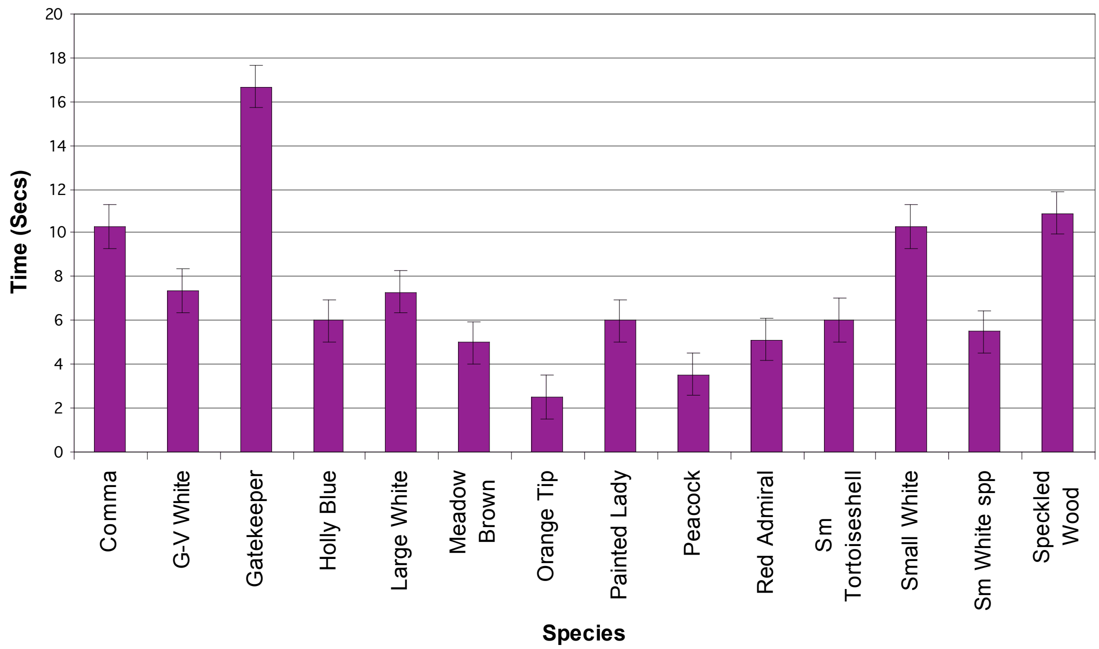

Figure 5: All butterfly visit flight paths in the observation season of 2000. Distinct corridors of activity are noticeable both across and along the garden.

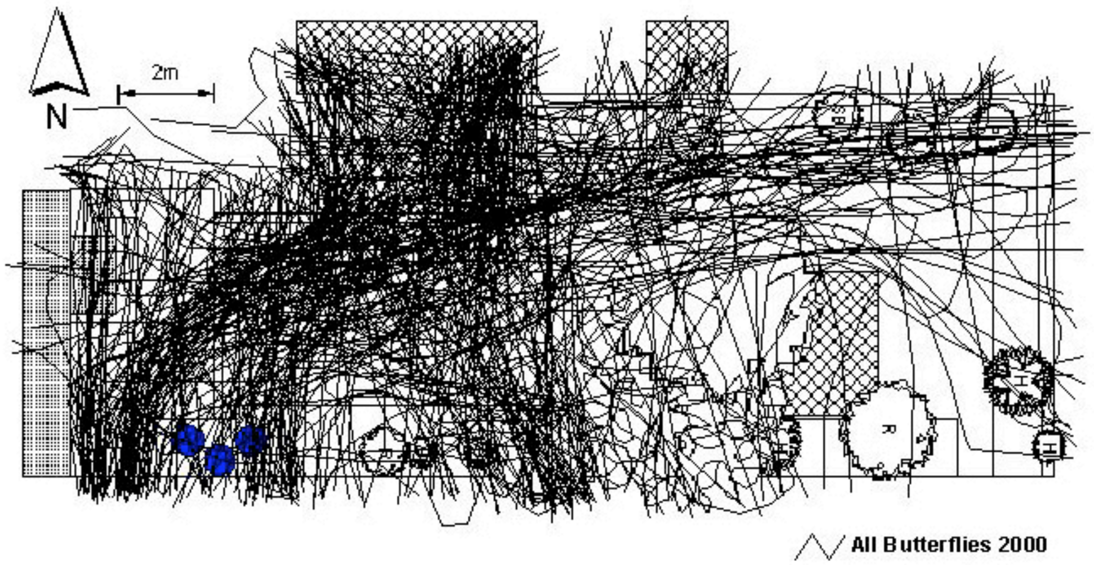

Image 1: Red Admiral feeding on hemp agrimony (*Eupatorium cannabinum*). Photo © C. Young.

Image 2: Gatekeeper feeding on late season meadowsweet (*Filipendula ulmaria*). Photo © C. Young.

Image 3: Speckled wood perched on laurel (*Prunus laurocerasus*). Photo © C. Young.

Image 4: Green-veined white feeding on marjoram (*Origanum vulgare*). Photo © C. Young.

Herpetofaunal Use of Edge and Interior Habitats in Urban Forest Remnants

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Abstract

While we know that reptiles and amphibians make use of urban forest remnants, little research has been conducted on whether certain species use edges and interiors of remnants to different extents. In our investigation, we used pitfall traps, funnel traps, and PVC pipe sampling arrays to survey the presence of herpetofauna in five urban forest remnants (between 3.0 and 16.6 hectares in size) in Gainesville, Florida, during the summers of 2005 and 2006. We then compared the average daily relative abundances of individual species and taxa groups (at order and suborder levels and also at the family level), as well as species richness and compositional similarity at edge locations (defined as 20 to 40 meters toward the interior from the remnant boundary) and interior locations (defined as over 40 meters from the remnant boundary). Our results showed that edge and interior locations did not differ in either the relative abundance of individual herpetofaunal species and taxa groups or species richness. In addition, our analysis of species composition showed that most remnants had very similar compositions at their edges and interiors. Furthermore, our vegetative analyses showed very few vegetative differences between edge and interior locations. Despite the lack of a difference in edge and interior habitat use by herpetofauna, a finding possibly due to a lack of difference in vegetative structure, study results did show that urban forest remnants serve as habitat to some herpetofaunal species that can tolerate conditions within small patches.

Key words: urban, herpetofauna, reptiles, amphibians, herps, edge, interior, habitat isolation, habitat use, forest remnant, habitat fragmentation

Introduction

Reptiles and amphibians face numerous challenges coexisting with an urbanizing world (Rubbo and Kiesecker 2005; McKinney 2006). Research has shown that herpetofauna can be negatively affected by the habitat isolation created by urbanization. Barriers to the dispersal of animals such as roadways (Houlahan and Findlay 2003; Ficetola and De Bernardi 2004; Cushman 2006; Parris 2006), the reduction of water and wetland quality through adjacent land use (Houlahan and Findlay 2003), and the alteration of water level and flow patterns (Richter and Azous 1995; Delis et al. 1996; Riley et al. 2005) all cause habitat degradation that particularly affects herpetofauna. Much attention has been paid to the effects of urbanization on amphibians because their need for access to water in which to breed makes their survival vulnerable to ecological alterations, and urbanization can have significant impacts on water quality (Riley et al. 2005; Rubbo and Kiesecker 2005). However, habitat fragmentation and other anthropogenic threats such as environmental pollution, over-harvesting, and the introduction of non-indigenous species put both reptiles and amphibians at substantial risk (Gibbons et al. 2000). The IUCN estimates that one third of herpetofaunal species worldwide are threatened with extinction (Baillie et al. 2004; Cushman 2006).

Urban and Edge Effects on Herpetofauna

Reptiles and amphibians can be found within forest remnants (Schlaepfer and Gavin 2001; Lehtinen et al. 2003; Urbina-Cardona et al. 2006), including fragments of forests completely surrounded by urbanization (Enge et al. 2004; Ficetola and De Bernardi 2004; Parris 2006). Habitat fragmentation, which can be caused by urbanization, creates a higher amount of edge habitat than interior habitat in urban forest remnants. Edges have long been recognized as having higher diversities and higher abundances of species than habitat interiors, particularly of game species and birds (Leopold 1933; Lay 1938; Yahner 1988). This pattern is partially due to factors such as the increased sunlight exposure and increased emergent vegetation at edges, as well as the increased abundance of invertebrates there (Murcia 1995; Harper et al. 2005). However, for herpetofauna, particularly amphibians, interior habitats generally offer cooler, moister conditions, and therefore may be more conducive to survival, particularly during dry periods (Schlaepfer and Gavin 2001; Lehtinen et al. 2003).

Past research comparing herpetofaunal use of edges and interiors of forest remnants has shown that species of herps can respond in different ways to habitat edges, and species can thus be categorized as edge-associated, interior-associated, and edge-indifferent (Schlaepfer and Gavin 2001; Lehtinen et al. 2003; Urbina-Cardona et al. 2006). These findings have varied depending upon the ecological system that was studied as well as the season in which it was studied. For example, Lehtinen et al. (2003) and Schlaepfer and Gavin (2001) found herp species to use edges and interiors of forest remnants differently within desert and pasture matrices, respectively, but these results were highly dependent upon whether it was the wet or dry season. In addition, Urbina-Cardona et al. (2006) found that groups of reptiles and amphibians used edges and interiors of remnants to differing extents within pasture matrices throughout the year, but that the variables influencing these patterns changed for the wet and dry seasons. Currently, very little is known about whether individual species or taxa groups use interior or edge areas of urban forest remnants differently.

Our objective in this study was to determine whether species and taxa groups of amphibians and reptiles use edge and interior habitats differently within urban forest remnants during the summer.

Methods

Study Site

We focused our study on five forest remnants on the University of Florida campus, located in Gainesville, Florida: Harmonic Woods (3.7 hectares); Graham Woods (3.0 hectares); Bartram-Carr Woods (3.5 hectares); Lake Alice Conservation Area (11.3 hectares); and Bivens Forest (16.6 hectares) (Figure 1). The University of Florida Gainesville campus is located in north-central peninsular Florida, where summers are hot, humid, and generally rainy. Two of the three smallest remnants, Harmonic Woods and Bartram-Carr Woods, consist mainly of upland mixed pine-hardwood forest, and both contain or are immediately adjacent to small streams or low-lying areas. The third small patch, Graham Woods, is made up of a mixture of bottomland hardwoods and upland mixed pine-hardwoods, and encloses a small network of seasonally fluctuating but permanent streams. One of the two largest remnants, Lake Alice Conservation Area, contains upland mixed pine-hardwood forest and some regenerating clear-cut habitat, and has some flood-plain forest created by a large marsh (25 hectares) adjacent to the remnant. The other large remnant, Bivens Forest, consists of an interior bottomland-hardwood swamp ringed by mixed pine-hardwood forest on three of its four edges. Its fourth edge is adjacent to a lake. All remnants except Harmonic Woods are subject to occasional flooding.

Herpetofaunal Sampling

We sampled herpetofauna from May to August during the summers of 2005 and 2006, using drift fence arrays with pitfall traps and funnel traps, along with polyvinyl chloride (PVC) pipe refugia to sample for tree frogs. We made drift fences out of approximately 30-centimeter-wide silt fencing (Enge 1997). For funnel traps, we modified the format described by Enge (1997), using aluminum window screening approximately 76 centimeters in length to build cylindrical traps of the same length with a funnel in one end and with the other end closed. Following a modification of a design by Moseley et al. (2003), we formed arrays in the shape of a Y, with three 7.6-meter-long drift fence wings conjoined around a single pitfall trap and placed at 120-degree angles to each other. We placed funnel traps at the distal ends of each wing, making sure the open funnels were flush to the ground and equally straddling the ends of the fences (Johnson, personal communication).

We made pitfall traps with 19.1-liter plastic buckets. To prevent desiccation of captured specimens, we placed a dampened sponge inside each trap and we re-dampened sponges each sampling day as necessary. Originally, we drilled holes into the bottom of the buckets for drainage. However, in remnants with a high ground-water level, water would flood the bucket from the bottom up. Therefore, in places that tended to flood, we installed buckets without holes in the bottom. We used iron rebar stakes to hold the buckets in the ground against hydrostatic pressure (Enge personal communication). We scooped out the rainwater collected in pitfall traps each sampling day as necessary.

We constructed PVC pipe refugia to attract various species of tree frogs. We used pipes of both 2.5 centimeter and 5 centimeter diameter-widths, with lengths of about 76 centimeters, driving them into the ground to depths that allowed the pipes to stand up on their own (Zacharow et al. 2003). We placed one pipe of each diameter width between each wing of the Y-shaped fence array (Moseley et al. 2003), resulting in six total PVC pipes per sampling array.

To compare edge and interior locations (Figure 2), we considered the first 40 meters from the remnant boundary toward the interior to be “edge,” and we deemed “interior” all space over 40 meters from the boundary of the remnant. We placed arrays at edge locations between 20 and 40 meters from boundary edges because of the proximity of the remnants to the urban environment and to protect against potential human interference (i.e., the public disturbing traps or trapped animals). Except for this specification, we placed sampling arrays randomly within edge and interior areas of the remnants. To assure some degree of equal sampling effort per forest remnant, we followed a one-array-per-2-hectare ratio, with a maximum of four arrays per remnant. We ensured that all arrays within the remnants were at least 100 meters apart from each other (Campbell and Christman 1982), though in two remnants, Lake Alice Conservation Area and Bivens Forest, unsuitable substrates permitted a maximum distance of only 80 meters between sampling arrays. Employing these parameters, we placed a total of seven interior and seven edge arrays in five forest remnants around the University of Florida campus.

We opened traps for periods of four days each and checked them systematically every day so that these observations coincided with the approximate time they had been set the day before. This ensured that all traps were open for the same amount of time each day (approximately 24 hours), allowing for equal sampling effort per trap. On the fourth day, we closed traps until the next sampling period. Each day, we identified the species of captured specimens and then promptly released them without marking them. We sampled herps from mid-May through early August. Arrays were open for 23 days during the summer of 2005 and for 24 days during the summer of 2006. Occasionally, flooding from heavy rains forced us to close some traps. In this event, we reopened the closed traps during the same week for the amount of sampling time lost to inclement weather. When trapped specimens were negatively affected by the presence of ants at sampling locations, we were forced to close funnel traps indefinitely.

Vegetation Sampling

To determine whether there were structural differences between the edges and interiors, we conducted vegetation sampling at randomly assigned locations in both. Due to logistical constraints, we assigned vegetation sampling points in the same manner and ratio as we had herpetofaunal sampling arrays; that is, one sample point per 2 hectares, with a maximum of four sample points per remnant. To ensure that sampling points would be contained within both edge and interior habitats, edge sampling points were located 20 meters from remnant boundaries and interior sampling points were located at least 60 meters from remnant boundaries. This system resulted in two points each for Harmonic Woods, Graham Woods, and Bartram-Carr Woods, and four points each for Lake Alice Conservation Area and Bivens Forest.

We sampled woody shrub (defined as being ≥ 1 meter in height, < 8 centimeters in diameter at breast height [DBH]) stem density on two randomly assigned, perpendicular 20-meter transects leading from the central sampling point location (James and Shugart 1970). We measured the number of trees (defined as being > 8 centimeters DBH) and standing snags in a 10-meter-radius subplot centered on the central sample point location. We scaled all the measures of shrub, tree, and snag density to densities per hectare. Following modified procedures from Tilghman (1987) and James and Shugart (1970), we randomly established four 1-square-meter subplots within 20 meters of each sample point center, and estimated several measures at each subplot. We counted woody shrub stems (≤ 8 centimeters DBH) in order to document shrubs less than 1 meter in height, and averaged counts over all four subplots. We used a spherical densiometer to measure the overstory canopy in all cardinal directions. If there was a significant mid-story (defined as < 5 meters) that prevented us from reasonably sighting the overstory canopy, then we used the location within 5 meters of the point that gave us the clearest view of the canopy. We averaged recorded measures per 1-square-meter subplot, and then per sample point.

Modifying the methods of Robel et al. (1970), we accounted for understory shrub cover by measuring the number of decimeters in each 1/2-meter section of a marked sighting pole (2 meters in height) that were more than 25% obstructed by vegetation. We placed the pole vertically at the center of each 1-square-meter subplot and observed to a distance of 4 meters, looking from a height of 1 meter and from each cardinal direction. We averaged data per 1/2-meter section of each 1-square-meter subplot, and then averaged per sample point. To account for vertical structure, we visually noted the presence of the following structural categories that were at < 1 meter in height, between 1 meter and 5 meters in height, and at ≥ 5 meters in height: grass, forbs, dead debris, shrubs (woody or herbaceous), trees (defined as plants with woody stems > 8 centimeters DBH), and vines. We visually estimated ground cover by classes representing percentages of cover (including 0%, > 0–10%, 11–25%, 26–50%, 51–75%, and > 75%) of bare ground, grass, dead debris, forbs, shrubs (woody or herbaceous), trees (woody stems > 8 centimeters DBH), and vines.

Herpetofauna Analyses

We conducted data analyses comparing the average daily relative abundance of individual species, at the order/suborder taxa level (including snakes, frogs, and lizards) and the family taxa level (ranids, hylids, skinks, and anoles) at both edges and interiors. We also analyzed overall species richness, comparing that of the edges to that of the interiors. We generated average daily relative abundances for each species by summing the count data for all edge and interior sampling locations of a given forest remnant (e.g., Harmonic Woods), and then dividing this by the total trap effort (i.e., number of trap days) carried out at the edge and interior locations of that remnant. Total trap effort was modified according to the sampling methodologies employed (e.g., 3 funnel traps and 1 pitfall trap = 4/4, or 100% operational) on each trap day. For example, if a total of 10 frogs were caught over 4 days in which one pitfall trap and only 2 of the 3 funnel traps were open, then we would calculate this average as: 10/(4 * [3/4]) = 3.33.

For most species, three funnel traps and one pitfall trap per array were the applicable sampling methodologies at each array. For hylids (tree frogs), our sampling involved only the 6 PVC pipes per array (e.g., 6 pipes = 6/6 or 100% operational). We caught brown anolis lizards (*Anolis sagrei*) using all sampling methods (e.g., 3 funnel traps, 1 pitfall, and 6 PVC pipes = 10/10 or 100% operational). We used this approach because our sampling effort at each array was occasionally reduced when traps or pipes were temporarily inoperable due to extreme weather or unknown disturbances (e.g., raccoon interference) or were intentionally removed due to ant predation.

In Harmonic Woods, Graham Woods, and Bartram-Carr Woods, there were only two herpetofaunal sampling arrays—one at the edge, one at the interior. Lake Alice Conservation Area and Bivens Forest were larger and therefore allowed for two sampling arrays at each edge and interior location. However, we inadvertently positioned one edge location in each of the larger remnants (at Bivens Forest and Lake Alice Conservation Area) too far from the boundaries of these remnants (that is, more than 20–40 meters from the patch boundaries). We excluded these arrays from our analysis in order to prevent undue bias on any actual difference in herpetofaunal habitat use that occurred between edges and interiors. Also, we sampled Bartram-Carr Woods only through the first week of July in 2006 because of unanticipated construction that began in that remnant.

We entered calculated data into a one-way ANOVA model blocked for forest remnant (that is, it was controlled statistically for effects contributed by individual remnants) in which average daily relative abundance was the dependent variable and edge or interior location was the independent variable. Because we were not interested in the contribution of sampling year on the variability of the data, we averaged the relative abundances for each analyzed species and group over both years. We tested the data for normality with the Ryan-Joiner test and for equal-variance with Levene's test. We attempted to use square-root transformations for non-normal and heteroskedastic distributions for individual species and groups. We used the non-parametric Friedman test to analyze species and groups that could not meet parametric test assumptions after transformation. Because there were five sampled forest remnants with both edge and interior locations, this resulted in a total of ten possible forest remnant locations. In order to prevent normality issues arising from too many zeros in the data, we statistically analyzed individual groups in each level of analysis only if they were present in at least half (5) of the ten possible forest remnant locations.

We calculated species richness at both the edge and interior for each forest remnant and entered it into a one-way ANOVA model blocked by forest remnant in which number of species was the dependant variable and edge or interior location was the independent variable ($\alpha = 0.1$). Similar to the count data, we averaged species richness data over both years. We tested normality and variance assumptions as previously described.

In order to gauge similarities in species assemblages at edges and interiors, we computed Horn-Morisita similarity index values between edges and interiors within each remnant. To do this, we employed the R Statistical Program, using the Vegan Community Analysis package. We chose the Horn-Morisita similarity index because it incorporates both presence/absence and abundance information, and we felt it was a more complete approach to computing similarity than other indices that employ only presence/absence information.

Vegetation Analyses

We analyzed measurements of shrub, tree, and snag densities, visual obstruction in each 1/2-meter height section, and canopy cover with the same ANOVA model we used for the analysis of herpetofauna. Normality and equal variance assumptions were also checked in a similar manner to our herpetofauna analysis. Comparing the vertical structure of edges and interiors, we analyzed single structural categories for the vertical heights noted (< 1 meter in height, between 1 and 5 meters in height, and ≥ 5 meters in height). In a manner similar to Tilghman (1987) and Karr (1968), in order to account for dead debris, trees, and vines we calculated the proportion of occurrence, that is, the proportion of 1-square-meter subplots in which each sample variable was found, for all subplots measured within a given remnant edge or interior. This created an index of relative presence in the vertical strata between 0 and 300. For example, if trees occurred in 25% of subplots at the < 1 meter height, 50% of subplots at the level between 1 and 5 meters in height, and 100% of subplots at the > 5-meters height, the index value for the plot for trees would be: (25 + 50 + 100) = 175. Because shrubs occurred only at the lower two levels, we analyzed the presence of shrubs out of an index of 200. Because grass and forbs occurred only at the < 1-meter height level, the presence of each of these components was analyzed out of an index of 100.

Next, we analyzed total vegetation structure at each vertical height level by considering the following two variables: vegetation structure alone (only live vegetation categories) and all structure (live vegetation categories + dead debris). For vegetation structure, we calculated the proportion of 1-square-meter subplots at which a structural category occurred at a given height level to create an index between 0 and 500 at the < 1-meter height level (all vegetation components); between 0 and 300 at the level between 1 and 5 meters in height (shrubs, trees, and vines); and between 0 and 200 at the > 5-meters height level (only trees and vines). We analyzed total structure (live vegetation categories + dead debris) at each height level in a similar way, but we calculated the relative structure out of an index between 0 and 600 at the < 1-meter height level, between 0 and 400 at the level between 1 and 5 meters in height, and between 0 and 300 at the > 5-meters height level because of the addition of dead debris. We analyzed the calculated index values for each category with the same ANOVA model we used in the herpetofaunal analyses, and we checked normality and equal variance assumptions in a similar manner to that used in the analysis of herpetofauna.

To analyze ground cover, we separately compared each cover class (0%, > 0–10%, 11–25%, 26–50%, 51–75%, and > 75%) of each ground cover variable (bare ground, grass, dead debris, forbs, shrubs, trees and vines) at edges and interiors. As a singular example, for dead debris we compared the 50–75% cover class between remnant edges and interiors. To do this, we calculated the proportion of occurrence (i.e., how many 1-square-meter subplots a cover class occurred in) of each cover class per cover variable over the four 1-square-meter subplots at each sample point location. We then calculated the average per remnant edge and interior. Due to an inconsistency in data collection, we were unable to analyze the > 0–10% and > 10–25% cover classes for ground cover variables. We entered the remaining data into the same ANOVA model previously described. For all statistical tests, we checked normality and equal variance assumptions as described above, and we tested non-normal distributions unaffected by square-root or log transformation with the non-parametric Friedman test. We used an $\alpha = 0.1$ for all statistical tests.

Results

Over the summers of 2005 and 2006, we checked 12 arrays on a total of 552.5 trapping days for tree frogs, 548.6 trapping days for brown anoles (*Anolis sagrei*), and 542.75 trapping days for all remaining species. We caught a total of 24 species in all arrays and detected an additional 7 species outside of the arrays (Appendix I). We did not include the species we detected outside of arrays in our analyses.

Individual Species

Only six species were present in enough forest remnant locations for both years to be analyzed individually. After analyzing the occurrences of brown anole (*Anolis sagrei*), greenhouse frog (*Eleutherodactylus planirostris*), green treefrog (*Hyla cinerea*), squirrel treefrog (*Hyla squirella*), bronze frog (*Lithobates clamitans*), and common ground skink (*Scincella lateralis*), we found that no species had significantly higher relative abundances at either edge or interior locations (Table 1).

Order and Family Subgroups

In order-level subgroups, including the order Anura (frogs) and the suborders Serpentes (snakes) and Lacertilia (lizards) of the order Squamata (scaled reptiles), we found that none of the groups showed significantly higher relative abundance at edges or interiors (Table 1). Among family-level subgroups, including the families Ranidae (true frogs), Hylidae (tree frogs), Scincidae (skinks), and Polychrotidae (anolis lizards), no group revealed significantly higher daily relative abundance at edges versus interiors (Table 1).

Species Richness and Composition

The number of species we analyzed at edge and interior locations was not significantly different (Table 1). The Horn similarity index, which we used to compare species compositional similarity between edges and interiors, is based on a scale of 0 to 1, with 0 representing a completely different species composition and 1 representing completely identical compositions. When we calculated the similarities between the edges and interiors of individual remnants, we found that similarity values ranged from 0.520 to 0.890, with a mean value of 0.775 (Table 2). This indicates that herpetofaunal species assemblages were highly similar at the edges and interiors of all the remnants we considered, with the exception of Lake Alice Conservation Area. Lake Alice Conservation Area, with a Horn Similarity Value of 0.520, had only a moderately similar species assemblage at its edge and interior.

Vegetation

Our analysis of average shrub stem density (< 1 m and ≥ 1 m in height), canopy cover, visual obstruction, vertical vegetative structure, and density of trees and snags showed no significant differences in the vegetation characteristics of the edge and interior areas. When we compared ground cover, we found a significantly greater occurrence of vines in interior locations—making up between 25 and 50% of the ground cover there—as compared to at edge locations (d.f. = 1, F = 7.08, P = 0.056). All other tests were not significant (P > 0.1).

Discussion

We found no difference in herpetofaunal use of edge or interior habitat for any individual species, family-level taxa group, or order-level taxa group. We also found no difference in species richness between edges and interiors. Further, our species composition similarity index values at edges and interiors ranged from moderately similar to highly similar, which indicates that the assemblage of herpetofaunal species at the edges and interiors of most remnants was largely the same. Therefore, from 20 meters up to approximately 100 meters from the edge, the herpetofauna analyzed in our study do not appear to use the edges or interiors of these small urban remnants differently.

One possible reason for the herps' lack of discrimination in these remnants could be the small amount of structural habitat differences found between the edge and interior habitats in this study, particularly in terms of variables such as canopy cover. In previous research of edge versus interior habitat segregation by herpetofauna in forest fragments, canopy cover has tended to be denser at interior locations (Schlaepfer and Gavin 2001;

Urbina-Cardona et al. 2006). Denser canopy cover was directly related to higher relative humidity at interiors by Urbina-Cardona et al. (2006), and was implied to have contributed to lower temperatures and higher humidity at interiors by Schlaepfer and Gavin (2001) and Lehtinen et al. (2003). These interior conditions partially drove the segregation of edge and interior habitat by some species of herpetofauna in these studies, at least on a seasonal basis. In addition, understory vegetation cover being denser at edges than interiors may also have contributed to habitat segregation in the study by Schlaepfer and Gavin (2001). The lack of such vegetative differences during the summer in our study suggests that these forest remnants are relatively homogenous up to 100 meters from the remnant boundary, and therefore may have contributed to the lack of significant difference in use of edge and interior habitats by herpetofauna.

Moreover, we sampled the herpetofauna in our study only during the summer rainy season, and species during this season, particularly amphibians, may have been inclined to use the entire forest remnant if they were dispersing in search of wetlands for breeding activities. This assertion is consistent with Lehtinen et al. (2003), who found that several species of frogs and reptiles were significant edge habitat avoiders during the dry season in isolated tropical forest patches in Madagascar, while most species of frogs and some species of reptiles were either edge-indifferent or interior-avoiding during the wet season. Lehtinen et al. (2003) reasoned that moisture-sensitive herpetofauna would be more willing to disperse to warmer edge habitats during the wet breeding season, whereas these species preferred the cooler, moister conditions offered by forest interiors during the dry season. Further, the study by Lehtinen et al. (2003) was conducted within forest remnants surrounded by a "hard" matrix of desert-like "sand-scrub" that may have been functionally similar to the urban matrix of buildings and roads surrounding several of the remnants in our study. These results indicate the need for additional research during both the wet and dry seasons in urban remnants.

Also, in our study only six species were sufficiently common to be analyzed individually. Our sampling methodology may not have been effective in capturing other species, particularly species that are largely fossorial, such as the Eastern glass lizard (*Ophisaurus ventralis*), or aquatic, like two-toed amphiuma (*Amphiuma means*). Other species of herpetofauna common to the Gainesville area simply may not be abundant in these urban remnants due to habitat isolation. Of the species we analyzed, none are overly rare in Florida, and two of them, brown anoles (*Anolis sagrei*) and greenhouse frogs (*Eleutherodactylus planirostris*), are introduced species and are often associated with disturbed areas. Given the lack of difference in vegetation at edges and interiors, it is therefore not unexpected that these species showed no differentiation in habitat use.

Lastly, the lack of a significant difference in habitat use of herpetofauna between edge and interior habitats may have been influenced by our method of determining edge and interior spaces. Because we considered the threat of trap disturbance by humans significant in this group of remnants, we determined it was necessary to place traps at edge locations at least 20 meters from patch boundaries. In previous studies comparing herpetofaunal habitat use of remnant edges and interiors, edge effects have been detected only a few meters (Lehtinen et al. 2003; Schlaepfer and Gavin 2001) and up to 20 meters from remnant borders (Urbina-Cardona et al. 2006). In our study, we did not account for potential variability in herpetofaunal use and compositional similarity for fewer than 20 meters from remnant boundaries, and we did not directly consider vegetation characteristics fewer than 20 meters from remnant boundaries. Perhaps the first 20 meters of the urban forest edge may demonstrate different herpetofaunal abundance, composition, and richness than interior locations. Future research should study this 0 to 20 meter range, but only in urban remnants where potential human disturbance of traps is minimal.

Despite the lack of an apparent difference in edge and interior habitat use in this study, the results reveal that urban forest remnants are used by a number of different herpetofaunal species. Although only six species were included in individual analysis here, a total of 31 species were noted over the course of two field seasons in these remnants, including seven species that were not caught in sampling arrays but were observed coincidentally during the sampling seasons (Appendix I).

Edge locations in urban remnants can provide herpetofauna access to habitat with high exposure to sunlight in the adjacent matrix, if not at edges themselves. It should be noted, however, that the presence of herpetofauna within these remnants is not necessarily an indicator of habitat quality. (For example, we did not determine whether or not these remnants serve as population sources or sinks.) The conservation of amphibians and reptiles in urban forests contributes toward the diversity of the surrounding urban environment, but studies are lacking in the literature. More research should be conducted on how herpetofauna use urban habitats.

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Table 1. Average daily relative abundance of herpetofauna species and groups, as well as species richness between edges and interiors of five urban forest remnants in Gainesville, Florida.

Shown are the means and standard error (SE) values for the average daily abundances and species richness of both edges and interiors, the test statistics (T.S.) and associated P-values for all individually analyzed species and groups. Also shown are the numbers of species per taxa group. Unless noted, statistical test is one-way ANOVA. For all tests, df = 1 and n = 5 for edge and interior areas.

Taxa Group	Number of Species per group	Taxa Group/ Species/ Species Richness	Edge		SE	Interior		SE	T.S.	P
Order-level	10	Anura	0.59	±	0.27	0.62	±	0.33	0.19	0.69
	7	Squamata, suborder Serpentes	0.03	±	0.02	0.04	±	0.01	0.17	0.70
	5	Squamata, suborder Lacertilia	0.43	±	0.13	0.23	±	0.09	1.43	0.30
Family-level	2	Hylidae**	0.27	±	0.19	0.36	±	0.32	0.00	1.00
	2	Polychrotidae*	0.15	±	0.04	0.12	±	0.06	0.19	0.69
	3	Ranidae	0.21	±	0.08	0.17	±	0.07	3.39	0.14
	3	Scincidae	0.27	±	0.11	0.11	±	0.05	2.63	0.18
		*Anolis sagrei**	0.15	±	0.03	0.12	±	0.06	0.42	0.55
		*Eleutherodactylus planirostris**	0.07	±	0.03	0.05	±	0.03	0.94	0.39
		*Hyla cinerea***	0.02	±	0.02	0.03	±	0.01	1.00	0.32
		*Hyla squirella**	0.25	±	0.19	0.33	±	0.31	0.33	0.56
		Lithobates clamitans	0.13	±	0.04	0.13	±	0.04	0.01	0.95
		Scincella lateralis	0.24	±	0.12	0.09	±	0.05	1.84	0.25
		Species Richness	**7.40**	**±**	**1.75**	**8.20**	**±**	**2.03**	**1.43**	**0.30**

*square-root transformed
**tested with non-parametric Friedman test

Table 2. Horn-Morisita Index compositional similarity values for species assemblages at edges and interiors within urban forest remnants in Gainesville, Florida. Values closer to 1 indicate similar species composition.

Remnant	Horn Similarity Index Value
Harmonic Woods	0.855
Graham Woods	0.741
Bartram-Carr Woods	0.863
Bivens Forest	0.897
Lake Alice Conservation Area	0.520
Mean	**0.775**

Appendix I. All species of herpetofauna detected during the summers of 2005 and 2006 in urban forest remnants in Gainesville, Florida. (Assume detection by trapping arrays unless otherwise noted.)

Scientific Name	Common Name	Order-level taxa Group	Family-level taxa group
*Alligator mississipiensis***	American Alligator	N/A	N/A
Anolis carolinensis	Green Anole	Lizard	Polychrotidae
Anolis sagrei	Cuban Brown Anole	Lizard	Polychrotidae
*Apalone ferox**	Florida Softshell Turtle	Turtle	N/A
Bufo terrestris	Southern Toad	Anura	Bufonidae*
Bufo quercicus	Oak Toad	Anura	Bufonidae*
Coluber constrictor	Black Racer	Snake	N/A
Diadolphis punctatus	Southern Ringnecked Snake	Snake	N/A
Eleutherodactylus planirostris	Greenhouse Frog	Anura	N/A
*Eurycea quadridigitata***	Dwarf Salamander	N/A	N/A
Farancia abacura	Mud Snake	Snake	N/A
Gastrophryne carolinensis	Eastern Narrowmouth Toad	Anura	N/A
Hyla cinerea	Green Treefrog	Anura	Hylidae
*Hyla gratiosa***	Barking Treefrog	Anura	Hylidae
Hyla squirella	Squirrel Treefrog	Anura	Hylidae
Lithobates catesbeianus	Bull Frog	Anura	Randiae
Lithobates clamitans	Bronze Frog	Anura	Ranidae
Lithobates sphenocephalus	Southern Leopard Frog	Anura	Ranidae
*Nerodia fasciata fasciata***	Southern Banded Watersnake	N/A	N/A
*Nerodia fasciata pictiventris***	Florida Banded Watersnake	N/A	N/A
Plestiodon fasciatus	Five-lined Skink	Lizard	Scincidae
Plestiodon laticeps	Broadheaded Skink	Lizard	Scincidae
Rhadinaea flavilata	Pinewoods Snake	Snake	N/A
Scaphiopus holbrookii	Eastern Spadefoot Toad	Anura	N/A
Scincella lateralis	Common Ground Skink	Lizard	Scincidae
*Sternotherus minor***	Loggerhead Musk Turtle	N/A	N/A
Storeria dekayi victa	Florida Brown Snake	Snake	N/A
*Terrapene carolina bauri**	Florida Box Turtle	Turtle	N/A
Thamnophis sauritus	Eastern Ribbon Snake	Snake	N/A
Thamnophis sirtalis	Eastern Garter Snake	Snake	N/A
*Trachemys scripta scripta***	Yellow-bellied Slider	N/A	N/A

*species caught in traps, but insufficient data for analysis
**species incidentally detected in urban forest remnants during sampling periods; not included in analysis

Figure 1: Forest remnants included in herpetofaunal sampling on the University of Florida campus in Gainesville, Florida.

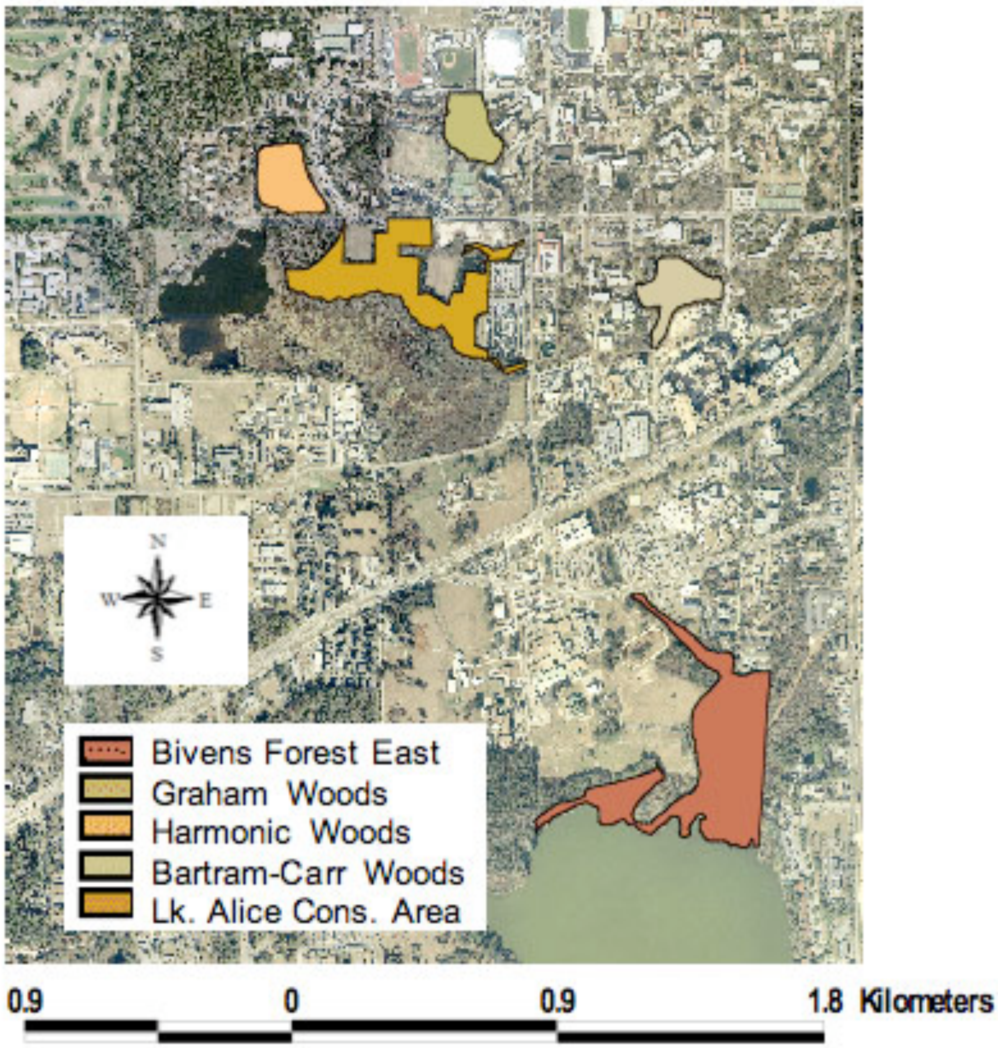

Figure 2: Illustration of edge and interior locations of herpetofaunal sampling arrays within forest remnants in Gainesville, Florida. An edge array was placed within 20 to 40 meters of the boundary of a remnant, and an interior array was situated more than 40 meters from a remnant boundary. Arrays were positioned at least 100 meters apart to maintain independence from each other.

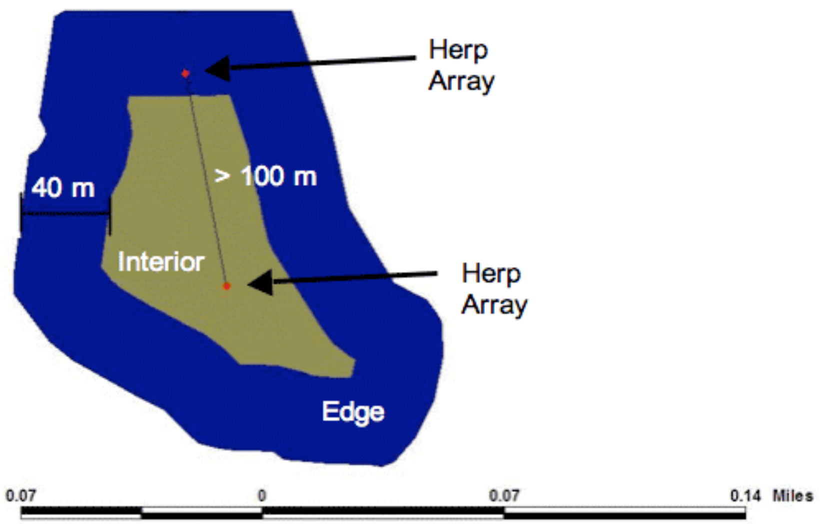

Image 1: Example of herpetofaunal sampling array used during this survey. Drift fences, center pitfall bucket, and PVC pipes are visible.

Image 2: American alligator (*Alligator mississippiensis*) noted swimming up a stream bordering Bartram-Carr Woods.

Image 3: Squirrel treefrog (*Hyla squirella*) found in PVC pipe at Bivens Forest.

Image 4: Eastern narrowmouth toad (*Gastrophryne carolinensis*) captured at Bivens Forest.

Image 5: Southern leopard frog (*Lithobates sphenocephalus*) captured at Lake Alice Conservation Area.

Image 6: Florida banded watersnake (*Nerodia fasciata pictiventris*) noted at Lake Alice Conservation Area.

Image 7: Green treefrog (*Hyla cinerea*) noted in PVC pipe in University of Florida forest remnant; not included in analysis.

Image 8: Bronze frog (*Lithobates clamitans*) captured at Bivens Forest.

A Baseline Characterization Approach to Wetland Enhancement in an Urban Watershed

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Abstract

During the 20th century, more acres of forested wetlands were lost than any other category of wetland, yet restoration or creation of this wetland type has been notably unsuccessful. Restoration of riparian forested wetlands that are located within highly urbanized landscapes is particularly problematic, due to the stresses placed on the wetland by historical alterations and disturbances and by current watershed land uses. The Teaneck Creek Conservancy has partnered with scientists and engineers at Rutgers University to provide a baseline characterization of the 46-acre Conservancy site located within Bergen County, New Jersey's Overpeck Park. The project goal is to rehabilitate 20 acres of forest and scrub/shrub wetland by establishing hydrologic conditions typically found in a temperate forested riparian corridor, on a site whose surrounding land use is categorized as 95 percent urban. To achieve the project goal, hydrologic connections must be reestablished between the creek and the interior surface and groundwater, and surface elevations must be lowered, historical debris removed, and native vegetation established to replace invasive species. This paper reviews briefly the current status of forested wetland restoration and the obstacles to achieving successful restoration of these ecosystems. We also describe the baseline characterization being conducted for the Teaneck Creek project to support efforts to establish a sustainable urban wetland system on the Conservancy site.

Key words: urban wetland, urbanization, riparian forest, restoration, hydrology, scrub/shrub wetland, restoration/creation

Introduction

Wetlands provide numerous benefits to humans (Costanza et al. 1997). Wetlands' capacity to improve water quality, provide flood storage, retain and remove nitrogen, host wildlife habitat, and promote the general preservation of diminishing open space (Hammer 1996; Richardson and Vepraskas 2001) is of particular importance when a wetland is situated in a highly urbanized area, as is the Teaneck Creek watershed. The 46-acre Teaneck Creek Conservancy (TCC) restoration site is in Overpeck Park in Bergen County, New Jersey, which is located within the New York–New Jersey metropolitan area, one of the most densely populated urban regions in the world. Teaneck Creek and its wetland system are surrounded by land use that is categorized as 95% urban (Figure

1). Over the past two centuries, these wetlands have been influenced by a number of anthropogenic impacts and have served as a repository for multiple layers of various fill materials (Arnold this volume). The effects of this historic degradation are critical factors in determining whether and how the wetlands on this site can be restored/enhanced (Wolin and Mackeigan 2005), and they dictate to some degree the actions required to achieve an increase in sustainable wetland acreage (Zedler 1999).

Scientific research to characterize existing hydrology, vegetation, and soils on the Conservancy site has been ongoing since 2003, and the data collected will serve as the basis for developing a Conceptual Restoration Plan. While it is obviously not possible to fully restore the Teaneck Creek watershed to some previously pristine state (Zedler and Leach 1998), our overall goal is to establish hydrologic conditions typically found in a New Jersey temperate forested riparian corridor. For the purposes of this project, we are defining "restoration" as the establishment of 20 acres of forested and scrub/shrub wetlands within the 46-acre site. Although we acknowledge that this is not the usual definition of "restoration," for the sake of simplicity we will use this term to refer to the project's objective. Specific goals for the project include protecting existing high-quality native areas, creating new wetland acreage through the removal of fill materials and the lowering of surface elevations, and reestablishing a hydrological connection between Teaneck Creek and the interior wetlands and groundwater.

Sustainable wetland ecosystems require specific combinations of water supply, topography, and soil characteristics (NRC 2001), and to determine the success of a wetland restoration or enhancement project, these interrelated attributes are typically compared to a specific wetland reference site. As we develop the restoration strategy for this site, our team is aware of the lack of success experienced by managers who have attempted to restore shrub swamp and forested wetland ecosystems both across the U.S. and in New Jersey. These two wetland types have been characterized as particularly difficult to restore (NRC 2001; Balzano et al. 2002; Minkin and Ladd 2003), in part because of the time and conditions needed to establish woody plants. The degree of difficulty encountered has been documented by the scientific community (NRC 2001), the U.S. Fish and Wildlife Service (Dahl 2000, 2005), the U.S. Army Corps of Engineers (Minkin and Ladd 2003), and the State of New Jersey (Balzano et al. 2002; ITRC 2005).

Although a gap exists in the peer-reviewed scientific literature describing successful restorations of forested riparian wetlands, reviews of regulatory permit information (Grayson et al. 1999; Dahl 2000; Sudol and Ambrose 2002; GAO 2005) and analysis of New Jersey wetland mitigation compliance (Balzano et al. 2002) verify that the success rate in restoring/creating freshwater riparian wetland systems is abysmally low. In the 2003 U.S. Army Corps of Engineers (USACE) study of overall wetland losses in New England (Minkin and Ladd 2003), forested wetlands accounted for 50% of all wetlands lost in this region (180 acres). However, the mitigation success to offset these losses totaled less than 20 acres. Field evaluation of 90 New Jersey freshwater wetland mitigation sites found only 1% of the proposed forested wetland acreage was achieved (Balzano et al. 2002).

In addition to the lack of reliable data for successful riparian wetland restoration, there is a similar lack of data for restoration of wetlands situated in highly urbanized areas. Despite some recent studies of urban wetlands (Ehrenfeld 2004, 2005; Burns et al. 2005; Wolin and Mackeigan 2005), the effects of surrounding urban land use on wetland hydrology, vegetation, and biogeochemical (Lamers et al. 2006) functions are not yet well understood. Urban wetlands differ from wetlands found in more natural settings in certain fundamental ways, including altered natural hydrology, high levels of anthropogenic site disturbance, and the frequent presence of invasive plant species (Guntenspergen and Dunn 1998; Ehrenfeld 2000). Urban wetlands may also experience continued anthropogenic disturbances after restoration work has been completed (Grayson et al. 1999; Magee and Kentula 2005).

Goals of the Teaneck Creek Wetland Restoration

Structural goals for this project include: 1) reestablishing a hydrologic connection between Teaneck Creek and the site's interior surface and ground waters; 2) the restoration of approximately 20 wetland acres to include riparian forest, scrub/shrub, and emergent water wetlands in locations where each type is sustainable under the given hydrologic regime and microtopography; and 3) within each wetland type, the establishment and survival of an appropriate native plant community. As a reference wetland to judge the project's success we will be using an on-site area where consistently saturated organic soils support diverse native vegetation. In addition to this on-site reference, we will identify a forested wetland site adjacent to the Tenakill Brook in Bergen County, New Jersey, as an off-site reference. We anticipate that achievement of the project goals will increase the residence time of Teaneck Creek water in the site's wetlands. Increased residence time will potentially increase the amount of nitrogen that these wetlands remove prior to water movement downstream into the lower estuary of the Hackensack River, where high porewater nitrogen levels have been observed in the salt marsh sediments (Ravit et al. in press).

Important factors to consider in meeting the project objectives are the current and historical alterations of the TCC wetlands and their surrounding urban hydrology, the large monospecific stands of *Phragmites australis*, the dominance of other aggressively invasive plants, and the large areas covered by the various historic fill materials. This paper will review issues related to freshwater wetland restoration, the conditions we encountered at Teaneck Creek, and the baseline characterization our team is using to develop a Conceptual Restoration Plan for the Conservancy site. Other papers in this volume discuss specific data related to the system's hydrology (Obropta et al. this volume) and vegetation (Ravit et al. this volume), and the effects of two disturbed upstream properties on the Conservancy restoration site (Bergstrom et al. this volume).

Issues in Forested Riparian Wetland Restoration/Creation

The TCC wetland degradation is historical, and so this project is not being undertaken as mitigation for wetland loss. However, today wetland fill permits allowing destruction of existing wetlands require compensatory mitigation. We use the term "restoration/creation" because much of the available data for management of forested riparian wetlands have been collected in conjunction with the U.S. Army Corps of Engineers (USACE) permitting process. Required mitigation may be achieved through restoration, creation, enhancement, and/or preservation of other wetlands, in order to compensate for the functions provided by the lost wetlands.

The greatest overall U.S. wetland losses have occurred in emergent and forested freshwater wetlands (Figure 2a), whose total acreage decreased by 6.9% in the decade prior to 1997 (Dahl 2000). Although forested wetlands accounted for up to 50% of wetland losses (Dahl 2000), the percentage of field-confirmed mitigation for these losses was only 5% (Minkin and Ladd 2003). More recent analyses (Robb 2002; GAO 2005) have found failure rates of over 70% for forested wetland restoration/creation. In a USACE study (Minkin and Ladd 2003), forested wetland impacts in New England totaled 178 acres, and the proposed "in-kind" mitigation was 25 acres. However, the actual successful forested wetland mitigation achieved was 0.5 acres. Analysis by the U.S. Department of the Interior (Dahl 2005) found increases of wetland acreage in the freshwater forested category (1998–2004) were due solely to natural succession that resulted in the movement of wetland acreage from the "shrub" to the "forested" category, with a corresponding decrease in shrub wetland acreage (Figure 2b).

Deciduous forested wetlands are the most abundant type of New Jersey wetland, equaling approximately 1/3 of the state's total wetland area (Ehrenfeld 2005). New Jersey's success rate in the mitigation of riparian and scrub/shrub wetland acreage has mirrored national trends. Field evaluation of 90 wetland mitigation sites concluded that although 41% of the mitigation projects proposed were forested freshwater, only 1% of the proposed acreage was achieved after an average of six years (Balzano et al. 2002). The reasons for the lack of success in restoring/creating shrub and riparian forested wetlands tend to fall into three broad categories: the topography, hydrology, and soils required to achieve targeted parameters.

In the Conservancy wetlands, these factors will be influenced to some degree by the stream channel itself, the adjacent upland land use (Zedler and Leach 1998), inputs from the overall catchment area (Mensing et al. 1998), and any surrounding anthropogenic disturbances, which may continue to occur post restoration (Burns et al. 2005; Wolin and Mackeigan 2005).

Hydrology

Hydrology is the dominant factor governing a wetland's type, development, maintenance, and functional attributes (Bedford 1996; NRC 2001). Hydrologic differences result from interactions between the wetland landscape and the hydrologic cycle, which in turn are driven by local climate conditions (Bedford 1996). Having a known and reliable water source is the most difficult factor to achieve when establishing wetlands (Minkin and Ladd 2003; Bedford 1996), and many wetland projects have been deemed unsuccessful because they lack suitable hydrology (Mitsch and Wilson 1996; NRC 2001; Balzano et al. 2002). As the degree of wetland degradation increases, the difficulties in restoring appropriate hydrology also increase (NRC 2001).

In New Jersey forested wetlands located in the Piedmont floodplain, a stable water table is primarily governed by the groundwater supply and source, which may be augmented by periodic over-bank flooding (Stolt et al. 2000). While the hydrology of undisturbed riparian wetlands is controlled by periodic river flooding, groundwater discharge and infiltrating, and precipitation (Wassen et al. 2003), urban wetlands typically have the additional factor of stormwater runoff inputs (Burns et al. 2005). Impervious surfaces and storm sewers accelerate the rate of stormwater movement into streams that drain into urban wetlands, where flow rates have been reported that are up to three times greater than the flows in undisturbed catchments (Burns et al. 2005). This is particularly true in densely populated locations such as Teaneck Creek, where the wetland is draining a highly developed regional catchment area of almost 300 acres (Bergstrom et al. this volume). In addition to determining flow rates, the water source will determine the nutrient and contaminant loadings entering an urban wetland.

Increases in surface water inputs can change the hydrology of an urban wetland, including the hydrograph, residence time, and temporal water variations (Bedford 1996; Zedler and Leach 1998), and urban hydrologic patterns are often quite different from the patterns found in natural wetland systems. An urban hydrologic pattern often seen is increased "flashiness": the rapid movement of water through urban storm systems into wetland stream(s), followed by a rapid elevation of stream water height, accelerated water flows through the stream, and then a rapid return to low flow water levels (Burns et al. 2005). Flashiness can also destabilize the stream channel (Sudduth and Meyer 2006), resulting in downcutting that can contribute to increased drainage of the wetland's subsurface water between storm events.

Restored/created freshwater wetlands have a tendency to exhibit greater "wetness," due to wetland engineers opting for a saturation period of 12.5% of the time. This is the upper limit of a transition zone described by Clark and Benforado (1981), whose range provided characterizations of upland versus wetland habitat; if a site is saturated less than 5% of the time it displays upland characteristics, and if saturated more than 12.5% of the time it will exhibit wetland characteristics. The USACE incorporated the 12.5% definition into their 1987 wetland delineation manual, and so wetland restorers use the conservative end of this scale, which results in wetter projects (Dahl 2005). This is especially problematic when attempting to restore forested riparian systems. If soils are too wet to support tree species, forested wetlands will not establish, and in fact wetlands that have been restored/created are often wetter than planned (NRC 2001).

Vegetation

Wetland plant communities are structured by fine-scale hydrologic conditions, and plant species cover is strongly correlated with mean water table depth, which may be altered or obscured by urban disturbances (Magee and Kentula 2005; Dwire et al. 2006). Predictors of wetland vegetation include water depth, inundation duration, and seasonal patterns of flooding, particularly with respect to woody plants, because reducing peak water flows enhances wetland succession from herbaceous to woody species (Toner and Keddy 1997). Differences of as little as 6 feet in the depth to the water table can shift inundated wet meadow plant communities to moist meadow communities, which are not inundated (Dwire et al. 2006).

While relatively little data have been collected on plant communities in forested urban wetland systems, diversity may be either quite high (Toner and Keddy 1997; Magee and Kentula 2005; Ehrenfeld 2005) or, conversely, species poor. In a set of 21 urban wetlands in northeastern New Jersey, species richness ranged from 29 to 119 species at a given site, and 15% of the species observed were exotic (Ehrenfeld 2005). Magee and Kentula (2005) observed high species richness (356 plant taxa) in urban wetlands, but more than 50% of these species were nonnative. Total vegetative cover is often lower in created versus natural wetlands, and the proportions of upland versus wetland species often differ. Structural and functional differences may result due to the wetland's age, species recruitment, and normal successional patterns (Grayson et al. 1999). Restoration success can be hampered by inappropriate actions of local property caretakers post-restoration, such as the practices of cutting wetland shrubs or regularly mowing newly created forested areas in an effort to give an advantage to woody seedlings (Minkin and Ladd 2003).

Deep flooding and long periods of ponding or standing water can decrease vegetation diversity and/or shrub densities, but conversely, these conditions may also decrease the number of invasive species able to establish (Ehrenfeld 2005; Dwire et al. 2006). In the few studies available, the majority of invasives observed were either upland or facultative upland species (Ehrenfeld 2005), suggesting that less saturated conditions may allow invasives to establish to the detriment of native wetland plant communities. Invasive species, particularly common reed (*Phragmites australis*), purple loosestrife (*Lythrum salicaria*), and multiflora rose (*Rosa multiflora*) were found to be common problems in eight restored New England wetlands (Minkin and Ladd 2003). Another problem is the introduction of cultivated varieties of native species, and the effect of these alien genotypes on wetland functions and/or other native species (Minkin and Ladd 2003). Heavy inputs of stormwater runoff can also potentially favor the dominance of invasive species (Joy Zedler personal communication). Wetland plants are affected by the amount of sedimentation and by nutrient inputs, both of which can enhance the growth of invasive species (Woo and Zedler 2002; Mahaney et al. 2004).

Soils

Undisturbed riparian wetland soils in the northeastern U.S. are often wet, acidic, and highly organic. However, soil characteristics that are important to nutrient cycling processes have been shown to be quite different in restored/created forested wetlands. In undisturbed riparian wetlands, the amount of soil organic matter is often two times higher than in constructed wetlands, and while sand may account for two thirds or more of the surface soil in restored/created systems, it is typically a negligible component of natural wetland soils (Campbell et al. 2002; Bruland and Richardson 2005). The proportion of silt and clay—often higher in natural wetlands—determines the soil particle size, which in turn determines permeability and porosity, and is inversely proportional to water holding capacity (Stolt et al. 2000). The cation exchange capacity (CEC), and levels of organic C and N have been found to be five to ten times higher in natural wetlands, and constructed wetlands typically exhibit a higher proportion of basic cations (Ca, Mg), and a higher pH than natural wetlands (Stolt et al. 2000).

Soil compaction appears to be common in wetland restoration projects, and created wetlands often exhibit a reduction of both large scale and microtopography, as well as an increase in the amount of low relief (Stolt et al. 2000). When an activity destroys fine-scale features such as microtopography (Stolt et al. 2000; Bruland and Richardson 2005), this reduction will result in a concomitant reduction of the "wetness" gradient that supports diverse plant species. The bulk density of soils in natural wetlands can range from 2-fold to an order of magnitude lower than the bulk density found in the restored/created wetlands soils, although the number of studies looking at this factor is small (Campbell et al. 2002; Bruland and Richardson 2005).

Location and Surrounding Land Use

Landscape position dictates the site hydrology and type of wetland that can be successfully restored and sustained (NRC 2001). However, degradation of the surrounding land can compromise wetland establishment and functionality, and so expectations and goals for urban freshwater wetland restorations need to be scaled to the surrounding landscape (Wolin and Mackeigan 2005). Parkyn et al. (2003) found isolated stretches of riparian buffer restoration produced few consistent improvements in water quality, habitat, or stream invertebrate communities. They suggest that "patches" of restoration may not be large enough to improve overall function of a given ecosystem, and so if upstream areas and/or tributaries remain disturbed, downstream restorations may face a continued risk. Location of compensatory wetland sites adjacent to roadways, highways, parking lots, and industrial development can alter hydrology and water quality (Guntenspergen and Dunn 1998), increasing the degree of difficulty in successfully establishing certain wetland functional targets (Minkin and Ladd 2003), and surrounding land use has been found to be a major determinant in species assemblages (Magee and Kentula 2005). Conversely, wetlands adjacent to anthropogenic disturbances may be highly functional in retention of floodwaters, nutrients, and sediments (Guntenspergen and Dunn 1998). Because a large hospital complex and a public school are directly upstream from the TCC restoration site and have permitted discharges into the creek, land use on these two parcels directly affects the water quality in the Teaneck Creek wetlands (Bergstrom et al. this volume).

Teaneck Creek Conservancy Restoration Area

The Teaneck Creek wetlands are situated adjacent to two major urban roadways (DeGraw Avenue on the southern boundary and Teaneck Road on the western boundary) at the northern terminus of the New Jersey turnpike (Interstate 95) and the eastern terminus of Interstate 80 (see Arnold and Berstrom et al. this volume for details of upstream conditions). South of the hospital, the creek flows under Teaneck Road, through the lawn of Thomas Jefferson Middle School, and under Fycke Lane, where it enters the wetland system. The stream bank on school property is in need of stabilization (Bergstrom et al. this volume) and is currently lined with the invasive plant Japanese knotweed (*Polygonum cuspidatum*), which is cut periodically by the school district and left to float downstream into the restoration site.

Site Characteristics

The topography of this system is characterized by a series of low-lying subwatersheds (Obropta et al. this volume), higher elevations due to the presence of various fill materials, a straightened creek channel with an adjacent clay fill berm that forms a levee, and, on the upland side of the berm, depressions with standing water containing monospecific stands of *Phragmites australis* (Figure 3). Teaneck Creek flows into Overpeck Creek, which is connected to the lower Hackensack River, a tidal estuarine system.

The Teaneck Creek connection with the lower estuary has been altered due to the installation of tide gates seven miles south of the site. These gates close during incoming tides, and therefore the creek does not experience a typical tidal flushing. Twice daily, when the tide gates close, the waters flowing downstream are retained in the system until the tide changes and the gates reopen, creating a backwater effect that produces a daily tidal pulse (Obropta et al. this volume). When high tides coincide with precipitation events, it is common for the creek banks in the southern portion of the site to overflow (Figure 4c). Although Teaneck Creek is only 1.5 miles in length, the hydrology in the Fycke Lane northern section is completely different from that in the DeGraw Avenue southern section. During low-intensity storms, the Fycke Lane waters rise quickly, but this section only overtops the stream banks during major storm events. When a storm ends, the Fycke Lane stream waters quickly return to their low level (Figure 4b), resulting in a very "flashy" hydrograph for this portion of the creek.

The hydrologic interface between Teaneck Creek, its tributaries, the groundwater, and the standing water depressions is unlike the connection found in a non-disturbed riparian corridor. In addition to two small tributary streams, there are six pipes that directly discharge stormwater from the Township of Teaneck into the wetland (Figure 4a). There are small groundwater seeps in some areas, but across most of the site the hydrologic connection between the groundwater and the creek has been eliminated due to the presence of underlying natural clay layers and clay fill dredge material (Obropta et al. this volume). In essence, much of the wetlands on this site appear to be functioning as perched bogs (Joan Ehrenfeld personal communication), dominated by precipitation and stormwater inputs.

The vegetation on the site (Ravit et al. this volume) is dominated by *Phragmites australis*, which is thriving in large, ponded areas that have formed in low-lying depressions. The newest invasive species to arrive in the system ca. 2005 is mile-a-minute vine (*Polygonum perfoliatum*), which now appears to be overpowering the *Phragmites* in certain sections (Figure 5a). In spite of the large areas covered by invasive monocultures, a forested wetland remains intact in the northeastern portion of the site (Steven Handel personal communication), where native wetland vegetation is thriving (Figure 5b). The hydric soils in this remnant area are continually saturated, and standing water is found here after a storm event. In spite of the site's invasive plant coverage (40% of the species observed covering approximately 40% of the site), total species diversity was found to be high (245 plant species).

A site assessment was completed for Bergen County in 1999. As part of this assessment, soil samples were collected from test pits throughout the site, and the soils were classified as Udorthents (Figure 6). No soil profile was observed in these soil borings, and the only hydric soils were located in the forested northeastern corner of the site adjacent to Fycke Lane. A cross section detailing the site soils (Figure 7) shows the presence of sand and clay fill material above the organic mat. However, patches of various substrates are scattered throughout the 46 acres, and include: 1) unconsolidated fill materials; 2) clay dredge sediments placed on the site as fill; 3) reduced organic wetland soils; and 4) in the northern portion of the stream bank, sand (Figure 8). In addition, there is a large area on the southern border of the site adjacent to DeGraw Avenue where construction debris, including asphalt and concrete, and discarded large household items have been dumped illegally (Figure 9). The wetland delineation completed in 2006 (Figure 10) shows that the majority of the site has been classified as wetland.

Baseline Site Characterization

To ensure a sustainable wetland restoration, the Interstate Technology Regulatory Council (ITRC 2005) recommends a thorough assessment of the wetlands being restored to "understand the hydrology, soil, and plants, and how they interact to affect the functions or values provided by the wetlands." This is a factor in the decision by the New Jersey Wetlands Mitigation Council (NJWMC) to fund a scientific baseline assessment prior to development of the Conceptual Restoration Plan, in the hopes that the Teaneck Creek restoration would not be another freshwater riparian wetland failure. During the three-year study, the Conservancy site has been characterized with respect to:

1. Surface water inputs, hydrologic flow rates, and nutrient loadings;
2. Groundwater depths to water table, flow rates, and nutrient loadings;
3. Presence, abundance, and location of native and invasive vegetation;
4. Soil characteristics associated with various hydrologic regimes on the site; and
5. Sediment denitrification potential pre-restoration.

These activities have been coordinated through the Rutgers Environmental Research Clinic (www.rerc.rutgers.edu). Rutgers University has also contracted with the U.S. Geological Survey (USGS) to train students in accepted hydrologic sampling techniques. Stormwater samples have been collected quarterly over the last two years and analyzed in the USGS laboratories (Obropta et al. this volume). Shallow groundwater monitoring wells, reaching a depth of 40 cm below the soil surface, were installed at 30 locations within the TCC site (Figure 10). To install these wells, a soil core was excavated using a hand auger, and a PVC shallow groundwater well containing screened holes to allow water movement into the well was placed in the hole. The excavated area remaining around the well was filled with sand; the sand and the adjacent soil surface were then capped with bentonite clay to preclude movement of water into the well from the surface, and the well was capped. Hydrology data has now been collected for over a year at each well by measuring depths of inundation and depths to groundwater on a weekly basis.

Analysis of soil samples was conducted to determine moisture content, nutrients, conductivity, pH, and micronutrients. These samples were collected at the locations where groundwater wells were installed, prior to placement of the wells (Figure 10). Samples were obtained using a corer 25 cm in length and 10 cm in diameter. Results of these analyses indicate a high degree of heterogeneity related to the hydrology and the amount and type of fill material present at each sampling location. Soil organic carbon proportions ranged from 2% to 22%, TKN values ranged from 0.08% to 0.57%, and ammonia concentrations ranged from 0.1 to 11.0 ppm. Soil pH varied from 6 to 7.85, and we hypothesize that the high end of this range is due to the decomposition of concrete debris. The soil categories range from clay to sandy loam. In addition to the clay fill material forming the creek bank berms, there are natural clay layers and lenses under most of the site at depths varying from 1 to 4 feet.

A vegetation analysis was also completed (Ravit et al. this volume). The site was organized into a series of 32 grids, and each grid was surveyed to determine the presence or absence and the abundance of both native and invasive vegetation. Plant Stewardship (PSI) and Floristic Indices (FI) were subsequently calculated for each grid. Data related to the plant species, depths to the water table, and soil properties are now being analyzed to determine which native plants might be sustainable in the different subwatersheds of the site, given the various combinations of hydrology and soils.

Because nitrogen (N) leaving this system can contribute to high rates of eutrophication in the lower Hackensack River estuary, we chose to target a decrease in N transport out of the system as a functional restoration goal. Denitrification rates are now being analyzed in soil samples taken from multiple locations on the site (Figure 10), and these rates will be used to calculate changes in denitrification potential of the Conservancy wetlands pre- and post-restoration. We also undertook a study to characterize the contribution of atmospheric deposition of carbon and nitrogen loadings to the site (Ravit et al. 2006). Samples were collected quarterly during 2005–2006 and analyzed for wet and dry deposition of organic and inorganic (nitrate, ammonia) N compounds. Wet deposition of inorganic N was ten times greater than dry deposition, and the range of nutrient concentrations measured was similar to the regional signals found for the New York–New Jersey region by Lovett et al. (2000), Meyers et al. (2001), and Seitzinger et al. (2005). When dry particle N deposition was compared in samples taken at various distances from the DeGraw Avenue roadway, inorganic N concentrations found at the roadside were 20–50% higher than 100 meters away from the road.

To achieve our goal of 20 acres of rehabilitated wetlands, there must be an increase in flooding and a subsequent retention of water by additional acreage within the TCC site. To achieve increased wetland acreage, changes must occur in the topography of the system, and these changes will take into account removal of debris, the water flow patterns of the six stormwater inputs, the inter subbasin water movements, and surface flows from the Teaneck Creek. The results of the baseline studies are being incorporated into a Conceptual Restoration Plan for the TCC site, which will include detailed grading plans, planting plans, and invasive species control plans. A secondary long-term project goal is to develop an Urban Wetland Model capable of describing the relationship between hydrology, vegetation, and soil denitrification within this urban wetland system.

Acknowledgments

Funding support for the Teaneck Creek studies has been provided by the New Jersey Wetlands Mitigation Council and the New Jersey Water Resources Research Institute. Maps are courtesy of Sean Walsh and Michael Mak (Rutgers University) and Jeremiah Bergstrom (TRC Omni Environmental). We gratefully acknowledge the unwavering support by the Teaneck Creek Conservancy staff for all the project scientists, students, and volunteers.

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Figure 1: Arial photograph of Teaneck Creek Conservancy wetlands and surrounding urban land use. (Photo courtesy of Bergen County Parks Department.)

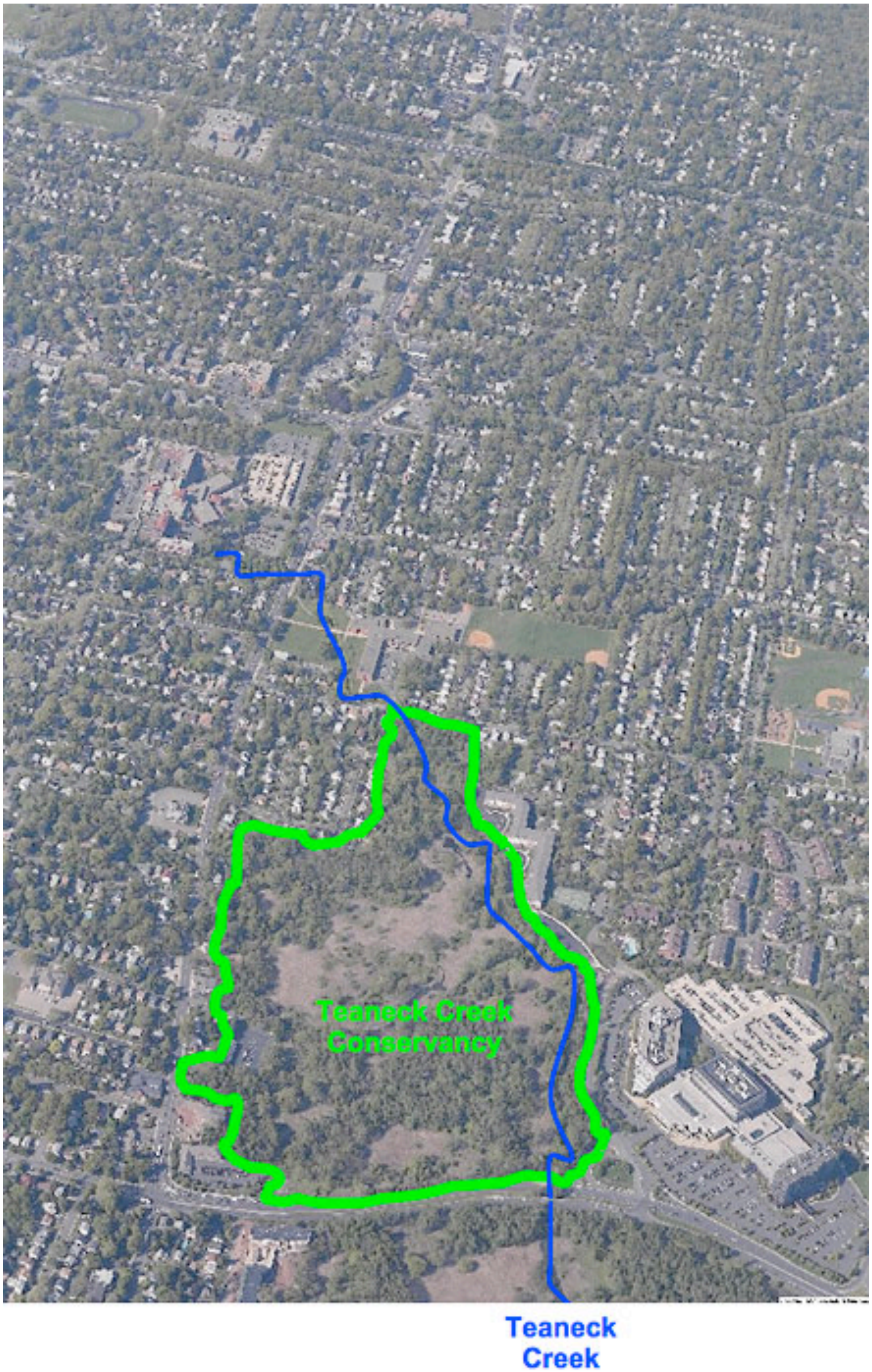


Figure 2: U.S. freshwater forested and scrub/shrub wetland acreage a) from 1950 through 2004; and b) in 1998 and 2004. Data reproduced from Dahl (2005).

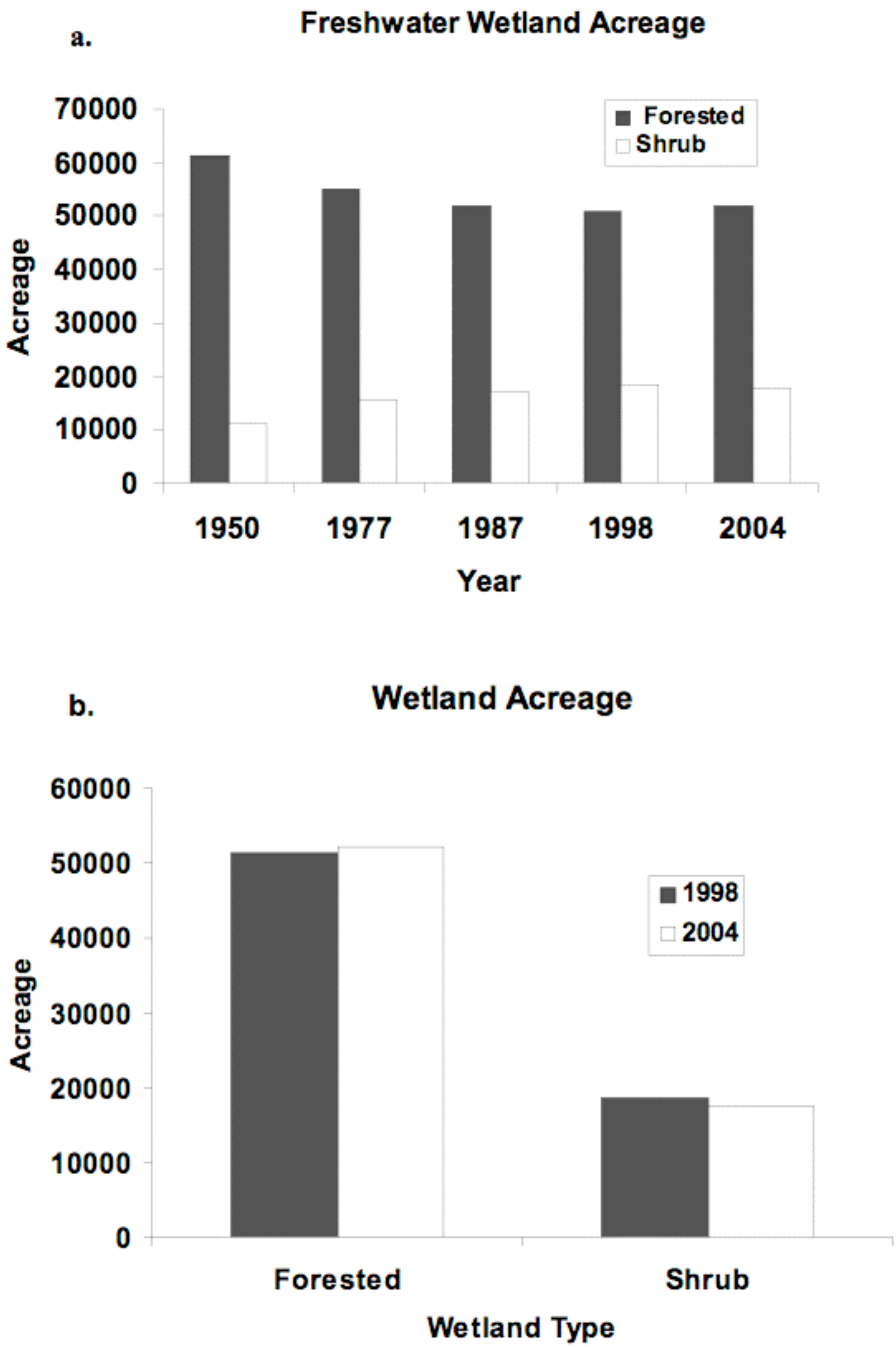


Figure 3: *Phragmites australis* monocultures in Teaneck Creek ponded depression areas.

Figure 4a: Six storm drains empty urban runoff directly into the Teaneck Creek.

Figure 4b: During a storm event, the northern portion of Teaneck Creek adjacent to Fycke Lane exhibits "flashy" hydrology. Photos 4b and 4c were taken within 10 minutes of each other following an intense storm on September 29, 2005.

Figure 4c: During a storm event, the southern stretch experiences bank overflows. Photos 4b and 4c were taken within 10 minutes of each other following an intense storm on September 29, 2005.

Figure 5a: Examples of Teaneck Creek vegetation: common reed (*Phragmites australis*) overgrown with porcelain berry (*Ampelopsis brevipedunculata*) and mile-a-minute vine (*Polygonum perfoliatum*).

Figure 5b: Examples of Teaneck Creek vegetation: native forested wetland vegetation.

Figure 6: Teaneck Creek Conservancy site map showing 1999 soil test pit soil categorizations. (Note cross section I-I'.)

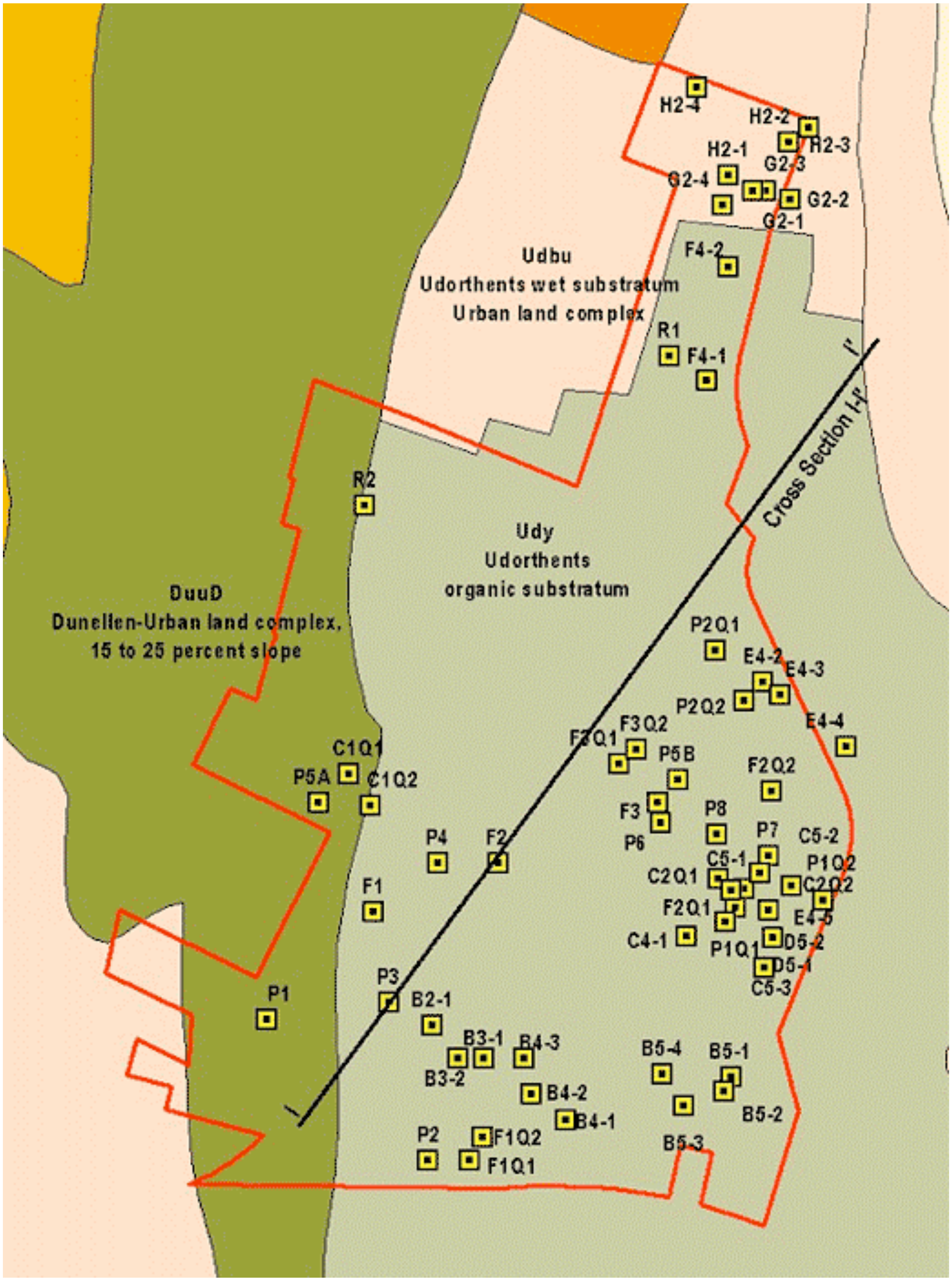


Figure 7: Teaneck Creek Conservancy 1999 soil test pit cross section showing substrate materials.

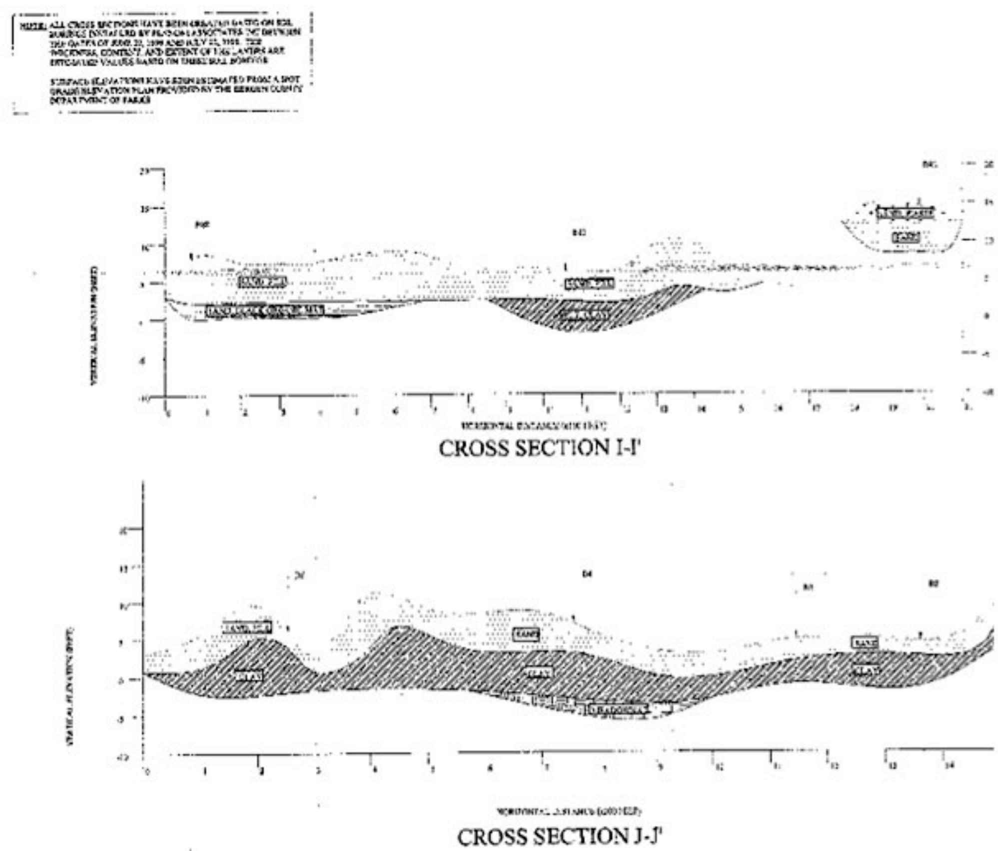


Figure 8: Soil cores obtained from: a) undisturbed location with native wetland vegetation and hydric soils; b) and c) from an area vegetated with monospecific stands of common reed (*Phragmites australis*).

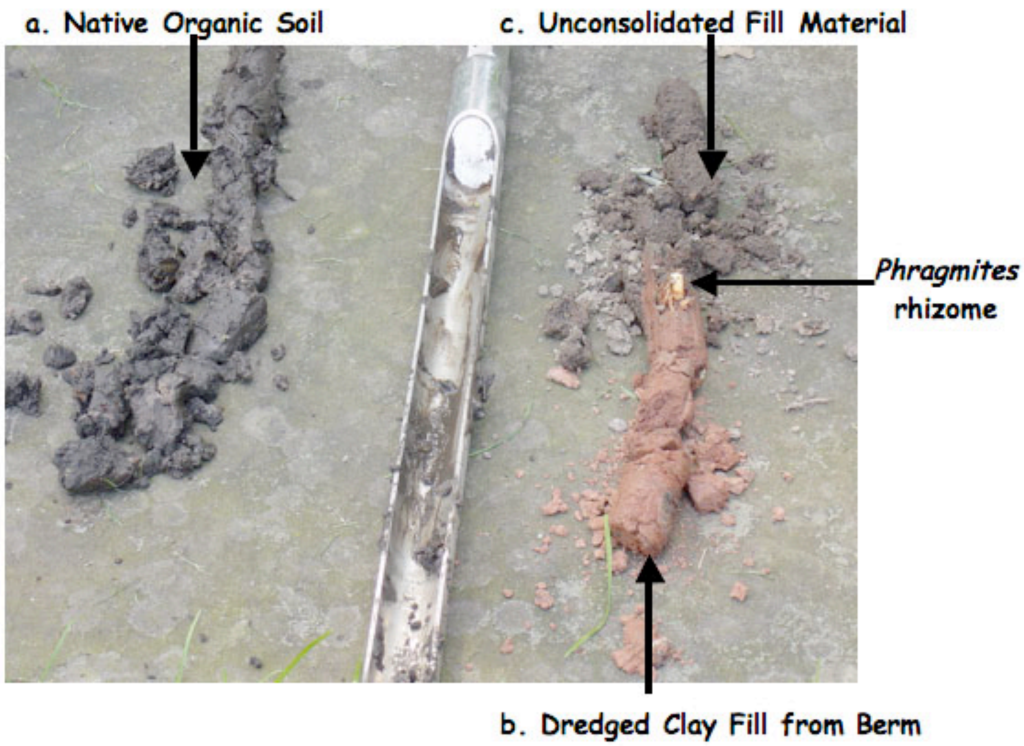


Figure 9: Discarded refrigerator debris serves as "natural" planter for garlic mustard (*Alliaria petiolata*) on Conservancy site.

Figure 10: Teaneck Creek Conservancy site map showing the wetland delineation completed in 2006. Circles indicate location of shallow groundwater wells and soil sampling locations.

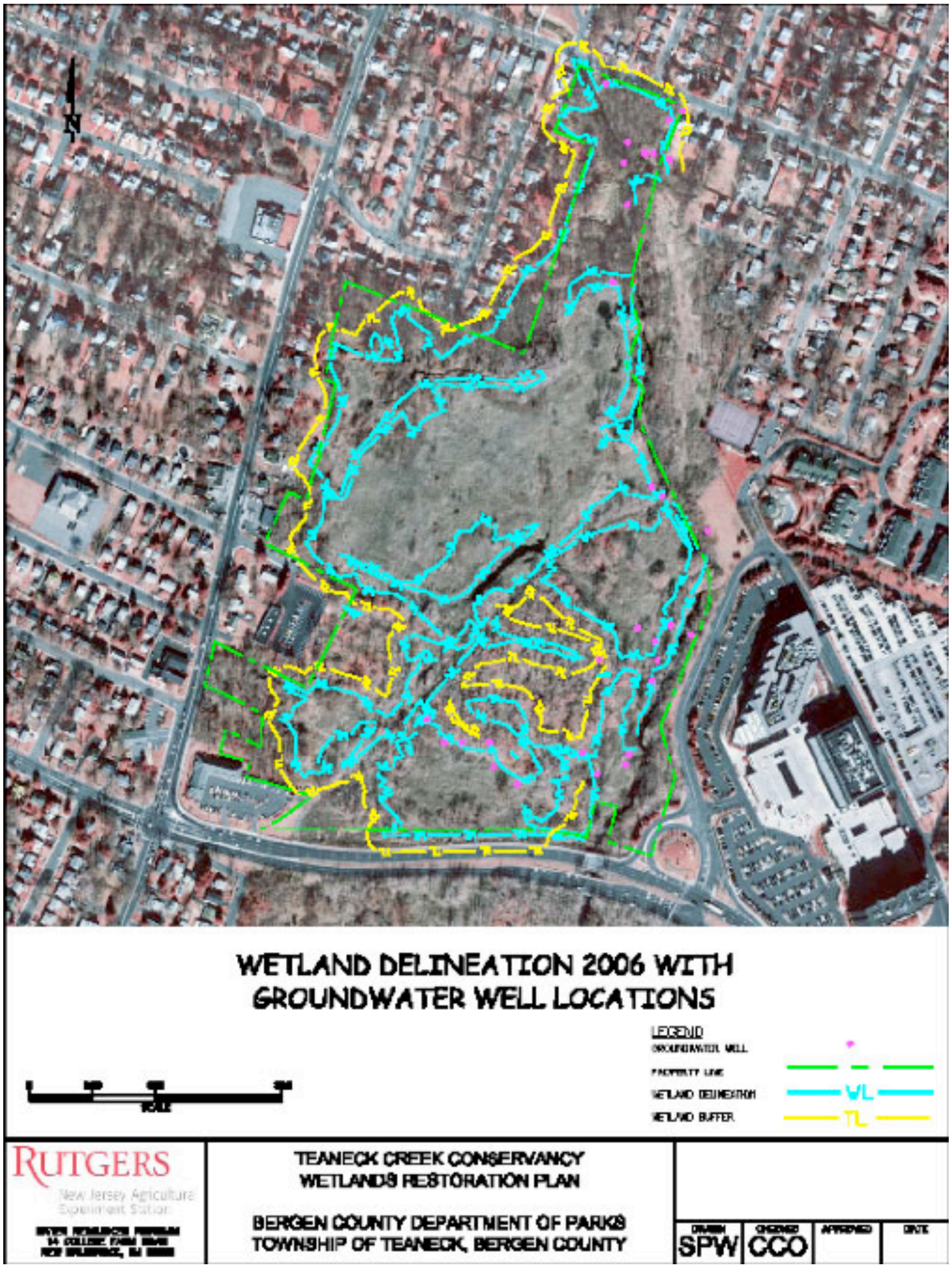
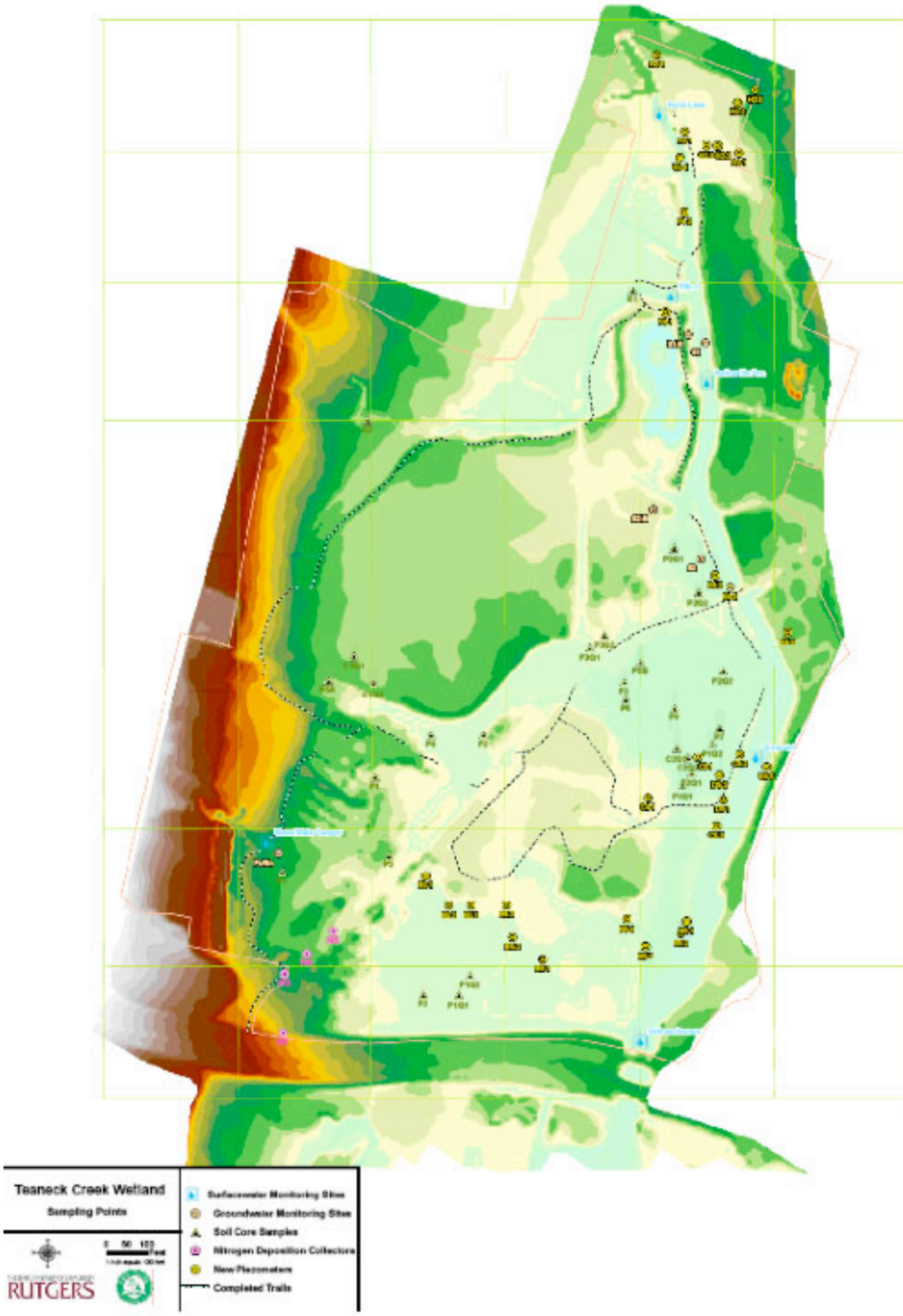


Figure 11: Map of all sampling locations.



A Historical Perspective on the Urban Wetlands of the Teaneck Creek Conservancy

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Abstract

The wetlands of northeastern New Jersey, formed after the Wisconsin glacier retreated approximately 10,000 years ago, have undergone extensive damage by humans since their formation. Projects undertaken to support the increasing urbanization of Bergen County, including interstate highway construction during the 20th century, caused particularly severe damage to the wetlands, and today, remaining Teaneck Creek wetlands are situated in a watershed whose land use is 95% urban. The 46-acre Teaneck Creek Conservancy site, owned by Bergen County, New Jersey, as an area within Overpeck Park, is managed by the Teaneck Creek Conservancy. Today, these wetlands serve as a stormwater retention basin for the Teaneck Creek watershed. The Conservancy aims to protect the site's least-disturbed wetlands and to reestablish 20 acres of forested riparian wetlands by reconnecting Teaneck Creek with the site's interior surface and ground waters, removing fill materials, eliminating and controlling invasive plant species, and planting native vegetation. The project is an interdisciplinary collaboration between the Teaneck Creek Conservancy, Bergen County, and Rutgers University.

Key words: urban restoration, urban wetland, urbanization, urban habitat, urban ecosystem, Bergen County

Introduction

During the 19th and 20th centuries, industry, transportation, population growth, and various cultural lifestyles in northeastern New Jersey set in motion accelerating changes to local ecosystems. Historically, the lands and waters of Teaneck Creek, in what is now Teaneck, New Jersey, have been utilized in multiple ways—several hundred years ago as Lenape Indian homeland and Colonial farmland, in the twentieth century as a site for transportation development and dumping (Taylor 1977), and today as public parkland. The Teaneck Creek Conservancy wetland baseline characterization highlighted in this volume will support the reestablishment of 20 acres of riparian forested wetlands on this 46-acre urban site. The project's approach to wetlands restoration is based on local ecology, interdisciplinary wetlands science, and an iterative approach to site investigation, research, and restoration planning. This approach acknowledges the need for in-depth understanding of the anthropogenically influenced site conditions. The project also recognizes the need for participation by local communities and government agencies in the reclamation, enhancement, and protection of urban natural resources.

The Conservancy was started after surveyors' flags were observed on the property and community leaders in Teaneck, New Jersey, decided to protect the land from development. When they discovered that Bergen County owned the land, these community advocates conceived a plan to transform the tract into a publicly accessible park. The nonprofit Conservancy was incorporated in 2001, and over an 18-month period it led a series of 17 community meetings to engage community residents and elected officials in its cause. The overall goal of the project that the Conservancy proposed to Bergen County was creation of a park that would use art and landscape design to integrate the natural, historical, and cultural history of the area. Two specific project goals were restoration and enhancement of the site's degraded wetland areas and reintroduction of native plant species (TCC 2001). In 2002, the Conservancy negotiated a long-term lease with Bergen County to develop and manage the 46-acre site for the purposes of public recreation, outdoor education, cultural programming, and enhancement of existing natural resources (TCC 2002). In 2004, the Bergen County Freeholders passed a resolution authorizing a memorandum of understanding among the County, the Conservancy, and Rutgers University to support the study, restoration, and protection of the Conservancy wetlands (MOU 2005).

Historical Perspective

The story of the decline and reclamation of Teaneck Creek's wetlands begins in the 1600s, when Lenape Indian leader Sachem Oratam deeded more than 2,000 acres to Dutch colonist Sarah Kiersted. The English Governor Philip Carteret granted a "patent" to Sarah Kiersted, a deed confirming her ownership of land that included the current Conservancy site. At the time Oratam deeded the property to Kiersted, a diverse ecosystem existed there with a wealth of water resources, including, according to the words of the English deed, "woods, pastures, fields, Meadows, Pools, Ponds, Islands, Creeks, Marshes, River." The water-rich ecosystem contained tributaries to the Hackensack River and provided habitat that the deed describes as conducive to "Hawking, Hunting fowling, fishing." These natural resources provided food for the Colonists and are characterized in the deed as part of the "Gaines and Proffits" of Oratam's gift to Kiersted. By the time the Conservancy was incorporated, 335 years after Oratam gave away this land, the property had been a dump site and degraded wetland for half a century, surrounded by "keep out" signs.

Teaneck Creek's wetlands declined during the late 19th- and 20th-century periods of industrialization, urbanization, and the resultant draining and filling of marshlands. *Draining for Profit and Draining for Health* (Waring 1879) sums up the then common perception of wetlands as wasteland. Waring described the New Jersey Meadowlands, of which the wetlands of Teaneck Creek are a historic remnant, as "pest" lands that offered huge financial potential if drained for development. "A single tract, over 20,000 acres in extent, the center of which is not seven miles from the heart of New York City, skirts the Hackensack River, serving as a barrier to intercourse between the town and the country…constituting a nuisance and an eyesore…. Virgin lands, replete with every element of fertility, capable of producing enough food for the support of millions of human being…all allowed to remain worse than useless…. The inherent wealth of the land is locked up, and all of its bad effects are produced, by the water with which it is constantly soaked or overflowed." Bergen County's Preliminary Assessment Report (PAR 2006) to the New Jersey Department of Environmental Protection Division of Responsible Party Site Remediation describes the site's previous land uses. The Sanborn Map (1926 through 1957) shows that parts of the site were used by a laundry, a construction company, a dance hall, and residences, among other uses. From 1899 until 1938, a trolley line ran through the site.

In the early 1950s, Bergen County developed a plan for the wetlands of Teaneck and Overpeck Creeks which proposed filling the wetlands with municipal waste and clean dredge, and then redeveloping the area as a 1,000-acre park. The Township of Teaneck transferred property for the creation of the public park and recreation area (Deed 1951). Overpeck Creek was widened and deepened through dredging, and tidal gates were constructed in the vicinity of the New Jersey Turnpike Overpass. Land elevations surrounding the creeks were raised above the water level by placement of sanitary waste and material dredged from Overpeck Creek. These fill activities resulted in the berming of Teaneck Creek, and the downstream tidegate caused the creek to be cut off from the tidal flow of the Hackensack River.

When the Conservancy acreage was originally transferred to Bergen County, the environmental issue concerning the Township of Teaneck was that they be allowed to continue to use the property as a waste dump and stormwater sump. The transfer deed includes the following provisions: (a) the right to use any or all of the premises for the disposal of garbage, ashes, refuse, and fill through any agency, contractor, or licensee engaged to remove such material for the Township of Teaneck now and in the future; (b) the right to continue the operation, maintenance, and enlargement of all existing disposal plants, sewage pumping stations, sanitary and storm drains, and rights of way, and to increase or provide new facilities as it may deem advisable; (c) the same rights and privileges were granted to the Bergen County Sewer Authorities (Deed 1951). After the land transfer, the 46 acres became known as "Area 1" of Overpeck Park. Although the wetlands of Teaneck Creek were not used for disposal of municipal waste, Area 1 experienced further degradation from dumping and filling by both private companies and the N.J. Department of Transportation, which used the site in the 1960s as a staging and disposal area for dredge and construction debris materials while building the New Jersey Turnpike and Interstate 80 (Figure 1).

Natural and manmade fill materials were deposited mainly on the southern and eastern sides of the Conservancy site (PAR 2006). Materials deposited on the site in the 1960s consisted primarily of domestic waste, including cans, bottles, clothing, and plastic, while the surficial debris included brick, glass, concrete, roofing materials, lumber, automotive parts, and appliances. An existing Area of Concern (PAR 2006) is groundwater contamination from benzene, tert-butyl alcohol, and methyl-tertiary-butyl-ether (MTBE) that leaked from underground gas station storage tanks on adjacent properties located along the western edge of the Conservancy site. The NJDEP has ongoing oversight of the groundwater contamination, which is currently being remediated.

Since the Conservancy began in 2001, volunteers have removed multiple Dumpster loads of debris, including automotive parts, construction materials, and discarded household appliances, from the site using only hand tools. The remaining fill is too heavy to lift or is located in inaccessible areas. Some concrete debris has been recycled into artworks by artist Ariane Burgess, who led volunteers in the creation of a Peace Labyrinth built from blocks of concrete found on site (Figure 2), and artist Lynne Hull, who created concrete "Migration Mileposts" (Figure 3) that celebrate the birds (documented by DNA or radio telemetry) that pass through the site when migrating along the Atlantic Flyway. Teaneck artist Richard K. Mills's "Concrete Jungle" lies among other huge slabs of concrete roadway next to Dragonfly Pond, an open water area (Figure 4).

Teaneck Creek Conservancy Today

Today the Conservancy's 46 acres are virtually the only undeveloped land in the Teaneck Creek watershed. In this highly urbanized area, the creek is a component of the receiving waters for the Township of Teaneck municipal stormwater system, serving as the discharge point for six stormwater outfalls that drain directly into Teaneck Creek and its wetlands. The channelization, downcutting, and berming of the creek and the clay dredge and debris that fill and compact the site all impede groundwater and creek connectivity. Today, precipitation and stormwater are the primary hydrologic inputs for the site's wetlands. In addition to the stormwater inputs, we discovered that a local hospital is permitted to pump 100,000 gallons of groundwater per day into Teaneck Creek in order to keep the hospital basement dry (W. Kinder personal communication). The banks of Teaneck Creek are cut and filled, eroded, and sometimes blown out during storms. The resulting siltation, as well as nonpoint source and thermal pollution that result from drainage from the hospital's sterilization facility, degrade the water quality in the creek.

Benthic studies (Serra 2001) revealed "poor" water quality, which supported only pollution-tolerant taxa identified as aquatic worms, black fly larvae, midge larvae, pouch, and other snails. Nonetheless, with its unique freshwater wetland system, Teaneck Creek does support other aquatic life, including killifish (*Fundulus* spp.), green frogs (*Rana clamitans*), bullfrogs (*Rana catesbeiana*), snapping turtles (*Chelydra serpentina*), and eastern box turtles (*Terrapene carolina carolina*), which have all been observed on site. Wildlife observation is a popular activity at the Conservancy. Bird species observed by scientists, volunteers, and visitors between 2003 and 2007 (Table 1) include Great Blue Heron (*Ardea herodias*), Green Heron (*Butorides virescens*), and Great Egret (*Ardea alba*). Mammal species observed include red fox (*Vulpes vulpes*) and white-tailed deer (*Odocoileus virginianus*). Rutgers scientists and engineers anticipate that stormwater retrofits and planned wetland and creek enhancements will improve water quality, biodiversity, and wildlife habitat at the site. Natural restorative processes are also playing a positive role in the Conservancy's rehabilitation. Native trees (e.g., *Acer rubrum*), shrubs (e.g., *Viburnum dentatum*), and herbaceous plants spontaneously grew from the seed bank when the burden of debris and invasive vegetation was removed from the wetlands by volunteers and by the contractor constructing our trail system.

During a field visit in 2004, Dr. Steven Handel noted an area on the northeastern edge of the site containing seeps, vernal pools, and native plant species that were characteristic of New Jersey riparian corridors of 350 years ago. Even in more degraded areas of the property, Dr. Handel observed remnants of native vegetation that would have been familiar to the Dutch colonists. Dr. Handel advised that these wetland fragments containing native vegetation be guarded from invasive plants through selective weeding, and that the northeastern section become the reference for restoration of the site. A specific example of the influence of scientists on all aspects of the project occurred when community leaders planned to build the Puffin Outdoor Classroom within the wetland remnant Dr. Handel suggested be the reference for the Conservancy's restoration. The original concept was to locate a classroom in this area to enable schoolchildren to appreciate the natural beauty and diversity of riparian wetlands. The non-scientists ultimately were persuaded that they were going to destroy what they treasured—the highest quality natural resources on the property. We redesigned the classroom and relocated it to an adjacent area that already was highly disturbed and covered by *Phragmites*. Invasive vegetation was removed, native wetland vegetation and seeds were planted, and additional native vegetation grew from the seed bank. The classroom that opened to the public in May 2006 overlooks the native vegetation of the remnant wetland (Figure 5). The restored wetland acreage in the outdoor classroom has met one of the wetland restoration and enhancement project deliverables funded through the N.J. Wetlands Mitigation Council grant supporting this project.

As degraded as this place was when the Conservancy was founded, as difficult as the work has been at many points, citizens have reclaimed and, with the help of scientists, are transforming the lands and waters of Teaneck Creek from a dump in to a unique wetland restoration and a renewed civic space (Figure 6).

Acknowledgments

We gratefully acknowledge the scientific support provided by faculty, staff, and students of Rutgers University, U.S. Geological Servey, and TRC Omni Environmental. Funding support for the Teaneck Creek Conservancy has been provided by the Puffin Foundation and the Rosenstein family. Funding for the site characterization and restoration/enhancement of 20 wetland acres has been provided by the New Jersey Wetland Mitigation Council. We thank Dr. Marion McClary, P. Warny, and B. Ballengee for identification of benthic invertebrate species and Dr. Peter Kallin and Richard Engsberg for identification of avian species.

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Table 1: Birds sighted within the Teaneck Creek Conservancy boundaries (2003–2007).

Common Name	**Scientific Name**	**Type**
American Robin	*Turdus migratorius*	Migrant
American Woodcock	*Scolopax minor*	Resident
Black-and-white Warbler	*Mniotilta varia*	Migrant
Black-capped Chickadee	*Poecile atricapilla*	Resident
Black-throated Blue Warbler	*Dendroica caerulescens*	Migrant
Blue Jay	*Cyanocitta cristata*	Resident
Blue-headed Vireo	*Vireo solitarius*	Migrant
Cape May Warbler	*Dendroica tigrina*	Migrant
Connecticut Warbler	*Oporornis agilis*	Migrant
Dark-eyed Junco	*Junco hyemalis*	Migrant
Downy Woodpecker	*Picoides pubescens*	Resident
Eastern Phoebe	*Sayornis phoebe*	Migrant
Eastern Towhee	*Pipilo erythrophthalmus*	Migrant
Gray Catbird	*Dumetella carolinensis*	Migrant
Great Blue Heron	*Ardea herodias*	Resident
Great Egret	*Ardea alba*	Migrant
Green Heron	*Butorides virescens*	Resident
Hairy Woodpecker	*Picoides villosus*	Migrant
Magnolia Warbler	*Dendroica magnolia*	Migrant
Mourning Dove	*Zenaida macroura*	Resident
Northern Cardinal	*Cardinalis cardinalis*	Resident
Northern Parula	*Parula americana*	Migrant
Palm Warbler	*Dendroica palmarum*	Migrant
Red-bellied Woodpecker	*Melanerpes carolinus*	Resident
Red-tailed Hawk	*Buteo jamaicensis*	Resident
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	Migrant
Ruby-crowned Kinglet	*Regulus calendula*	Migrant
Sharp-shinned Hawk	*Accipiter striatus*	Resident
Swamp Sparrow	*Melospiza georgiana*	Migrant
White-throated Sparrow	*Zonotrichia albicollis*	Migrant
Yellow-bellied Sapsucker	*Sphyrapicus varius*	Migrant
Yellow-rumped Warbler	*Dendroica coronata*	Migrant

Figure 1: Increasing urbanization of the land surrounding Teaneck Creek Conservancy wetlands 1954 to 1995 (photo courtesy of Richard K. Mills).

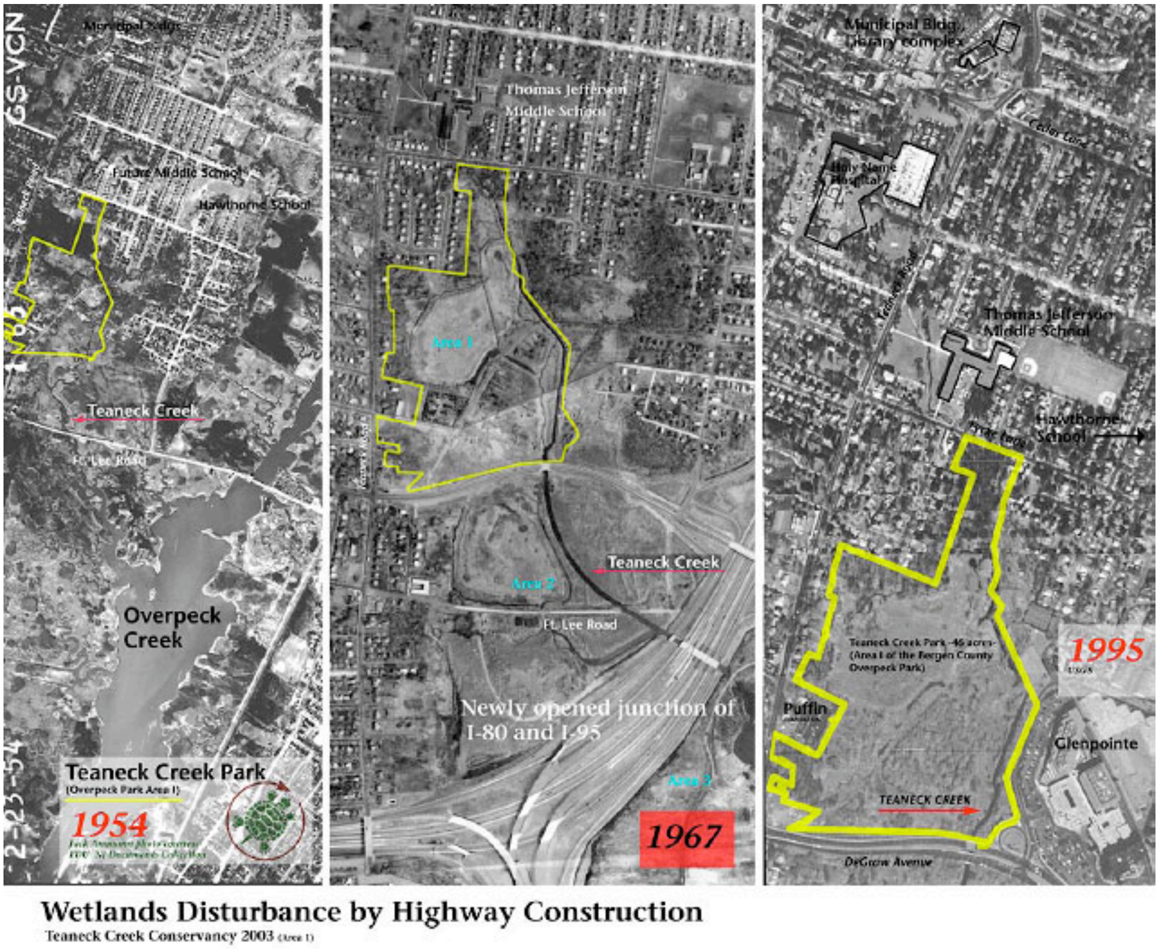

Figure 2: The Turtle Peace Labyrinth was constructed from concrete debris found on-site at the Teaneck Creek Conservancy. To create the labyrinth pathway, volunteers relocated the pieces of rubble by hand from their original location on the banks of the creek to the site of a former ball field (photo courtesy of Richard K. Mills).

Figure 3: An engraved and painted concrete rubble Migration Milepost depicting species observed on the Teaneck Creek Conservancy site using the Atlantic Flyway as their migratory route.

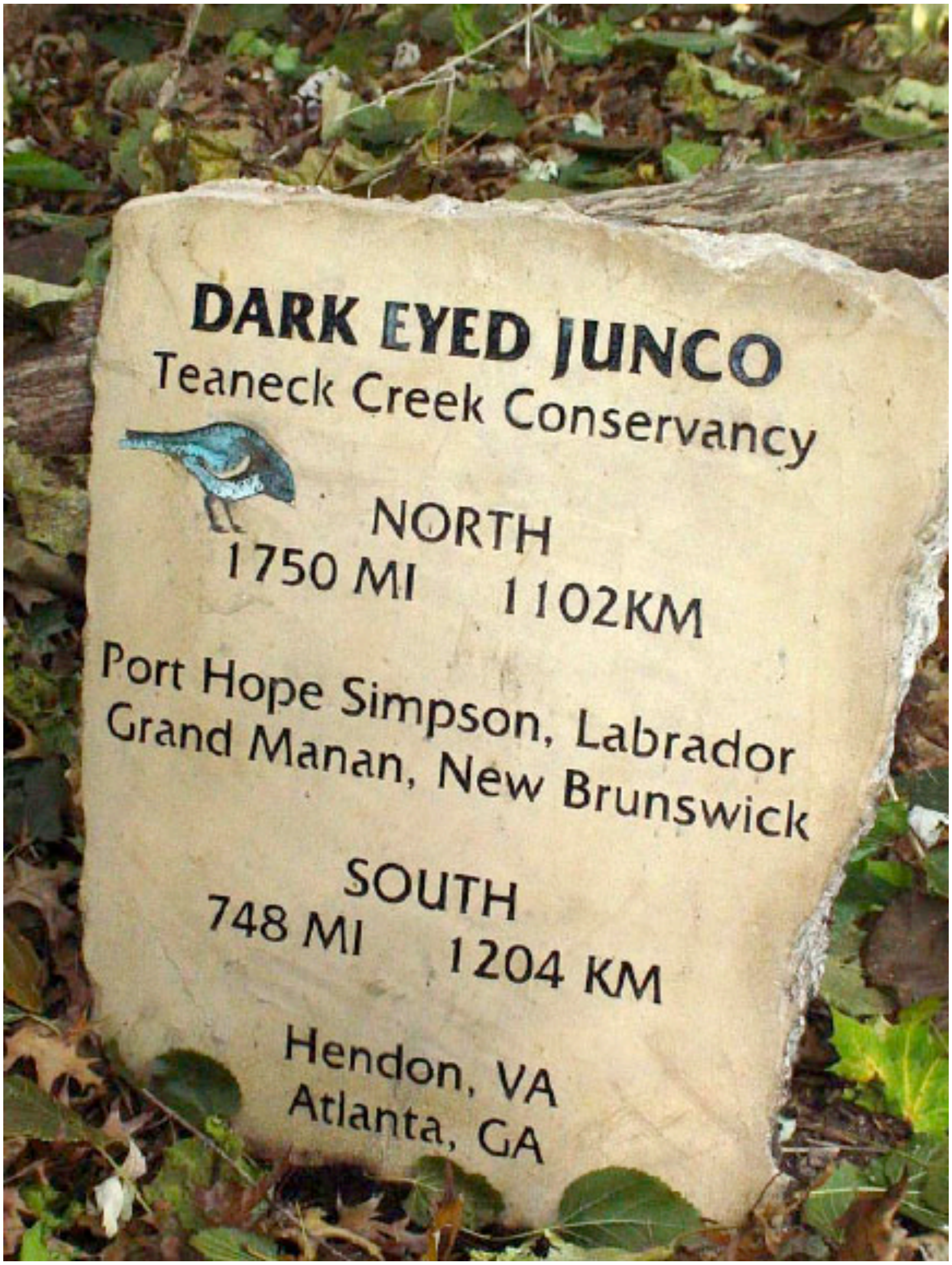

Figure 4: Concrete Jungle sculpture garden created by artist-in-residence Richard K. Mills from Teaneck Creek Conservancy debris.

Figure 5: The Puffin Outdoor Classroom located in an area formerly degraded and dominated by *Phragmites australis*. After the *Phragmites* were removed, native tree, shrub, and herbaceous species were planted by local volunteers, project scientists, and engineers.

Figure 6: Local schoolchildren visit Teaneck Creek and walk through the Conservancy site on the newly built trail system, July 2006.

Implementing Restoration Projects Upstream from the Teaneck Creek Conservancy

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Abstract

During initial investigations, scientists identified several off-site situations that were negatively affecting the health and diversity of the wetland and riparian habitats in the Teaneck Creek Conservancy restoration site. Significant off-site influences include high nitrogen inputs and non-point source pollution generated by a local hospital and the extensive presence of invasive species, chiefly Japanese knotweed (*Polygonum cuspidatum*), along the upstream banks of Teaneck Creek. This upstream source of high nitrogen loadings and the seeds of invasive species continue to threaten efforts to achieve a successful and sustainable long-term wetlands restoration on the Teaneck Creek Conservancy site. To address the nitrogen inputs, the restoration team has partnered with Holy Name Hospital, situated at the headwaters of Teaneck Creek, to develop a stormwater runoff management program. To address the downstream spread of invasive species, a partnership was formed with the Teaneck Board of Education to manage invasive species adjacent to the northern entrance to the Teaneck Creek Conservancy site. Working with the restoration project partners, Holy Name Hospital and the Teaneck Board of Education have developed plans to address stormwater runoff and erosion impacts, implement an invasive species management program in partnership with the U.S. Fish and Wildlife Service to control the Japanese knotweed, and reestablish a native riparian vegetative buffer along the entire length of Teaneck Creek upstream of the Conservancy property. Such alliances formed to deal with upstream factors illustrate the type of approach required to develop successful and sustainable long-term ecological restorations in urban areas.

Key words: stormwater, restoration, stabilization, streams, wetlands, rain garden, invasive species

Introduction

A key obstacle to achieving successful and sustainable urban wetland restoration is the influence of off-site environmental conditions (Ravit et al. this volume). During the wetlands research and site investigations at the Teaneck Creek Conservancy, partners identified several off-site locations that were negatively impacting the health and diversity of the wetland and riparian habitats in the restoration site.

Teaneck Creek Headwaters

The headwaters of Teaneck Creek are located on property owned by Holy Name Hospital, which is immediately upstream of the Thomas Jefferson

Middle School and the Conservancy restoration site. The hospital holds an NJDEP permit allowing the discharge of 100,000 gal day^{-1} of groundwater, which is pumped from the hospital basement into Teaneck Creek (Arnold this volume). This high quality discharge originates in a small garden and tumbles through a series of rocky pools toward Teaneck Road, approximately 300 feet south of the hospital. Two additional pipes from the hospital property discharge into Teaneck Creek before it reaches a culvert located under Teaneck Road and leading to the Thomas Jefferson Middle School property. The first pipe drains heated water from an on-site sterilization facility, and the second pipe contains stormwater runoff from the hospital parking deck and parking lots. There is a visible change in water quality in the stream where this nonpoint source (NPS) pollution enters the creek. In addition to these two pollution sources, runoff from slightly over an acre of parking lots flows southward, where it runs along a curb to a catch basin in the southwest corner of the parking lot (Figure 1). At this catch basin, the runoff enters a pipe and is immediately discharged into Teaneck Creek prior to the creek entering the Teaneck Road culvert. This water carries whatever pollutants (suspended solids, oil, grease, metals) are washed from the parking lot's asphalt surface. The site currently routes parking area runoff directly into Teaneck Creek. During winter months, the hospital uses urea to deice its parking structure, causing high loadings of ammonia to flow directly into Teaneck Creek.

Teaneck Creek Existing Conditions

The headwater flows originating at Holy Name Hospital combine in the storm sewer system (a total watershed drainage area of almost 300 acres) and then discharge through a 7-1/2-by-5-foot, elliptical concrete pipe onto the property of the Thomas Jefferson Middle School. Teaneck Creek flows through the school property for approximately 900 feet before entering a culvert located underneath Fycke Lane, which discharges into the northern entrance of the Conservancy. The upstream section of the stream consists of an open channel with extensive eroding bank areas (Figure 4). High velocity discharges from the culvert outfall pipe into the stream have undermined the stream banks and caused a portion of the side bank to collapse into the streambed, causing serious safety and liability concerns. The downstream section of the stream on school property has sloping banks, which are shallow enough to minimize erosion. However, as the stream abruptly turns near Fycke Lane before discharging into the Conservancy, its slope changes and the velocity along the bank increases, causing erosion near the twin box culverts exiting the school property (Figure 5). Due to the extensive presence of the invasive Japanese knotweed (*Polygonum cuspidatum*) plant, the stream bank areas and soils have become highly erodible. During heavy rainfall events, the Japanese knotweed is continually spread downstream from the school site into the forested wetlands and stream corridors of the Conservancy.

Restoration Approach

Teaneck Creek Invasive Control

One of the most significant off-site influences on this site is extensive invasive species colonization, dominated by Japanese knotweed, along Teaneck Creek at the Thomas Jefferson Middle School property immediately upstream from the restoration site (Figure 6). This upstream source of seed and stems continues to threaten the project's efforts to manage Japanese knotweed and other invasive species in the riparian and wetlands areas along Teaneck Creek. Once this source of invasive vegetation was identified, team members approached the Teaneck Board of Education and formed a partnership to repair Teaneck Creek as it flows through school property.

Stormwater Management

Holy Name staff contacted the Conservancy to identify steps the hospital could take to help with the wetlands restoration project. After discussions with the wetlands restoration scientists, it was determined that the most significant contribution the hospital could make would be to construct a rain garden on their property to treat runoff from the parking lot. The Rutgers Water Resources Program engineered an appropriate rain garden design to address the hospital parking area drainage patterns (Figures 2, 3).

The proposed rain garden design routes runoff from the parking lot through a set of curb cuts into a series of bioretention cells, which are incorporated into the landscaping between the hospital parking lot and Teaneck Road. The cells are designed to hold and infiltrate the NJDEP-designated, 1.25-inch Water Quality Storm. The cells are connected by grass swales that allow excess runoff from larger storms to be routed to the existing catch basin, bypassing the bioretention cells, thus minimizing erosion and damage to the rain garden system. This approach to stormwater management is designed to provide treatment for the runoff from approximately 90% of all precipitation events, significantly reducing suspended solids, oil, and grease runoff into Teaneck Creek (NJDEP 2004). The rain garden is scheduled for construction in the spring of 2008.

Stream Restoration and Stabilization

As the site exists today, high stream flows are causing substantial stream bank erosion in the Teaneck Creek reach adjacent to the school. This erosion creates hazardous conditions on the school property in the form of steep, unstable stream banks. To remedy this problem, the Teaneck Board of Education has proposed a stabilization and restoration project, whose goals include moderating high stream flows, stabilizing the stream banks and channel, and reestablishing a native riparian buffer. Working in conjunction with the Conservancy project partners, the Teaneck Board of Education has submitted permit applications to the New Jersey Department of Environmental Protection (NJDEP). The proposed school property restoration plan addresses existing stream bank erosion and stormwater impacts, implements an invasive species management program in partnership with the U.S. Fish and Wildlife Service to control the Japanese knotweed, and commits to reestablishing a native riparian vegetative buffer along the entire length of Teaneck Creek where it flows through the Thomas Jefferson Middle School property. The Conservancy partners are working closely with the Teaneck Board of Education to provide Best Management Practices (BMPs) that will reduce ongoing maintenance and improve school safety and liability issues, while improving and enhancing this valuable ecological resource.

In evaluating stream bank stabilization and restoration options on the school property, the engineering team prepared hydrologic and hydraulic calculations to determine stream flows and velocities during storm events (USACE 1991). Using hydrologic soil group, land use, and impervious cover percentages, the analysis calculated a composite curve number of 82 for the nearly 300-acre drainage area (Table 1). As part of the design of the stabilization and restoration plan, the team also calculated the flow and frequencies associated with various storms events that could potentially affect Teaneck Creek (Table 2). The first phase of this project will stabilize approximately 200 linear feet of stream channel immediately downstream from the existing 7-1/2-foot-by-5-foot reinforced concrete pipe outlet discharging onto the school property. This 200-foot segment is currently experiencing extreme erosion and sedimentation due to the pipe discharge. The project proposes regrading and stabilizing the stream banks with cobble and natural "rip-rap" stone, installation of live staking, and extensive planting of native riparian shrubs and trees. In addition, a stabilized outlet and boulder rock-vanes are proposed to reduce velocities and redirect flows away from the side banks and toward the center of the stream (Rosgen 2001). All proposed stream channel modifications have been designed to achieve no net fill within the stream channel and floodplain (Figure 7).

The second phase of the project is restoration and stabilization of stream banks along an additional 600 linear feet of stream (Figure 8). Work in this section will include removal of a pedestrian bridge crossing the stream, regrading of stream banks to a 3:1 slope, and construction of a 4-foot-wide safety shelf. The regrading activities will remove the invasive Japanese knotweed and establish native riparian vegetation. Plantings and stabilization efforts will be enhanced with the installation of coconut fiber logs and boulders and the use of erosion-control mat or turf-reinforced mat.

Urban Restoration Partnerships

When working in urban regions, wetland habitat restoration efforts should look beyond the borders of the specific project site to evaluate potential affects coming from upstream and other off-site sources (Wolin and Mackeigan 2005). Without investigating off-site areas, unpredictable and unexpected conditions related to stream flows, stormwater drainage, landscape management, and maintenance can significantly influence the success of a restoration effort. A key step in successfully building support for the Teaneck Creek Conservancy project has been the effort to identify key property owners, managers, and information sources and establish strategic partnerships beyond the borders of the project site. Through informal and public meetings, local education outreach efforts, and work with citizen volunteers, project partners have obtained valuable insights and information. By partnering with community leaders and neighboring property owners, scientists and engineers have shared knowledge and built trust within the community. These outreach activities will lead to additional ecological improvements beyond the original Teaneck Creek restoration site, helping to ensure the success of the Teaneck Creek restoration efforts.

Acknowledgments

Funding support for Teaneck Creek studies has been provided by the New Jersey Wetlands Mitigation Council. We thank the staff of the Rutgers Cooperative Extension Water Services group for their contributions to this project, and we gratefully acknowledge the support of Holy Name Hospital, the Teaneck Board of Education, and the Thomas Jefferson Middle School, Teaneck, NJ.

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Table 1: Teaneck Creek Watershed Land-use Characterization

Sub Area	Area (sq. feet)	Soil Group	Land Use (Zoning Name)	Max. Imp. Cover (%)	Pervious Cover (%)	Runoff Curve Number (From%)	(CN*Area)
1	41.75	B	RS – Residential Single Family	40%	60%	72	3,014
2	118.59	B	RS – Residential Single Family	40%	60%	72	8,562
3	25.89	B	RS – Residential Single Family	40%	60%	72	1,869
4	0.53	B	B2 – Business District Office	65%	35%	83	44
5	20.98	B	B2 – Business District Office	65%	35%	83	1,740
6	77.00	B	P – Public Land District	70%	30%	85	6,553
7	24.18	B	P – Public Land District	70%	30%	85	2,058
8	1,303,760.87	B	RS – Residential Single Family	40%	60%	72	94,131,535
9	647,571.81	B	RS – Residential Single Family	40%	60%	72	46,754,685
10	0.90	B	RS – Residential Single Family	40%	60%	72	65
11	153,304.24	B	H – Hospital	70%	30%	85	13,046,191
12	1,502,052.57	B	RS – Residential Single Family	40%	60%	72	108,448,196
13	1,777.53	B	B2 – Business District Office	65%	35%	83	147,446
14	7,264.05	B	B2 –	65%	35%	83	602,553

			Business District Office				
15	960.29	B	B2 – Business District Office	65%	35%	83	79,656
16	86,986.89	B	B2 – Business District Office	65%	35%	83	7,215,563
17	0.01	B	B2 – Business District Office	65%	35%	83	1
18	25,364.95	B	B2 – Business District Office	65%	35%	83	2,104,023
19	21,122.57	B	P – Public Land District	70%	30%	85	1,797,531
20	4,567.64	B	P – Public Land District	70%	30%	85	388,706
21	13,919.72	B	B1 – Business District Retail	90%	10%	94	1,304,278
22	4,251.43	B	B2 – Business District Office	65%	35%	83	352,656
23	1,408.19	B	P – Public Land District	70%	30%	85	119,837
24	70,476.04	B	RS – Residential Single Family	40%	60%	72	5,088,370
25	2,050.77	B	B2 – Business District Office	65%	35%	83	170,111
26	892,710.16	C	RS – Residential Single Family	40%	60%	84	74,630,569
27	3,395,934.33	C	RS – Residential Single Family	40%	60%	84	283,900,110
28	630,625.42	C	H – Hospital	70%	30%	91	57,260,788
29	1,311,170.55	C	RS –	40%	60%	84	109,613,858

			Residential Single Family				
30	761,314.97	C	B2 – Business District Office	65%	35%	90	68,213,821
31	327,562.85	C	B2 – Business District Office	65%	35%	90	29,349,631
32	256,631.90	C	B2 – Business District Office	65%	35%	90	22,994,218
33	151,630.16	C	P – Public Land District	70%	30%	91	13,768,019
34	109,661.46	C	B1 – Business Retail	90%	10%	96	10,483,636
35	132,762.54	C	B1 – Business Retail	90%	10%	96	12,692,099
36	89,402.10	C	P – Public Land District	70%	30%	91	8,117,711
37	467,967.51	C	P – Public Land District	70%	30%	91	42,491,450
38	145,945.07	C	RS – Residential Single Family	40%	60%	84	12,201,008
39	26.49	C	RS – Residential Single Family	40%	60%	84	2,215

Data sources: NJDEP Land Use Land Cover 2002, Teaneck Township Zoning, and SSURGO Hydrologic Soils Classification, Bergen County. (B soils have a moderate infiltration rate; C soils have a slow infiltration rate).

Table 2: Teaneck Creek Storm Event Associated Flow Rates

Event Frequency	**Flow Rate (cfs)**
2-year	95.93
10-year	309.78
25-year	482.96
50-year	601.97
100-year	734.81

Note: Event frequencies required by NJDEP permitting process. Rainfall totals used to calculate stream flows are NRCS rainfall estimates.

Figure 1: Conceptual design plan for Holy Name Hospital rain garden and parking lot drainage area.

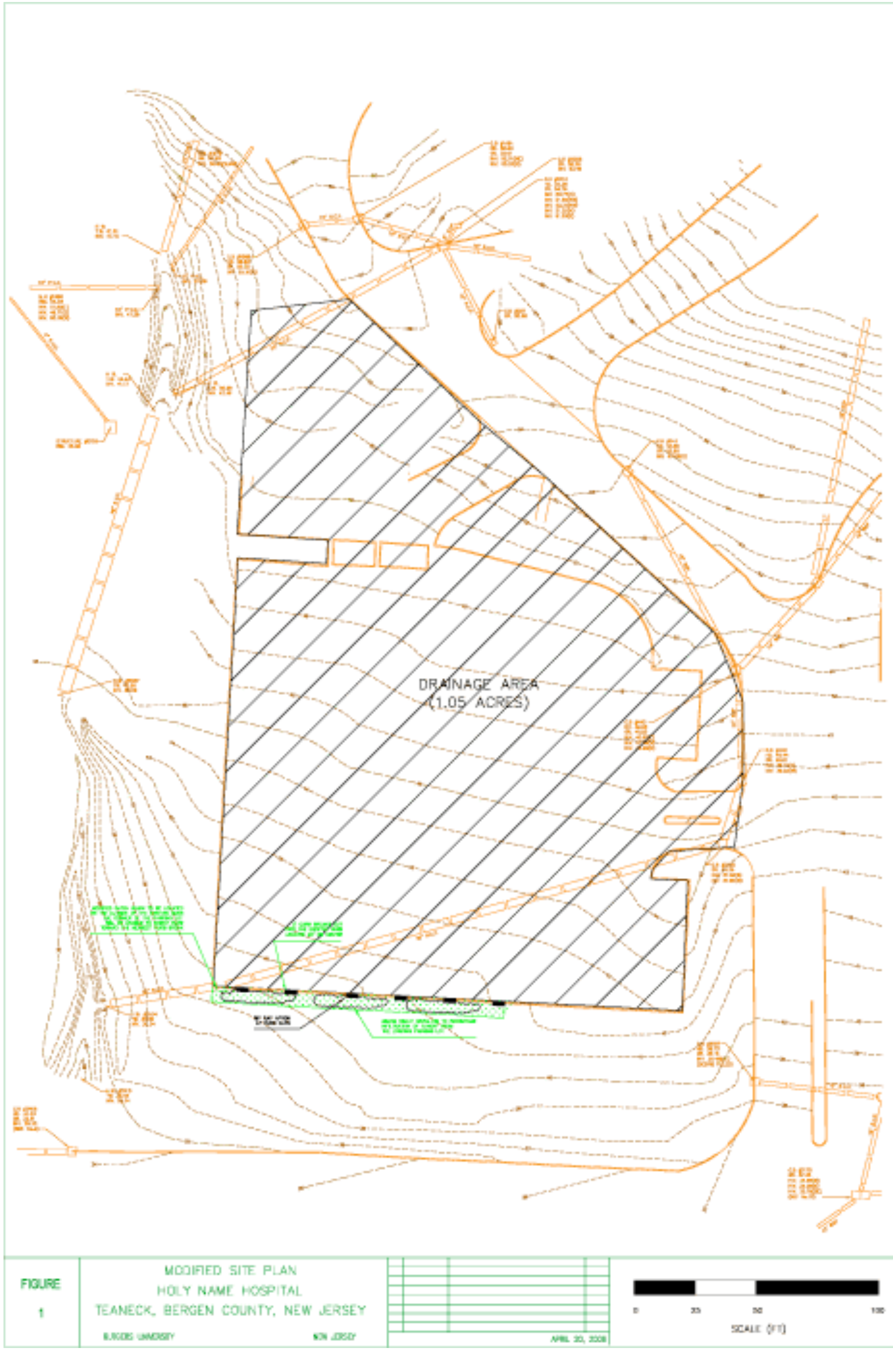


Figure 2: Conceptual details for Holy Name Hospital rain garden.

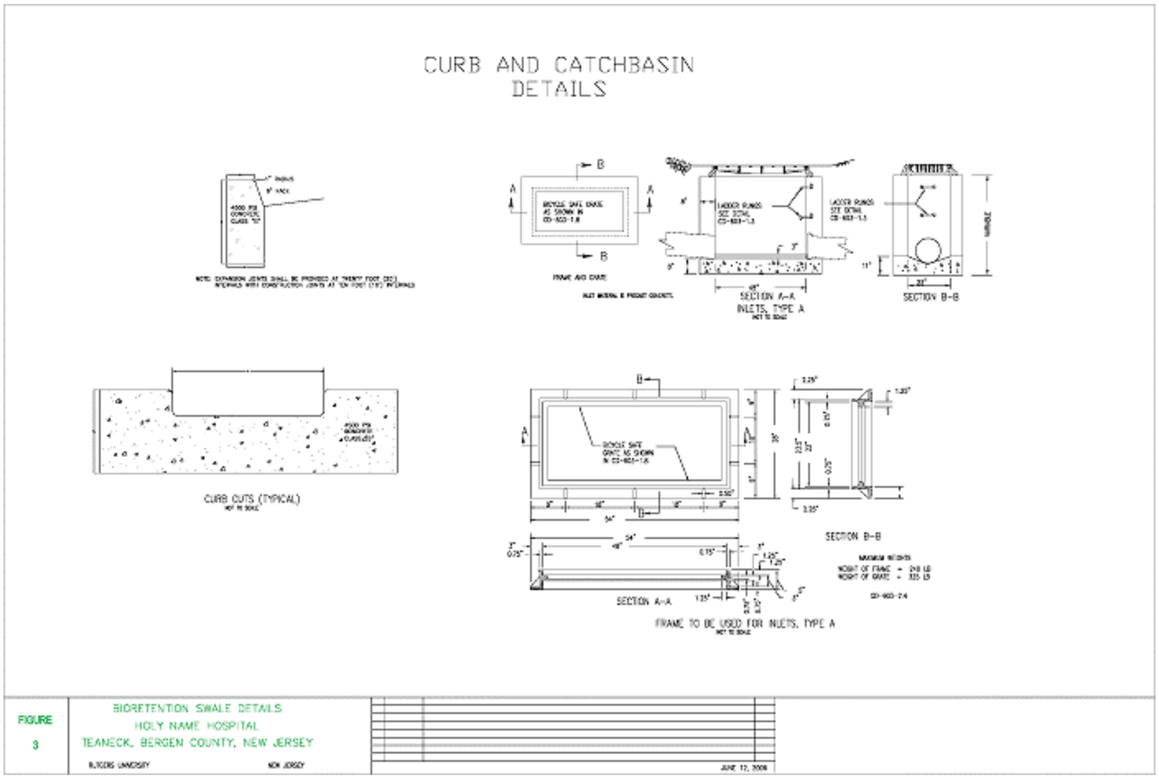


Figure 3: Conceptual profile for Holy Name Hospital rain garden.

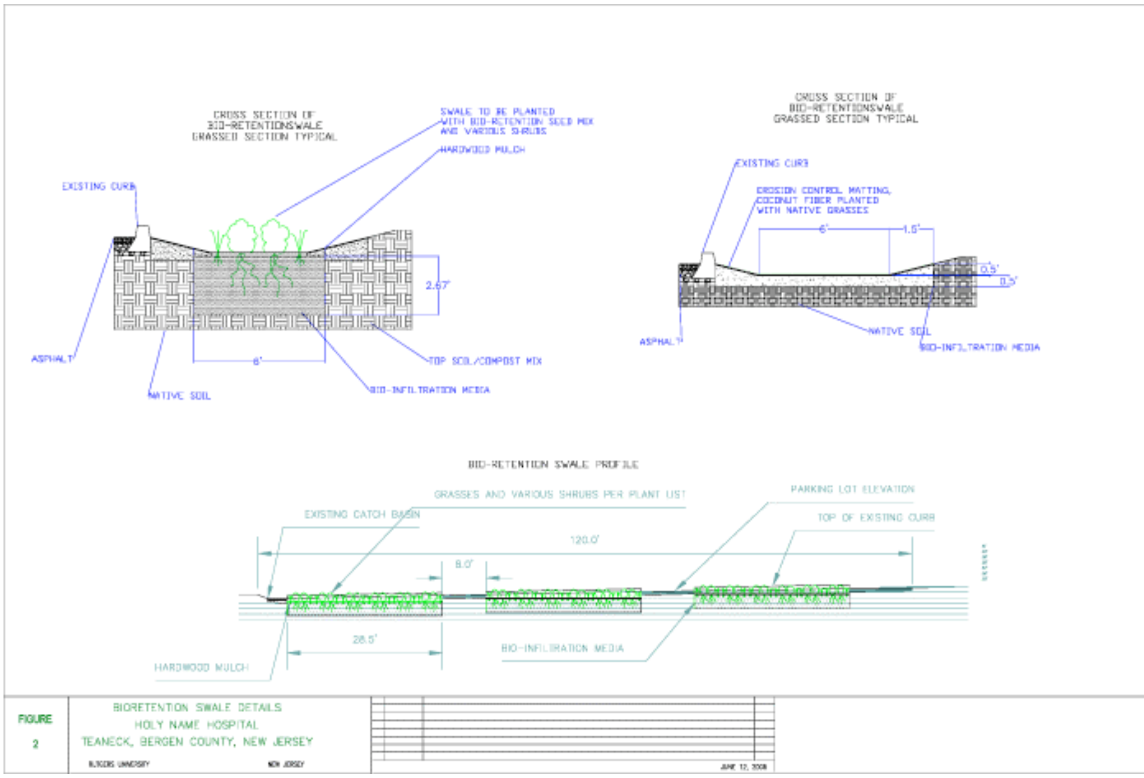


Figure 4: Erosion downstream from discharge pipe at Thomas Jefferson Middle School.

Figure 5: Teaneck Creek near Fycke Lane at Thomas Jefferson Middle School.

Figure 6: Japanese knotweed (*Polygonum cuspidatum*) colonization along Teaneck Creek on Thomas Jefferson Middle School property.

Figure 7: Outlet stabilization plan for Thomas Jefferson Middle School.

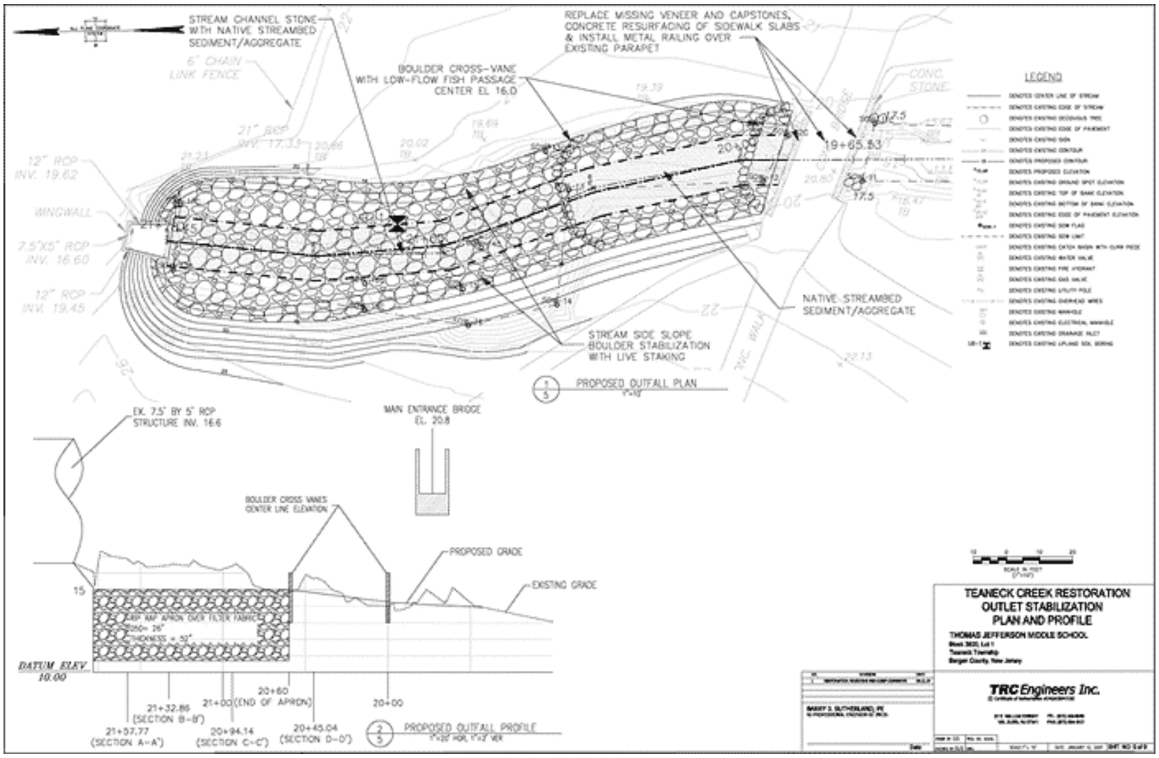
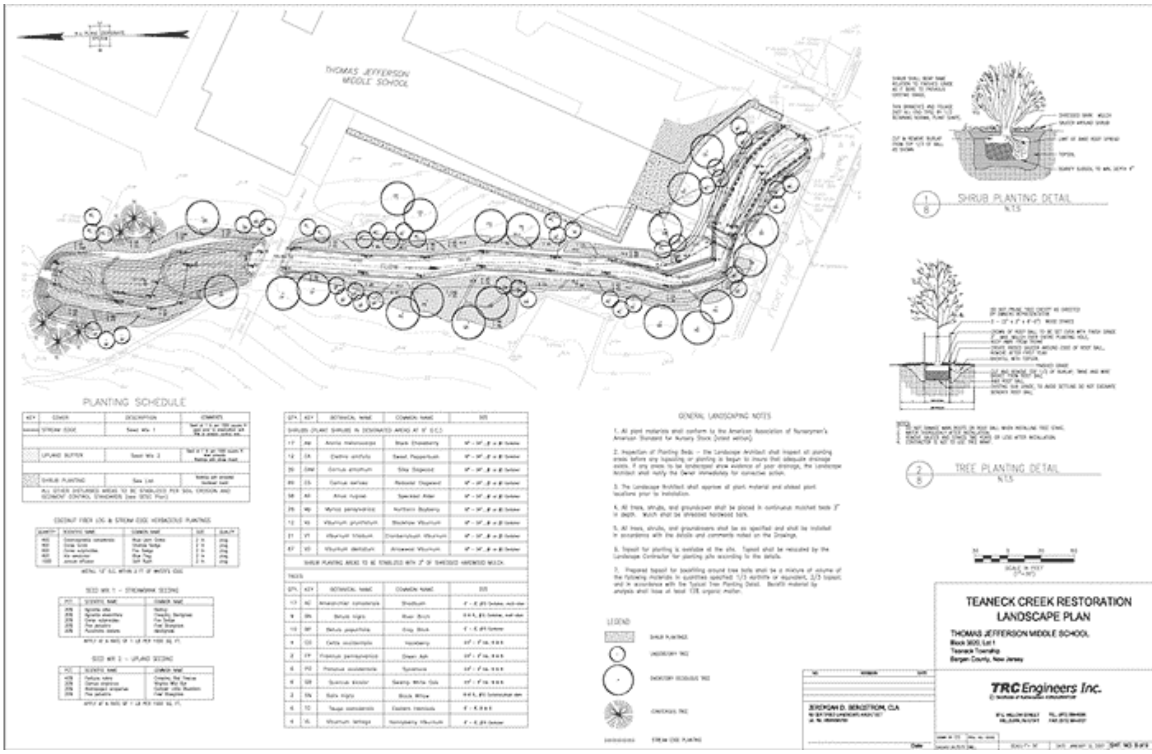


Figure 8: Landscape plan for Thomas Jefferson Middle School.



Modeling Urban Wetland Hydrology for the Restoration of a Forested Riparian Wetland Ecosystem

by Christopher Obropta[1,2], Peter Kallin[3], Michael Mak[4], and Beth Ravit[1]

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Abstract

To achieve our goal of a sustainable wetland system within a highly urbanized watershed, we required a model of the site's existing hydrology. This model will be used to develop a Conceptual Restoration Plan incorporating hydrology capable of sustaining the reestablished wetland system. Initial data suggests that the current system hydrology is dominated predominately by surface water flows. We have utilized the USEPA SWMM model to characterize water movement through 46 subbasins on this site. These simulated surface water flows will be used in conjunction with ground water, vegetation, and soil data to develop a Conceptual Restoration Plan for the site and to predict surface water movement through the reestablished wetlands.

Key words: Riparian wetland ecosystem, hydrology, SWMM model, water budget, runoff, aquaclude, perched bog

Introduction

Wetlands can be highly variable ecosystems that are characterized by fluctuating water levels and the prevalence of saturated soil conditions during the growing season. Riparian wetland ecosystems are positioned downstream of headwaters and typically receive runoff from their adjacent watershed (Grayson et al. 1999; Thurston 1999). Due to urbanization that occurred during the 20th century, many wetlands in highly developed areas in the Northeast United States have been cut off from their historic water sources. The hydrology of these urban wetland systems, including the inflows from their surrounding watershed, has been radically altered (Ehrenfeld et al. 2003).

To reestablish a sustainable 20-acre urban wetland system on the 46-acre Teaneck Creek Conservancy site, it is critically important to understand the site's existing hydrology. Based on data collected from over 40 groundwater wells installed on the site, information obtained from a wetland delineation, and soil profiles taken along a transect traversing the site from east to west, we have concluded that in areas where wetlands will be reestablished, subsurface and groundwater movement is currently negligible. Surface water flows in these areas dominate the hydrology because of the presence of fill materials, including a clay berm located adjacent to Teaneck Creek and a clay layer underlying most of the site at depths of 1 to 4 feet. Due to these historical disturbances, the wetland system currently appears to be functioning as a perched bog rather than a riparian corridor wetland. Therefore, we have prioritized characterization of surface water flow and development of a model to simulate these flows as the first step in determining pre-restoration baseline hydrology.

A comprehensive water budget is necessary to characterize the hydrology of an urban wetland system, but it is difficult to estimate the various components of urban hydrology or to create hydrologic simulations over extended time periods (Drexier et al. 1999). Although some water budgets have attempted to describe wetland hydrology (Konyha et al. 1995; Reinelt and Horner 1995; Hawk et al. 1999; Arnold et al. 2001; Kirk et al. 2004; Zhang and Mitsch 2005), models capable of describing urban wetland water flows are extremely few (Drexier et al. 1999; Raisin et al. 1999). We are aware of only one peer-reviewed study (Owen 1995) that attempted to develop a comprehensive urban water budget. This data gap is especially critical since current wetland modeling is derived from traditional pond design engineering (Konyha et al. 1995), which is a serious limitation when modeling wetland water fluctuations that are typically more subtle than water movement captured by pond models.

Lack of reliable data creates a challenge in determining how an urban wetland interacts with the adjacent watershed. Development of a hydrologic model that can accurately describe a given urban wetland is a necessary first step in the successful reestablishment of sustainable wetlands on a restoration site. The goal of this study was to characterize surface water movement as it currently exists in the urban wetlands of Teaneck Creek.

Modeling the Urban Teaneck Creek Surface Waters

Surface hydrology, in conjunction with groundwater hydrology and soil characteristics, controls the hydrology budget of a wetland. In highly urbanized locations such as the Conservancy site, the input of stormwater runoff into local wetlands is a potentially critical component of the water budget. High amounts of impervious cover (roofs, road surfaces) in urban areas increase stormwater runoff velocities and volumes. These increased velocities produce water budgets that differ from those of wetlands in non-urban settings (Göbel et al. 2004). Urban surface water inflows to the Conservancy site occur via both stream overbank flow and from six storm drains that discharge directly into the wetland system. Precipitation is most likely the dominant factor in a hydrologic simulation of the Conservancy's wetlands.

The basic hydrologic parameters of wetland water budgets include surface water influxes, precipitation, groundwater influxes, storage of water, percolation, and evapotranspiration (Owen 1995; Hawk et al. 1999; Reinelt and Horner 1995). There are three approaches used to model wetland hydrology: single event models, stochastic models, and comprehensive water budgets (Koob et al. 1999). The mass balance approach provides a framework for developing a water budget, which seeks to incorporate the parameters that control a wetland's hydrology. Further generalizations or additional parameters, such as a proposed restoration design of the system's hydrology may also be included in a model (Owen 1995; Reinelt and Horner 1995; WDWBM 1997; Yu and Schwartz 1998; Drexier et al. 1999; Raisin et al.

1999; Kincanon and McAnally 2004; Göbel et al. 2004; Mo et al. 2005; Xiong and Melching 2005; Zhang and Mitsch 2005).

Because urban hydrology may be subject to more highly fluctuating environmental conditions than a non-urbanized system, there may be an advantage in applying a stochastic model to urban wetlands, since this model type allows the incorporation of uncertainty into the model results. This approach contrasts with deterministic models, which produce identical results when provided with constant input parameters. Another alternative is to use a deterministic model with variable inputs to examine a range of conditions (e.g., dry conditions, wet conditions, average conditions).

Teaneck Creek Surface Water Hydrology

If an urban wetland system is characterized by minimal or nonexistent groundwater interactions, then the urban wetland may require a non-traditional modeling approach. In urban systems where the groundwater component is minimal, the most effective modeling approach to simulate hydrologic conditions may be the application of a nonlinear reservoir method, such as the United States Environmental Protection Agency Storm Water Management Model (SWMM). SWMM is a comprehensive deterministic model for urban stormwater runoff, capable of considering both water quality and quantity during a single event or on a continuous time frame (Huber and Dickinson 1988; Tsihrintzis and Hamid 1998; Bhaduri et al. 2001; Burian et al. 2001; Choi and Ball 2002; Lin et al. 2006; Smith et al. 2005; Xiong and Melching 2005). SWMM is designed to simulate real-time storm events based on spatial and temporal rainfall, evaporation, topography, impervious cover, percolation, depression storage values for impervious and pervious regions, storm drainage attributes such as slope and geometry, Manning's n, and infiltration rates (Burian et al. 2001; Bhaduri et al. 2001; Choi and Ball 2002). Based on these parameters, SWMM will model infiltration and storage and divert the remaining runoff as sheet flow (Burian et al. 2001). SWMM includes four simulation blocks to model urban stormwater runoff: Runoff; Transport; Extran; and Storage/Treatment.

When integrated with a GIS platform, SWMM is capable of developing simulations for defined subwatersheds existing within the boundaries of a given system. The watershed boundary is divided into smaller subdivisions based on land use, soil characteristics, impervious attributes, and topography (Smith et al. 2005), and this allows SWMM to generate runoff hydrographs based on daily rainfall data for each delineated subwatershed (Smith et al. 2005). An inflow of precipitation data will produce outflows of infiltration, evaporation, and surface runoff. Surface runoff will occur when each subbasin or reservoir reaches maximum storage. The depth of water for each subcatchment will be calculated continuously over the desired time step, through continuous calculations of the water balance. For each subwatershed, SWMM can simulate an individual rainfall event or a continuous simulation in time steps of minutes to years based on the system being modeled.

The SWMM model exhibits the highest potential to accurately simulate hydrological processes occurring within an urban wetland, and would thus be able to provide a solid framework for developing an accurate water budget for an urban wetland system. Through SWMM, the characteristics that define urban wetland systems with limited groundwater influences may be simulated on a continuous basis, providing a comprehensive description of the interaction of urban wetlands with the surrounding watershed. For these reasons, we chose SWMM to model the Teaneck Creek water flows. We chose to simulate the response of the Conservancy wetlands over a five-year period that included wet, dry, and average meteorological conditions (Table 1).

Materials and Methods

The wetlands on the 46-acre Teaneck Creek Conservancy site were delineated based upon vegetation, soils, and hydrology (Ravit et al. this volume). Soil characteristics of these wetlands have been highly modified by anthropogenic activity during major roadway construction in the 1950s and by current urban conditions (Arnold this volume), and these soil attributes are incorporated into the infiltration calculation in the SWMM model. Sewer system record survey maps of the Township of Teaneck (1972), 2002 NJDEP Orthoimagery, and 10-meter and 2-foot Digital Elevation Models (DEMs) were obtained to delineate the extent of the sewersheds draining into the wetland using ArcGIS 9.0. We analyzed the 10-meter DEMs, in conjunction with the invert elevations of the storm sewer lines in the township, to define the catchments and to provide the basis for assigning individual drainage areas to each catch basin. We delineated a total of 46 sub-sewersheds within the sewershed draining into the Conservancy wetland (Figure 1) and the size and slope characteristics of each were determined in ArcGIS.

The 46 sub-sewersheds (subcatchments) with their corresponding attributes and dimensions were constructed in EPA SWMM 5.0. The attributes of the subcatchments required to run a storm simulation consist of: area; width; slope percentage; percentage of imperviousness; infiltration method; and the outlet junction. The NRCS TR-55 SCS curve number infiltration method was used, based on the 1/8-acre or less (65% imperviousness) average residential lot size and the particular hydrologic soil group existing in each sub-sewershed (SCS 1986). The hydrologic soil group of each sub-sewershed was provided by the NRCS SSURGO soils data layer imported into ArcGIS 9.0. Once the entire sewershed was defined in SWMM, the six outfalls of the sewer lines and sub-sewersheds were modeled to complete the storm sewer portion of the system. The wetland subcatchments were then created using 2-meter DEMs within the boundary of the Conservancy. The attributes for the subcatchments were measured through ArcGIS 9.0 and imported into SWMM. Six subcatchments were delineated within the site, some of which flowed in different directions depending on the water elevations within the basins. This was simulated in SWMM using weirs and diversion structures, and each wetland basin was modeled as a pond with storage defined by the topography.

There are six stormwater inflows to the wetland and eight locations where water discharges to the Teaneck Creek and its tributaries. Figure 2 is a graphical representation of the predicted surface water routing through the wetland and Figure 3 shows the geographical location of the various wetland basins. Routing of water from the sewer system through the wetland and into Teaneck Creek was predicted using the 2-foot GIS contours for the wetland. This routing was field-verified by on-site visits during two rainfall events. Complete details of the SWMM model can be found in Mak (2007).

Model Field-Calibration

Two rainfall events were recorded at a representative location of the modeled system. A pressure transducer and a rain gauge were installed at Stormwater Canyon (S-1 in Figure 2), which receives runoff representative of the other sub-sewersheds within the drainage system and is the primary source of water to the largest wetland area that will be reestablished. The rainfall events were input into SWMM to calibrate the model through a comparison of the predicted flow versus the actual flow measured during these two storm events. The pressure transducer recorded water depth throughout the storm at 4-minute intervals, requiring a rating curve to calculate the actual flow through the canyon. Previously recorded Stormwater Canyon flow measurements were used to develop the rating curve for the two storm events. The recorded rainfall data were input into SWMM with the corresponding dates and time steps of the storm events. The output data from each simulation were then imported into Microsoft Excel for model validation. For these storm events, two subsets of simulations were run for the model calibration. The parameters adjusted during the calibration of the model were the curve numbers representing the infiltration routing processes, the percentage of impervious area with no depression storage, and percentage impervious cover values for each subbasin in the sub-sewershed under review. Plots of observed versus measured flow for each calibration simulation were then analyzed for the validation of the model.

Validation

To validate the model, we used a numerical integration method (trapezoidal rule) to analyze the measured versus predicted values. We calculated the total runoff volume for each simulation using the trapezoidal rule and compared this to the measured flows. At the calibration point, the measured versus predicted values for total runoff volume differed by only 2.06% (Mak 2007).

Water Budget Calculations

Once the model was calibrated and validated, we used it to generate annual rainfall simulations to develop a water budget. To simulate an annual rainfall event, 15-minute and hourly precipitation data in DSI-3260 and DS-3240 format, respectively, were imported into SWMM. Due to completeness of the data set and relative proximity to the project site, the precipitation records from Newark Airport (Table 1) were used for these annual simulations. Figure 4 shows the overall logic flow of how the model was developed and used to calculate annual water budgets.

The SWMM model was used to predict the volume of water draining into and out of the TCC wetland from the surrounding sewershed. Using this information, we created a monthly water budget for the entire wetland for the years of 2000 through 2005. The calculation of the water budget was done in Microsoft Excel using runoff data imported from EPA SWMM 5.0, the New Jersey State Climatologist (http://climate.rutgers.edu/stateclim_v1/monthlydata/index.html), and the National Climatic Data Center (NOAA) (www.ncdc.noaa.gov). The calculation of the water budget follows a mass balance approach provided by Mitsch and Gosselink (2000) and Owen (1995). The general mass balance exists as (change in storage = input – output). The mass balance applied to the wetland is derived from the expression:

Equation 1: Water Budget Equation

$\Delta S = P + S_i + G_i - AET - I - S_o - G_o \pm T$

Where:

ΔS = change in storage volume

P = precipitation

S_i = surface water inflow

G_i = ground water inflow

AET = actual evapotranspiration

I = infiltration

S_o = surface water outflow

G_o = ground water outflow

T = tidal flow

For each annual simulation, surface water inflows (S_i) and outflows (S_o) in cubic feet per second (CFS) were imported from EPA SWMM 5.0 and converted into units of acre-feet for the water budget calculations. Hourly precipitation values (DS-3240 format) from Newark International Airport (Station #286026) were obtained from the National Climatic Data Center (NOAA) (www.ncdc.noaa.gov) for the years 2000 through 2005. The Newark station is located approximately 16 miles from the Conservancy wetlands and contains the most complete hourly rainfall data sets of any station in the vicinity. The precipitation (P) inputs for the wetland itself were calculated by summing the hourly data (in inches) and converting to acre-feet.

Potential evapotranspiration (PET) was calculated on a monthly basis using the Thornthwaite equation (Mitsch and Gosselink 2000):

Equation 2: Potential Evapotranspiration (PET)

$$PET_i = 1.6 \left(\frac{10T_i}{I} \right)^a$$

PET_i = PET for month I (mm/mo)

T_i = mean monthly temperature (^oC)

I = local heat index, $\sum \left(\frac{T_i}{5} \right)^{1.514}$

A = (0.675. $I^3 - 77.1.I^2 + 17{,}920. I + 492{,}390.10^{-6}$

We chose the Thornthwaite method because of its simplicity and reasonable accuracy (Mitsch and Gosselink 2000). Only air temperature is required to derive values for PET occurring within the wetland. Air temperature data were retrieved from a continuous weather monitoring station located in Lyndhurst, New Jersey, approximately 10 miles from the Conservancy. These data were provided by the Meadowlands Environmental Research Institute (MERI). Data were retrieved from this station because of its close proximity to the project area and the availability of the data. Actual evapotranspiration (AET) values were derived by applying a correction factor to the calculated PET.

For the purposes of this simulation, we assumed groundwater inflows (G_i) to be negligible and did not include them in water budget calculations. Although there is some evidence of groundwater movement in portions of the wetland, a highly impermeable clay layer exists underneath much of the system, minimizing the influences of ground water. The existence of a dense clay layer under most of the wetland acts as an aquaclude and causes the system to act essentially as a perched bog, with some infiltration into surficial sediments above the clay layer and very slow movement toward the creek. We have observed a few seeps along the creek bank in several areas that flow for a few days after large rainfalls, which support this assessment. These infiltration losses (I) were calculated by SWMM based on the soil characteristics in the wetland basins and converted from inches to acre-feet. We calculated the total infiltration loss for the entire system by summing the values for the individual wetland basins for each month during the 6-year simulation period.

All of the model inputs and collected data were imported into Excel to compute the monthly budgets for the years 2000 through 2005 to simulate the current conditions of the existing wetland. We combined monthly precipitation totals and simulated runoff totals to represent the total inflow into the system, and we combined actual evapotranspiration, infiltration loss into the wetland, and simulated outflow totals to represent the total outflow from the system.

Results

The change in storage of the system each month was calculated by subtracting the total outputs from the total inputs of the system. This represents the amount of water stored in or removed from the wetland system each month. To calculate the cumulative storage for the wetland system, we added the change in storage for each month to the previous month's cumulative storage, resulting in the cumulative storage plot shown in Figure 5. Table 1 summarizes the monthly and annual precipitation values for the six-year period of analysis. Years 2000 (44.45 inches) and 2005 (47.78 inches) were slightly below the six-year mean precipitation (51.06 inches); the amount of water in the wetland at the end of those years was roughly the same as at the beginning. Year 2001 (37.47 inches) was the driest year analyzed; the wetland ended the year with a deficit of about 70 acre-feet compared to the beginning of the year. This deficit did not fully recover until the end of 2003 (54.77 inches), which was the wettest year in the period analyzed.

We averaged the monthly change in storage values (all Januaries, all Februaries, etc.) over the six years to generate average monthly storage changes. These are shown in Figure 6, along with the cumulative plot of the average values. During "average" precipitation years, the wetland gains water in the spring and fall and loses water in the summer. The detailed data used for calculations of the water budget are included in Mak (2007).

Discussion

A methodology has been developed for analyzing the water budget of the Teaneck Creek urban wetlands, based on a surface water–dominated system. While the results presented here are for the entire TCC wetland complex, the SWMM model can be used to analyze water budgets for each of the individual wetland basins shown in Figure 3. The model can be used to analyze each wetland basin, separately or in combination, and to evaluate the effects of various restoration options, such as grading changes or installing water-control structures. Also, the model can be used in combination with water quality data to analyze nutrient loadings to various areas within the wetland.

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Table 1: Monthly precipitation (inches) by year as measured at Newark Airport.

YEAR	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOTAL
2000	2.79	1.51	2.77	3.31	3.89	5.30	7.30	4.57	3.66	0.54	4.08	4.73	44.45
2001	1.45	1.98	4.72	2.29	3.03	7.43	1.76	4.55	5.44	0.82	1.36	2.64	37.47
2002	1.21	0.91	3.99	5.49	5.12	5.36	1.70	3.93	4.79	8.33	5.73	4.00	50.56
2003	3.34	2.66	4.09	2.76	3.45	6.29	2.96	6.72	6.93	5.90	3.94	5.73	54.77
2004	2.10	3.19	3.12	5.04	4.60	2.58	8.39	3.38	8.76	0.96	4.87	3.72	50.71
2005	4.36	2.80	4.84	3.84	1.64	2.28	4.18	0.40	2.61	12.40	4.28	4.15	47.78

Driest year = 2001
Wettest year = 2003
Average years = 2000, 2005
2001–2006 Mean = 51.06"

Figure 1: Sewershed System based on the Township of Teaneck Digital Elevation Model (DEM) — 10 meter.

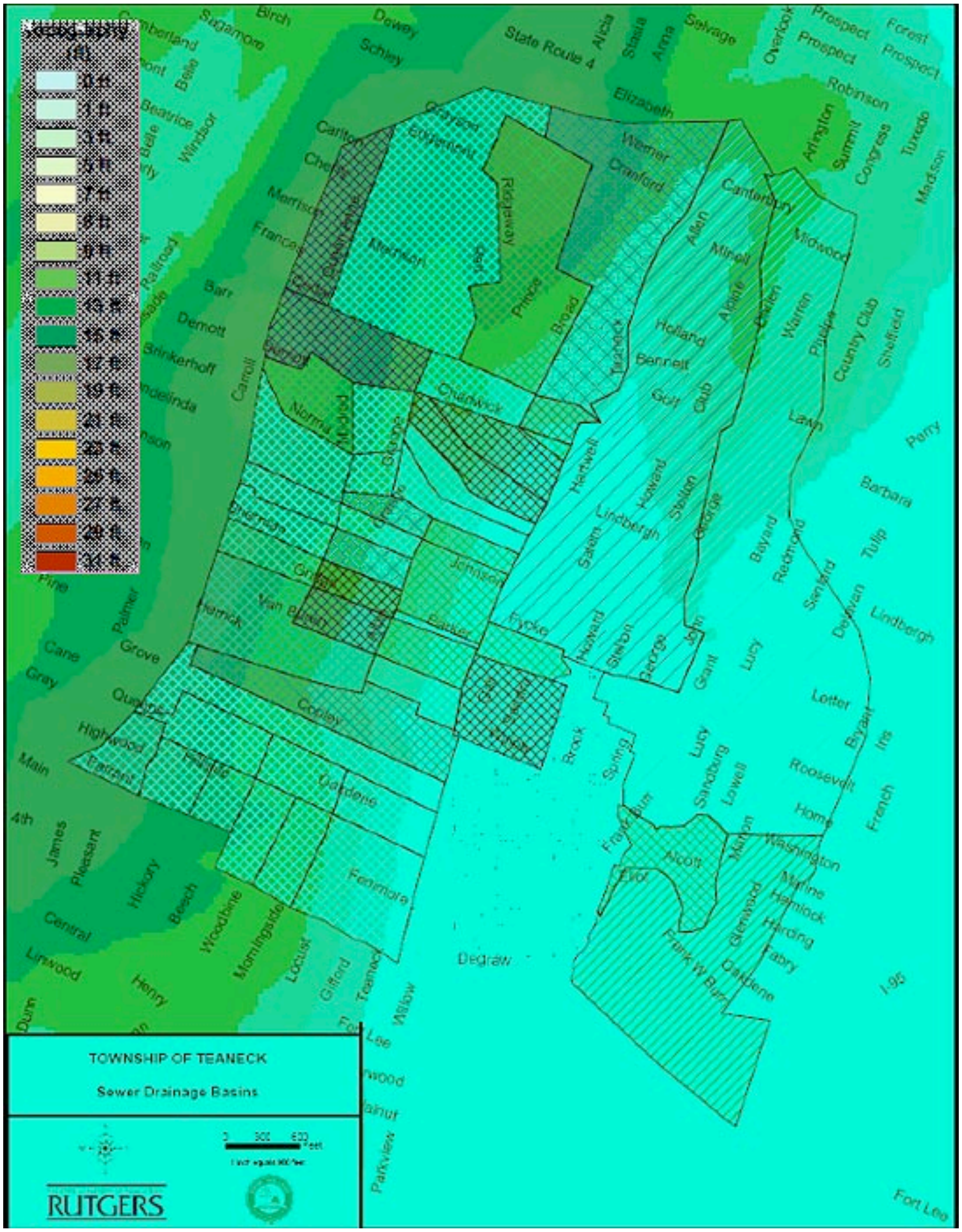


Figure 2: Surface water routing through Teaneck Creek Conservancy wetlands. S = inflows to the Conservancy wetlands; O = outflows from the Conservancy wetlands.

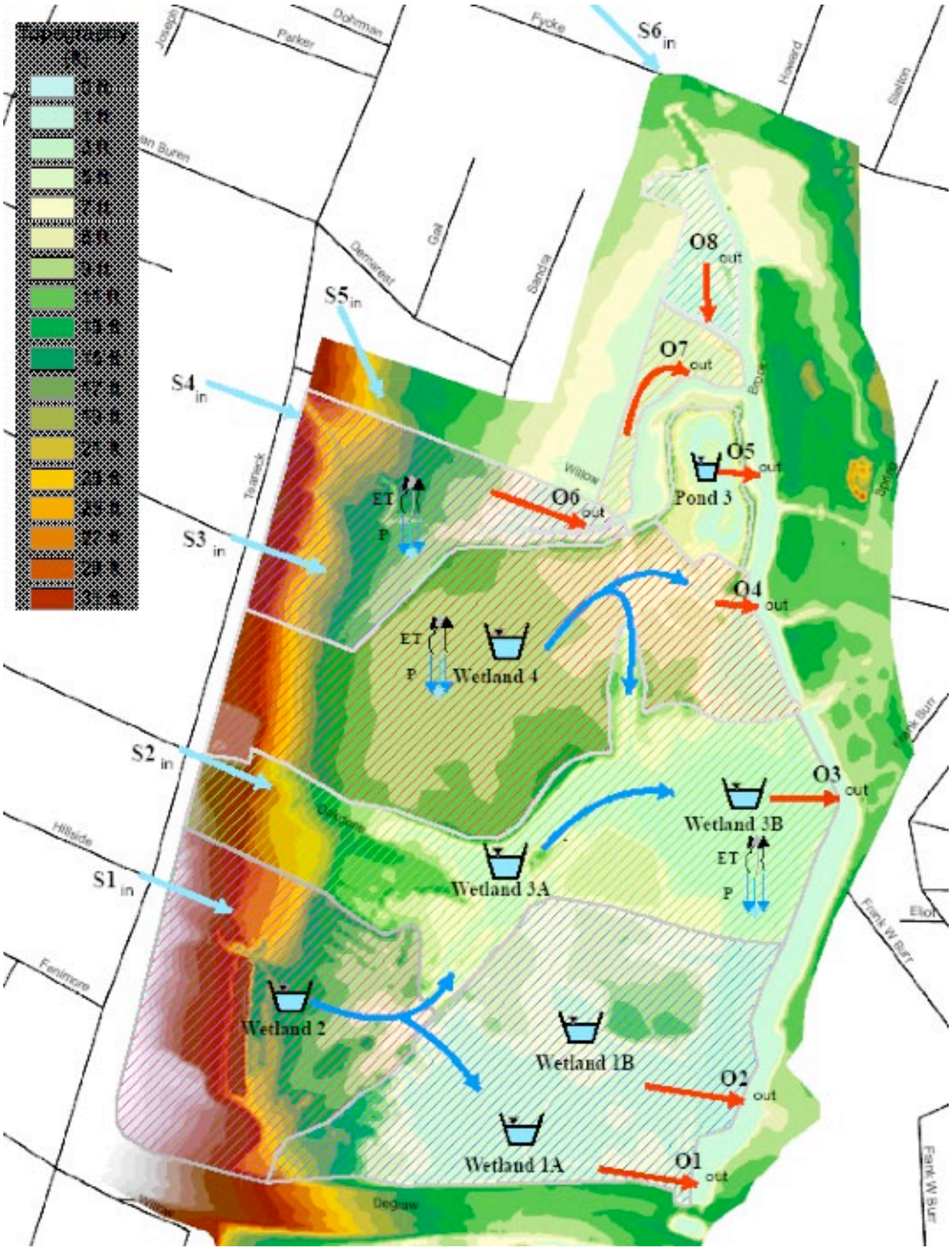


Figure 3: Teaneck Creek Conservancy wetland areas.

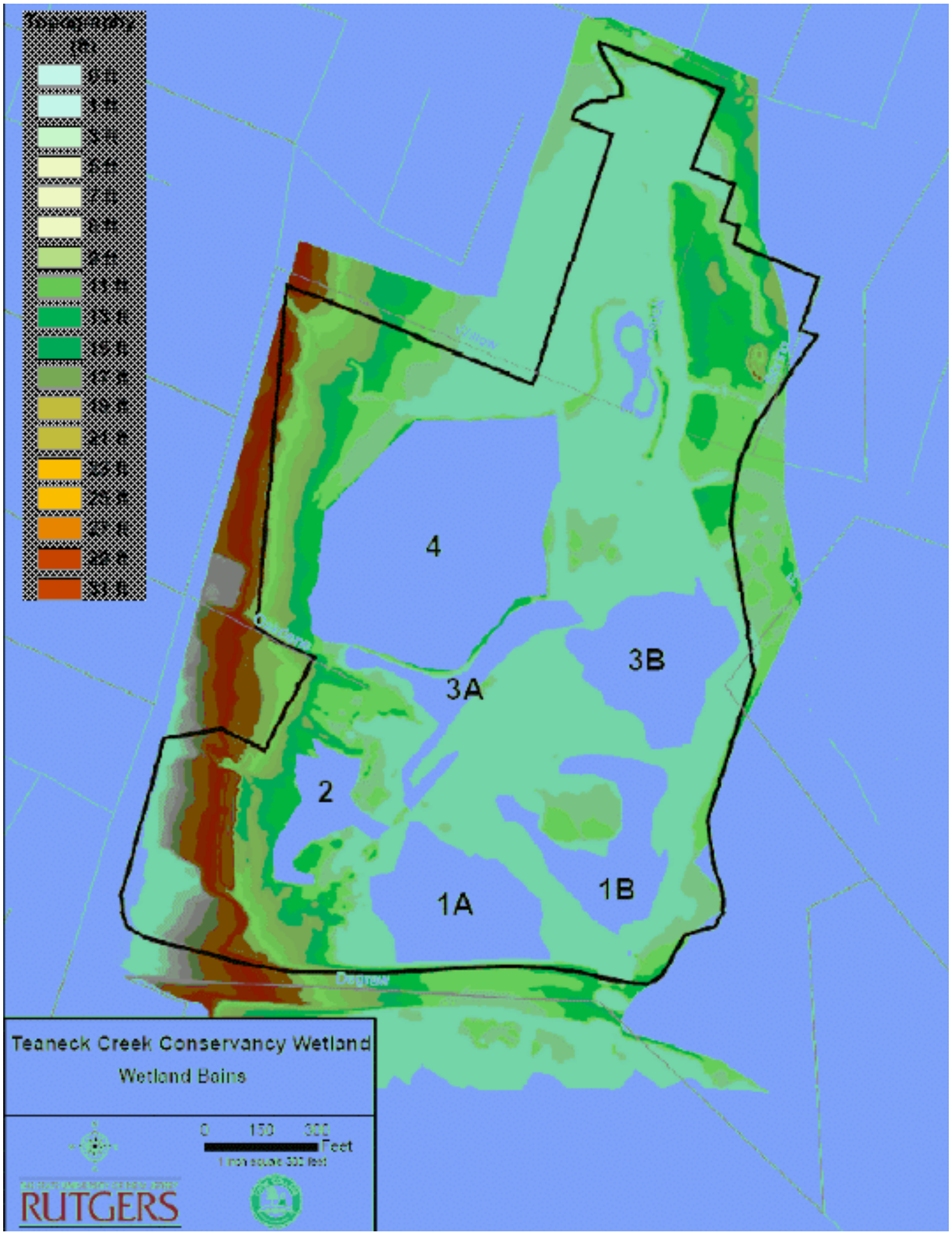


Figure 4: Logic flow for model development and water budget calculations.

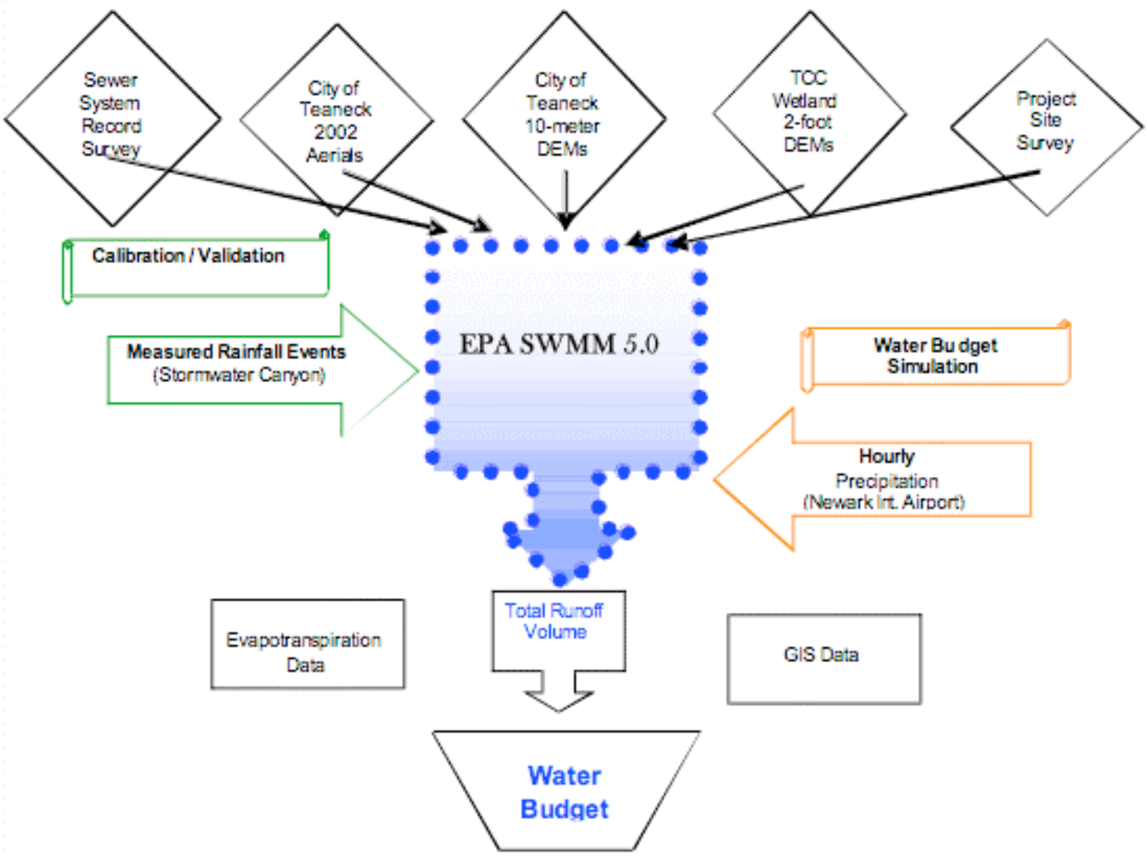


Figure 5: Cumulative monthly water budget for the period 2000–2005.

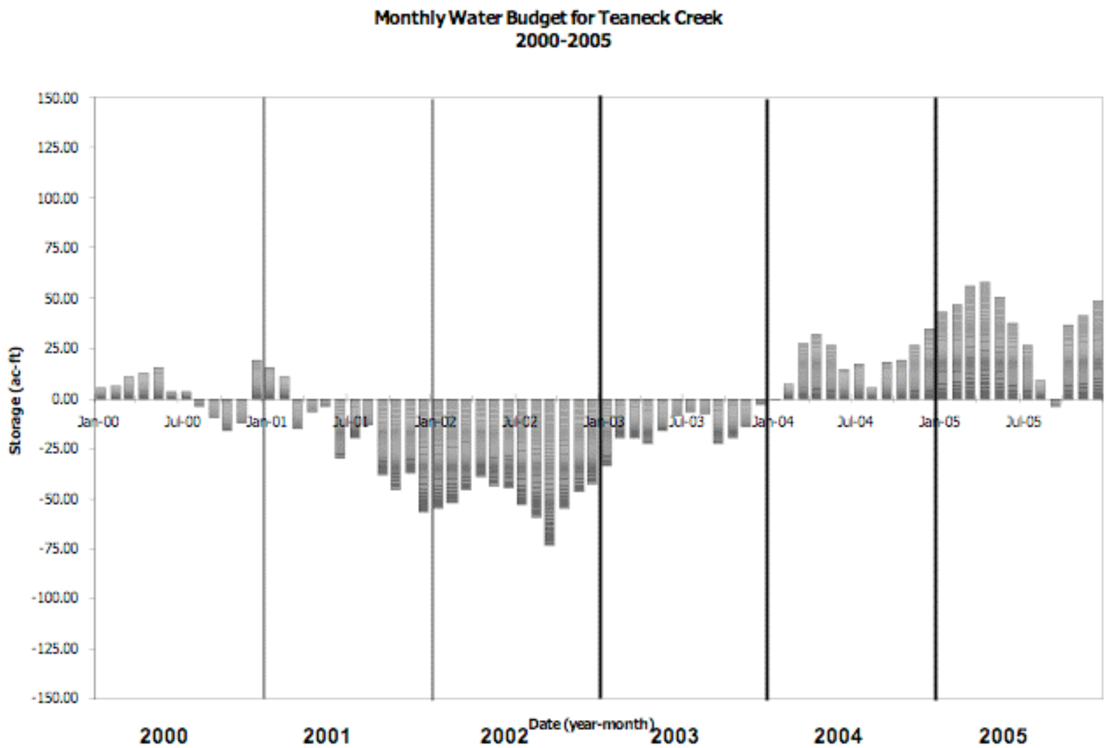
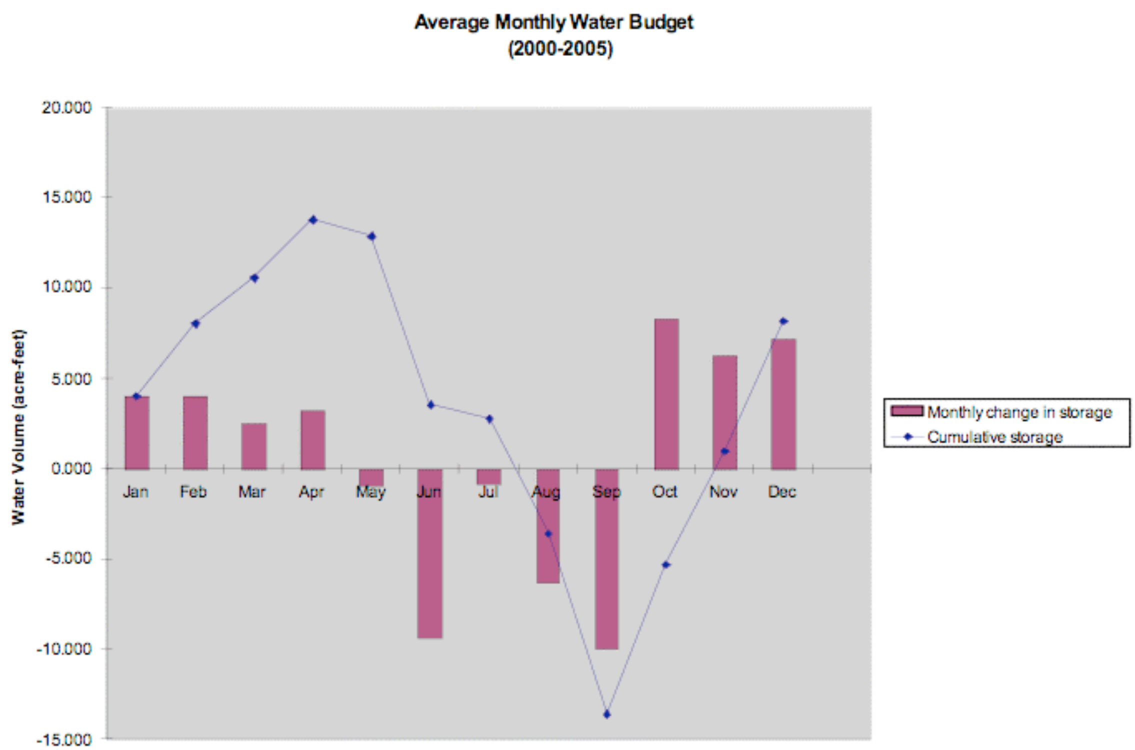


Figure 6: Average monthly Teaneck Creek Conservancy water budget for the period 2000–2005.



A Vegetation Survey of Teaneck Creek Wetlands

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Abstract

A project goal for the restoration of Teaneck Creek wetlands is to establish native plant communities within these rehabilitated wetland areas and to eliminate or control the spread of invasive plants. To determine the location of the existing native vegetation and to characterize the substrate quality (native hydric soils versus fill materials) and moisture (wet versus dry) associated with this plant community, we visually identified and ranked the abundance of the flora on the site. Using the New Jersey Coefficient of Conservatism (NJ CC), we calculated a Floristic Quality Assessment Index (FQAI) for twenty-nine 100-meter by 100-meter sampling units. Plant diversity was found to be high (245 species) compared to other New Jersey urban wetlands, and native species comprised 60% of the total number of plant species observed. Two thirds of the total number of tree and shrub species were native, while only half the vine/forb/herb species were native. Introduced species were found to have invaded a minimum of 30% of each sampling unit and a maximum of over 50% in a *Phragmites*-dominated interior area, where plant diversity was the lowest seen on the site. The ten highest FQAI-value native species were predominantly wetland plants. A comparison of the FQAI value with the soil type and moisture properties indicates that wet soils may be the more important of the two variables in structuring the existing vegetation at this site. The FQAI score identified a high quality wetland area that must be guarded from disturbance during restoration activities. The FQAI score, in combination with soil properties and/or moisture content, will be used to inform the decision-making process as the Teaneck Creek wetland Conceptual Restoration Plan is developed.

Key words: urban, wetland, native, invasive, Floristic Quality Assessment Index, diversity, restoration, hydrology, Conceptual Restoration Plan

Introduction

Existing vegetation on the 46-acre Teaneck Creek Conservancy wetland site (Figure 1) consists of a mixture of native and introduced plant species. Many of the non-native species, such as garlic mustard (*Alliaria petiolata*), Japanese knotweed (*Polygonum cuspidatum*), mile-a-minute vine (*Polygonum perfoliatum*), and multiflora rose (*Rosa multiflora*), are aggressively invasive and have formed monospecific expanses in certain areas of the site. Overall goals for rehabilitation of the site include reestablishing the hydrologic connectivity between Teaneck Creek and its interior surface and ground waters, and the removal of fill materials, resulting in reestablishment of 20 acres of wetlands. Specific project goals include the establishment of native wetland flora typically found in northern New Jersey riparian corridors, the protection of existing native plants growing in hydric soils, and the elimination of invasive vegetation within these wetland areas. Although this is not the usual definition of restoration, for the sake of simplicity, we will use this term to refer to these project goals.

As an aide in characterizing the site, we utilized a Floristic Quality Assessment Index (FQAI) to describe and evaluate the existing flora (Lopez and Fennessey 2002). The FQAI has been adopted in several other geographic locations for the purposes of wetland assessment (Mushet et al. 2002; Cohen et al. 2004; Bourdaghs et al. 2006; Miller and Waldrop 2006). It is used to characterize the conservation value of multiple site locations that potentially may be altered by restoration activities. This methodology assigns a subjective ranking called a "coefficient of conservatism" (CC) to each plant species. Species more likely to be found in natural areas are assigned higher numbers, while species commonly found in disturbed areas are given lower numbers (Matthews et al. 2005).

Using the values obtained in the FQAI characterization, the restoration approach will prioritize high FQAI-value areas that should remain undisturbed during and following wetland restoration on the site. Low FQAI-value areas will be considered as candidates for hydrologic and soil restoration activities followed by subsequent replanting with native species. We also used the FQAI value to test whether hydrology and/or soil properties were factors in determining the vegetation patterns observed.

Methods

Field Sampling

In order to obtain a coarse-scale view of the vegetation on the Teaneck Creek Conservancy site at a resolution of 1 hectare, we established a grid system (100-meter by 100-meter sampling units) and overlaid it onto an aerial GIS based map of the site (Figure 2). Sample units were labeled from south to north with alphabetic letters and from west to east with numbers. Each unit was visited at least twice during the summer and fall of the 2006 growing season, beginning in late May and ending in early November. Sampling activities were performed by a single observer who made multiple traverses within each sampling unit, recording plant species present and visually estimating coverage of each species. We note the following difficulty in data collection: Although our objective was to traverse each sample unit completely, due to the density of invasive vegetation and the presence of standing water, there were portions of the interior areas which were not totally accessible. In these cases, the observer traversed as much of the sample unit as was physically possible, but our data may contain sampling errors as a result of these physical limitations.

Observers made a visual estimate of plant abundance based on the percent cover of each species visible within the sample unit. A scored five-level scale was employed: The lowest score (1) = "rare" was assigned if the species occurred as a single plant, or only a few individuals, or if the populations were very small and highly localized. A species was scored as (2) = "few" if it occurred in several small populations throughout the unit, or as many isolated individuals that constituted less than 10% of the overall cover. A species was scored as (3) =

"occasional" if it contributed approximately 10% to 40% of the total cover, or if it occurred in several substantial populations within the unit. Species that occupied 40–60% of the sampling unit, or that were distributed as individuals throughout the unit in virtually all locations were scored as (4) = "common." Species that constituted > 60% of the total unit cover were scored as (5) = "abundant." The highest abundance level attained by a species throughout all sampling events was retained when data from all site visits were consolidated.

After the vegetation in each sampling unit was identified, we obtained the New Jersey coefficient of conservation (NJ CC) for each species (Bowman 2006). This coefficient describes the habitat requirements for a particular species, including its sensitivity to disturbance (Matthews et al. 2005). Coefficient values ranged from 0 to 10, and introduced plants are always assigned a 0. The NJ CC for all species within a sample unit was then used to calculate a Floristic Quality Assessment Index (FQAI) for each sampling unit cell.

Soil and Moisture Properties

Dr. Kallin assigned a wetness rating to each sampling unit cell based on the dominant hydrologic condition(s) observed while performing the site's wetland delineation (Ravit et al. this volume). This characterization was based on the presence of saturated soil, inundation, hydric soil criteria, water table data, and a visual determination as to the proportion of the sample unit that met hydric soil criteria, with (1) = primarily wet (> 60%); (2) = primarily dry (< 40%); (3) = mixed (40–60%). This was based on criteria in the Federal Manual for Identifying and Delineating Jurisdictional Wetlands (FMIDJW 1989). Utilizing multiple soil borings in each sampling unit, Dr. Kallin also characterized the soil quality with respect to the type and source of the dominant substrate material(s), assigning values as (1) = primarily native soil; (2) = primarily dredge fill; (3) = primarily dredge fill with debris; and (4) = mixed. The use of the term "native" describes non-fill substrate that exhibited soil horizons and textures indicative of a native glacial soil and that had native vegetation growing in the surrounding area. A visual evaluation of each sampling unit was also conducted.

Statistical Analysis

All analyses of variances (ANOVAs) were conducted using SAS System GLM (SAS Software, Version 9.1). Due to the high level of heterogeneity on this site as a result of anthropogenic disturbances, we set the threshold for significant differences between sampling units at the $\alpha = 0.10$ level. We acknowledge that this choice was somewhat arbitrary, but due to the heterogeneity and the fact that there were only 28 sample units, we opted to use a less restrictive alpha test. Due to the coarse scale of the sampling in this study, and because the Simpson Diversity Index is weighted toward abundances of the most common species, we used this index to determine plant species diversity (PC-Ord, Version 4). ANOVA was used to test for differences in the diversity scores among the sampling units, and two-factorial ANOVAs (Independent Variables = MOISTURE × SOIL, Dependent Variable = FQAI score) were used to test if there were interactions that might influence the FQAI value. We note that the FQAI value is not an abundance measure, and so weighs the presence of rare and common species equally. Conversely, the Simpson Diversity Index is weighted toward abundance of the most common species (Magurran 1988), and so better describes the presence of dominant invasive monocultures.

Results

Overall, the number of plant species found on the Conservancy site was high compared to other New Jersey urban wetlands (Ehrenfeld 2005). A total of 245 plant species (contact author for full plant list) were identified within the Teaneck Creek Conservancy, and all species observed have been reported as present in the New York metropolitan region (Clemants and Moore 2003). The number of species within a given sample unit ranged from a low of 20 to a high of 83, with a mean per sampling unit of 50 species (Table 1). Of the plants identified, 145 were native species and 98 were species that have been introduced to this area.

Thirty-three species were observed in more than 50% of the sampling units (Table 2). The 4 most widely distributed species, found in over 90% of the sampling units, included common reed (*Phragmites australis*), garlic mustard (*Alliaria petiolata*), porcelainberry (*Ampelopsis brevipedunculata*), and multiflora rose (*Rosa multiflora*), which are all considered invasive. We note that although *Phragmites australis* can be categorized as a native species (Clemants and Moore 2003), there is a genotype which originated outside the U.S. that has invaded and replaced native genotypes throughout eastern coastal marshes (Saltonstall 2002). Although the plant found on the Conservancy site has not been genetically tested, because the invasive genotype dominates the nearby Hackensack Meadowlands ecosystem, we are assuming that our plant is the invasive form, and so have treated it as nonnative in our analyses. All the sample units were heavily invaded by nonnative species, although the number of widely distributed native species (19) was slightly greater than the number of widely distributed introduced species (13). Across the entire site, more than 40% of the species identified were nonnative, and five sample units had more than 50% nonnative species cover. Trees and shrubs exhibited the highest proportion of native species (approximately two thirds of the total number identified) as compared to vines and forbs (approximately half the species were native). The most commonly observed native plants tended to be wetland species, while the highly distributed introduced plants were predominately upland species.

The ratio of the numbers of native versus introduced species per sample unit ranged from 0.7 to 2.4, with a mean of 1.5 (Table 3). This ratio was higher under wet (Figure 3) versus mixed or dry conditions ($F_{2,25} = 2.46$, $p = 0.1$), suggesting that wetter hydrology may favor native species. The top ten high NJ CC value native plants were wood bulrush (*Scirpus expansus*) (obligate wetland, or OBL); hobblebush (*Viburnum alnifolium*) (wetland or upland, or FAC); bitternut hickory (*Carya cordiformis*), (facultative upland, or FACU); smartweed (*Polygonum amphibium* var. *emersum*) (OBL); wild leek (*Allium tricoccum*) (FAC+); spring cress (*Cardamine bulbosa*) (OBL); American linden (*Tilia americana*) (FACU); false hellebore (*Veratrum viride*) (facultative wetland, or FACW+); swamp white oak (*Quercus bicolor*) (FACW+); and wild yam (*Dioscorea villosa*) (FAC+). Except for American linden and bitternut hickory, these species are all obligate or facultative wetland species.

Soil quality was highest in the sampling units at the northern end of the property and in portions of the eastern and western borders (Figure 4), where the soils were composed of primarily native organic material. The interior of the site was dominated by dredged materials, and the soil adjacent to the southern boundary is unconsolidated fill/debris. However, the soil type was not found to be a significant factor in determining the number of plant species, the ratio between native versus introduced species, or the FQAI score within a sampling unit. A two-factor ANOVA comparing the FQAI values found no interactions between moisture and soil type.

The highest quality FQAI sampling units were located at the northeastern portion of the site (Figure 5), which had the lowest proportion of introduced plant species (30%). The FQAI score ranged from a high of 22.8 in this northeastern corner to a low of 6.3 in the *Phragmites*-dominated interior and areas adjacent to the DeGraw Avenue southern boundary of the property. Diversity (as measured by the Simpson Diversity Index) was found to be significantly lower ($F_{26,1} = 63.84$, $p = 0.098$) in the Phragmites-dominated D3 sampling unit than in the high FQAI G2 and H2 areas (Figure 6).

Discussion

Although surrounded by highly urbanized land use, the forested wetlands of the Conservancy contained 245 different plant species. Significant differences were found in the distribution of native versus introduced plant species, and in habitat conservation values across the site. The overall number of native species was 60% of the total species on site, a proportion quite similar to that observed by Clemants and Moore (2003) in their survey of native and nonnative flora in large northern urban areas. We note that at the Conservancy site, the proportion of introduced species is four times greater than that reported by Ehrenfeld (2005) in northern New Jersey forested wetlands. However, because the two studies used different sampling methods, it is possible that differences in the proportion of nonnative species are the result of sampling methodologies.

Habitat values as described by the FQAI score appear to be more strongly influenced by hydrology than the various soil substrates. The importance of hydrology in determining wetland vegetation is well documented (Toner and Keddy 1997; Magee and Kentula 2005; Dwire et al. 2006), and in this study, 9 of the grids with the highest FQAI values were associated with wet or mixed moisture regimes versus 3 high FQAI-value grids characterized as dry. Conversely, 8 of the highest FQAI-value locations were composed of fill or mixed materials, while only 4 high FQAI-value locations had native soils. The two highest FQAI values were associated with wet and native soils (locations G2 and H2), and these areas must be protected from disturbance during and following restoration activities. However, it is obvious from our observations that these two variables alone will not guarantee high FQAI scores (see locations B5 and G1).

The results of this study will be used to delineate low FQAI-value areas where removal of fill and/or reintroduction of saturated hydrology could produce environmental conditions that would support replanting of native wetland flora (A2, A3, A4, C3, D4, F4). Conversely, areas that have been filled, yet exhibit high FQAI values, may be better left as they currently are (B3, B4, C1, C2). One question left to be decided is how to address relatively large wet areas with low FQAI value (see D3, D4, E3) that are currently functioning as a Phragmites-dominated detention basin for stormwater storage.

Future analyses will combine hydrology information related to the subwatersheds on site (Obropta et al. this volume) with data from the vegetation and soil surveys to determine native vegetation best able to survive in the reestablished wetland areas. A second vegetation study has now been set up that tests the ability of different facultative wetland plants to survive in field plots under the various combinations of wet versus dry, and native versus fill soils. The results of this study will help identify plant species likely to survive under environmental conditions that will be present in the Conservancy's rehabilitated wetlands. This study also shows the need for a comprehensive invasive control plan to be included as a component of the Conceptual Restoration Plan.

Acknowledgments

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Table 1: The effect of moisture and soil properties on the number of species (mean ± standard deviation), the ratio of native to introduced species, and the FQAI scores in the Teaneck Creek Conservancy site. Intro = Introduced non-native species; FQAI = Floristic Quality Assessment Index.

Moisture	No. of Species	Native:Intro	FQAI
Dry	58 + 15.5	1.4 + 0.37	16.0 + 2.51
Wet	49 + 21.1	1.7 + 0.44	15.6 + 4.43
Mix	46 + 12.3	1.3 + 0.39	13.8 + 3.42
	NS	$F_{2,25} = 2.46$, $p = 0.10$	NS

Soil	No. of Species	Native:Intro	FQAI
Dredge	45 + 22.9	1.6 + 0.43	13.1 + 5.10
Fill	52 + 16.0	1.3 + 0.39	14.7 + 3.25
Mix	53 + 22.8	1.6 + 0.51	15.6 + 3.03
Native	50 + 17.1	1.6 + 0.45	16.9 + 3.64
	NS	NS	NS

Table 2: The 33 species distributed in 50% or more of the 28 Teaneck Creek Conservancy sampling units surveyed. Wetland plant indicators: OBL = wetland plant (99% of time); FAC = occurs in wetland or upland; FAC W = usually occurs in wetland (67–99% of time); FAC U = occasionally occurs in wetlands (1–33% of time). Number of grids = the number of sampling units where a species was observed.

Native					
Scientific Name	**Common Name**	**Wetland Indicator**	**Growth Habitat**	**No. of Grids**	**Percent of Grids**
Acer negundo	Box elder	FAC +	Tree	15	> 50%
Acer rubrum	Red maple	FAC	Tree	22	> 75%
Acer saccharinum	Silver maple	FACW	Tree	21	> 75%
Ageratina altissima	Rough snakeroot	FACU-	Forb/Herb	15	> 50%
Allium vineale	Wild onion	FACU-	Forb/Herb	15	> 50%
Fraxinus pennsylvanica	Green ash	FACW	Tree	14	50%
Geum canadense	White avens	FACU	Forb/Herb	24	> 75%
Impatiens capensis	Jewelweed	FACW	Forb/Herb	25	> 75%
Juglans nigra	Black walnut	FACU	Tree	19	> 50%
Oenothera biennis	Common evening primrose	FACU-	Forb/Herb	14	50%
Parthenocissus quinquefolia	Virginia creeper	FACU	Vine	20	> 50%
Phragmites australis	Common reed	FACW	Graminoid	26	90%
Phytolacca americana	Pokeweed	FACU+	Forb/Herb	21	> 75%
Polygonum virginianum	Jumpseed	FAC	Forb/Herb	17	> 50%
Populus deltoides	Eastern cottonwood	FAC	Tree	21	> 75%
Prunus serotina	Black cherry	FACU	Tree	19	> 50%
Salix nigra	Black willow	FACW+	Tree	14	50%
Symplocarpus foetidus	Skunk cabbage	OBL	Forb/Herb	15	> 50%
Toxicodendron radicans	Poison ivy	FAC	Vine	17	> 50%
Ulmus americana	American elm	FACW-	Tree	21	> 75%

Introduced					
Scientific Name	**Common Name**	**Wetland Indicator**	**Growth Habitat**	**No. of Grids**	**Percent of Grids**
Acer platanoides	Norway maple		Tree	15	> 50%
Ailanthus altissima	Tree of heaven		Tree	16	> 50%
Alliaria petiolata	Garlic mustard	FACU-	Forb/Herb	26	90%
Ampelopsis brevipedunculata	Porcelainberry		Vine	29	100%
Artemisia vulgaris	Mugwort		Forb/Herb	15	> 50%
Catalpa bignonioides	Southern catalpa	UPL	Tree	15	> 50%
Morus alba	White mulberry	UPL	Tree	16	> 50%
Polygonum cuspidatum	Japanese knotweed	FACU-	Forb/Herb	17	> 50%
Polygonum perfoliatum	Mile-a-minute vine	FAC	Vine	25	> 75%
Robinia pseudoacacia	Black locust	FACU-	Tree	16	> 50%
Rosa multiflora	Multiflora rose	FACU	Shrub	26	90%
Setaria spp.	Foxtail grass		Graminoid	14	50%
Solanum dulcamara	Bittersweet nightshade		Vine	15	> 50%

Table 3: Attributes of the 28 individual sampling unit 100 meter by 100 meter cells. Diversity scores were computed using the Simpson Diversity Index. Designations for soil properties: 1 = "native;" 2 = "dredge fill;" 3 = "fill + debris;" 4 = "mixed." Designations for soil moisture: 1 = "Dry;" 2 = "Wet;" 3 = "Mixed."

Sampling Grid	Soil	Moisture	No. Species	Native:Intro	FQAI	Diversity
G2	1	2	78	1.79	23.1	0.984
H2	1	2	65	2.42	22.4	0.987
B4	2	3	56	1.95	18.7	0.979
D5	4	2	83	1.68	18.5	0.986
C2	3	1	74	0.90	18.4	0.985
C4	2	1	70	1.69	17.9	0.984
C5	1	2	62	1.82	17.8	0.983
D2	4	2	56	2.11	17.6	0.980
C1	3	1	42	1.21	17.1	0.973
B3	3	2	37	1.64	16.5	0.969
E4	3	2	62	1.48	16.5	0.983
F3	1	2	44	1.10	16.5	0.975
B2	3	3	52	0.73	15.9	0.978
E2	2	2	62	1.95	15.8	0.982
F4	1	3	43	1.39	15.6	0.975
F2	1	3	36	1.25	15.0	0.970
C3	3	3	67	1.23	14.8	0.984
B5	1	2	58	1.23	14.7	0.980
E1	4	2	31	1.82	14.2	0.963
A1	3	1	65	1.83	13.4	0.983
G1	1	2	22	2.14	13.4	0.950
B1	1	1	42	1.47	13.2	0.973
A2	4	2	40	0.90	12.0	0.972
A4	3	3	39	1.60	11.9	0.972
D4	2	3	46	1.09	10.9	0.976
D3	2	2	14	1.80	8.5	0.916
A3	3	3	28	0.87	7.8	0.960
E3	2	2	20	1.00	6.6	0.942

Figure 1: A map of New Jersey showing the location of the Teaneck Creek Conservancy restoration site.

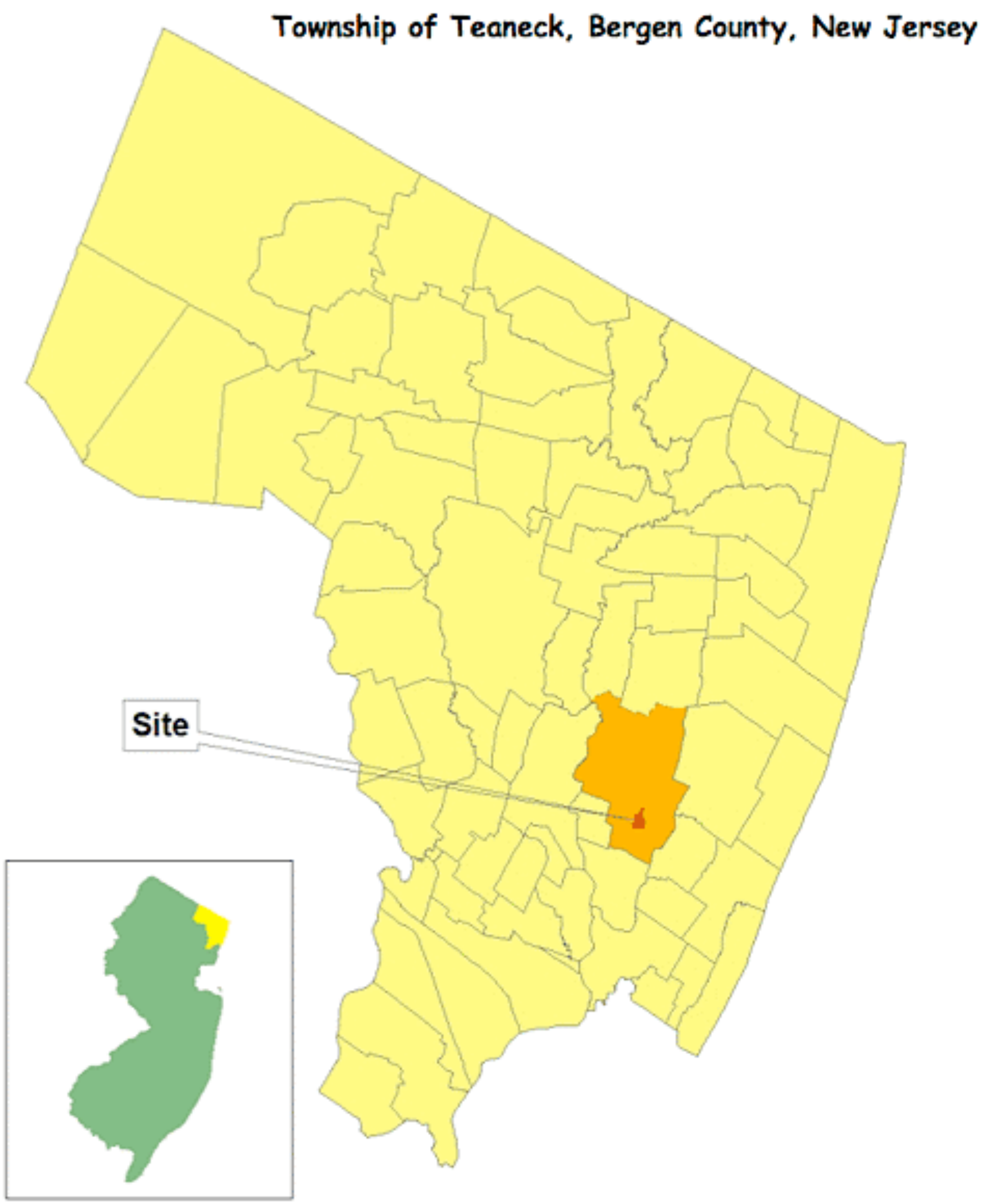

Figure 2: Teaneck Creek Conservancy (site outlined in blue) aerial map overlain with 100 m × 100 m sampling unit cells. Map courtesy of Bergen County.

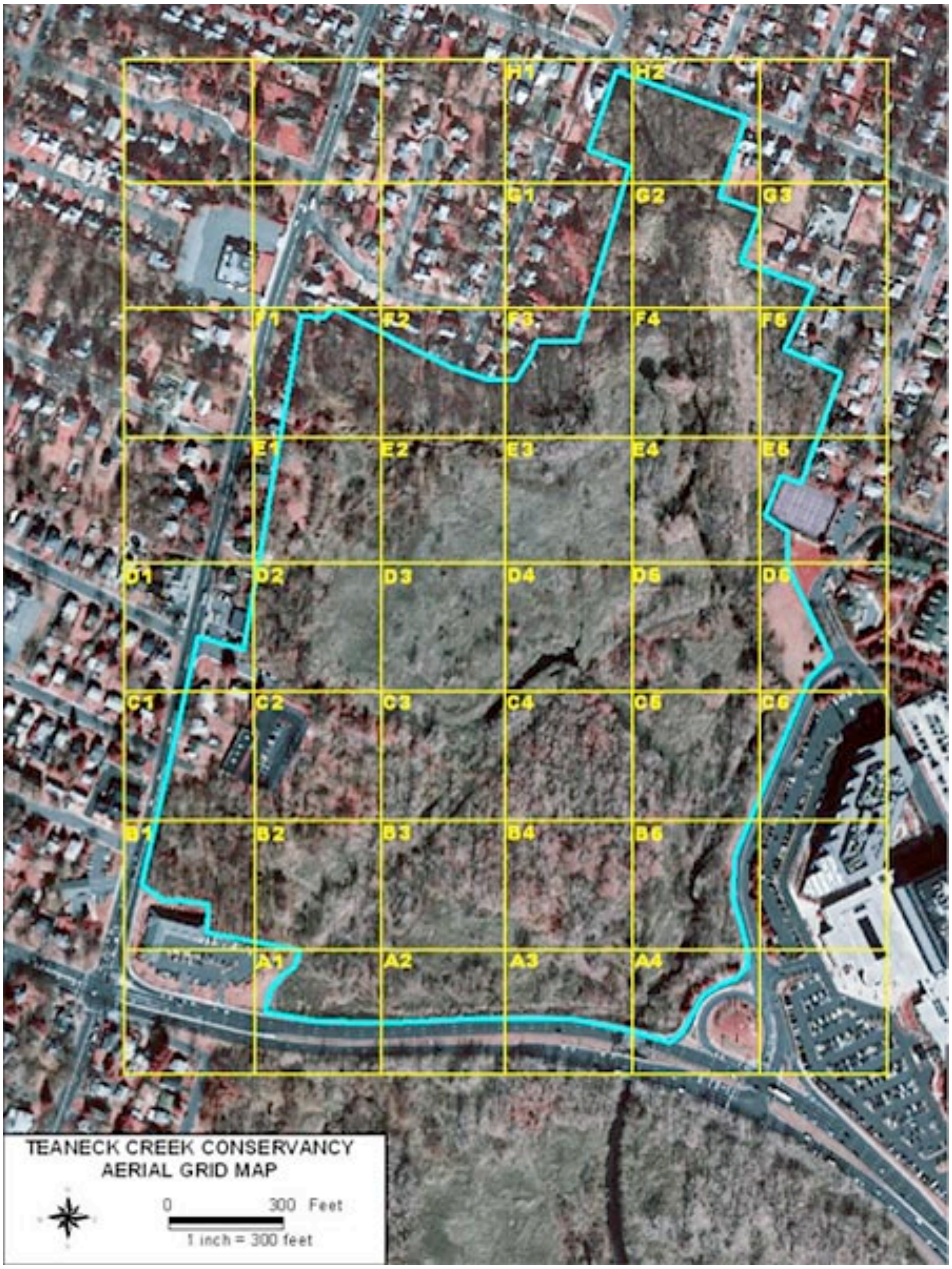

Figure 3: Dominant soil moisture property of each Teaneck Creek 100 m × 100 m sampling unit.

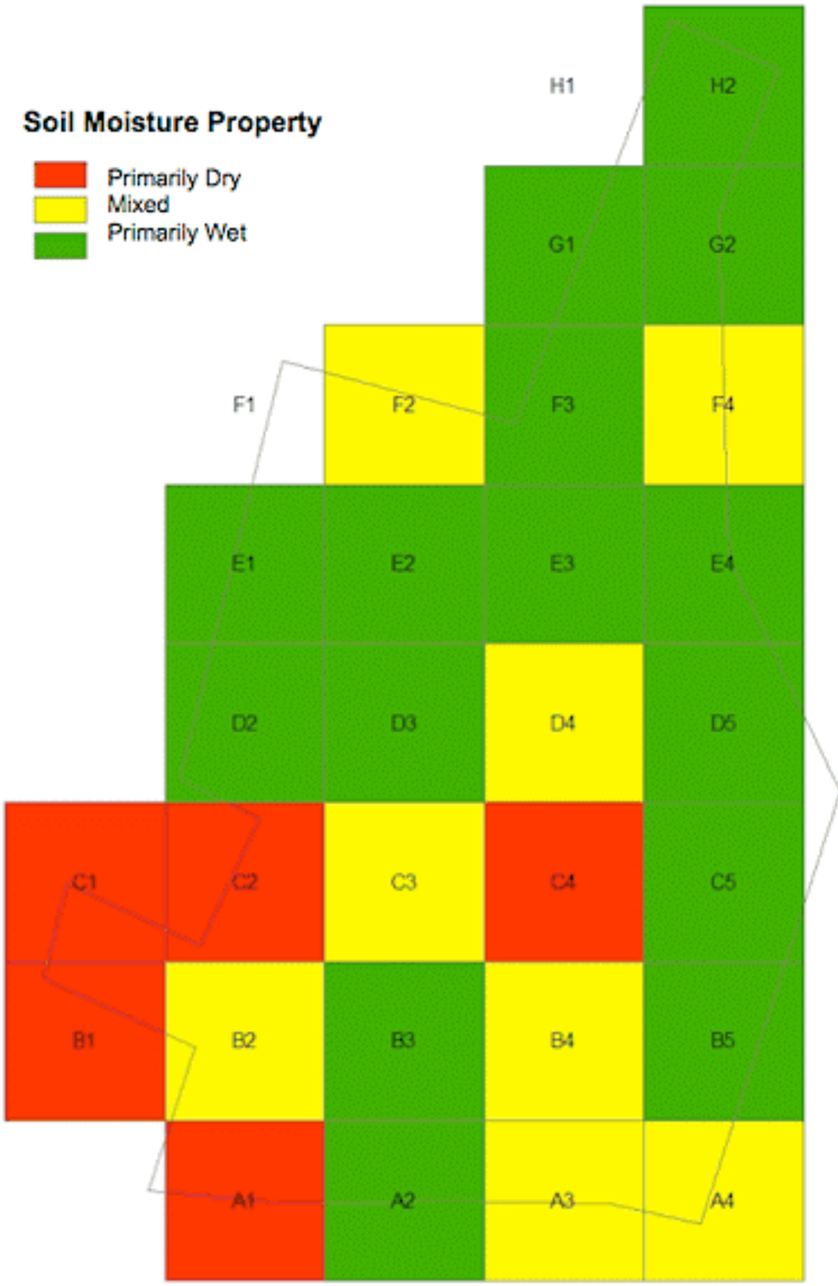

Figure 4: Dominant soil properties of each 100 m × 100 m Teaneck Creek sampling unit.

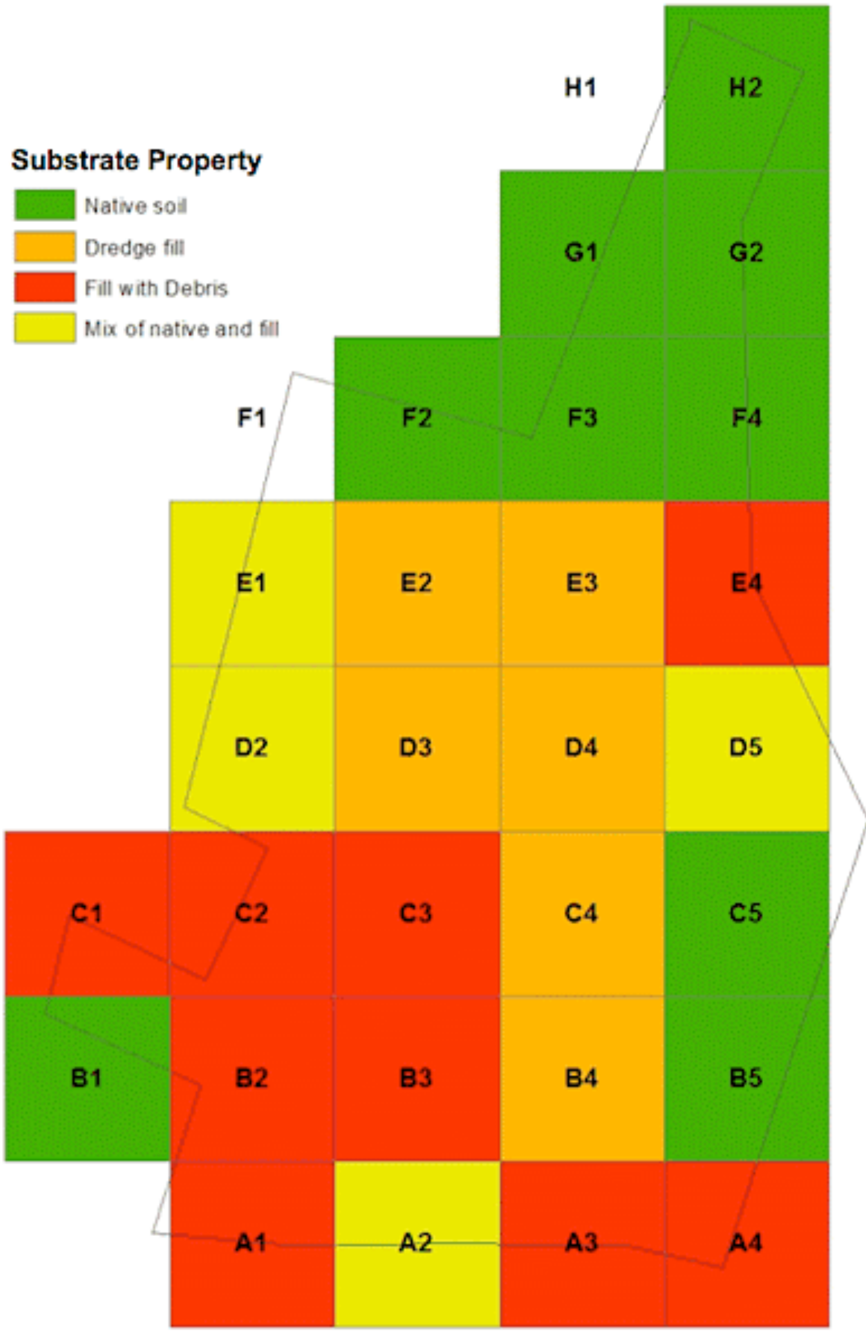

Figure 5: Floristic quality of each 100 m × 100 m Teaneck Creek sampling unit.

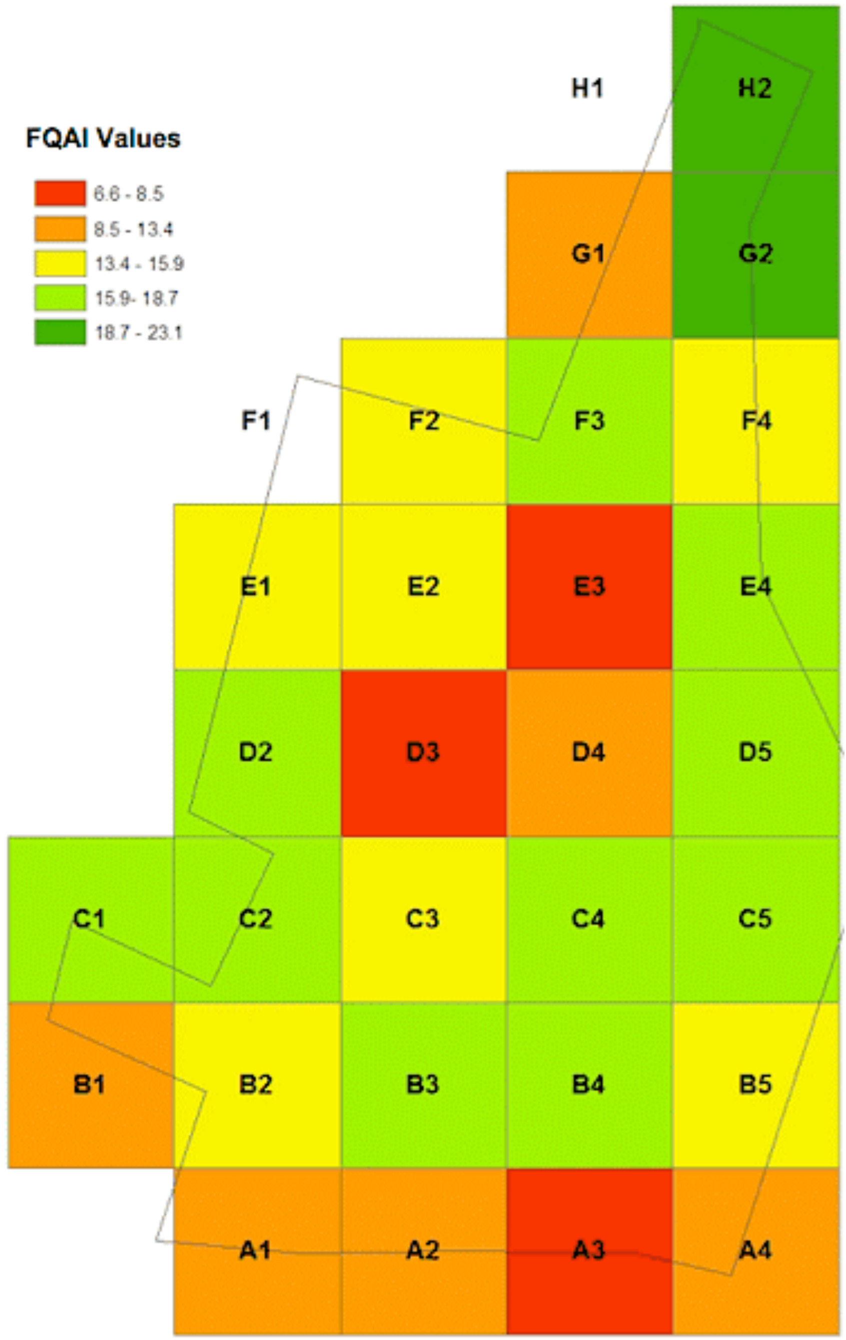

Figure 6: Plant species diversity score for each 100 m × 100 m Teaneck Creek sampling unit as measured by the Simpson Diversity Index.