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Issue 8
Spring/Summer 2003

The
Kentucky Institute
for the
Environment
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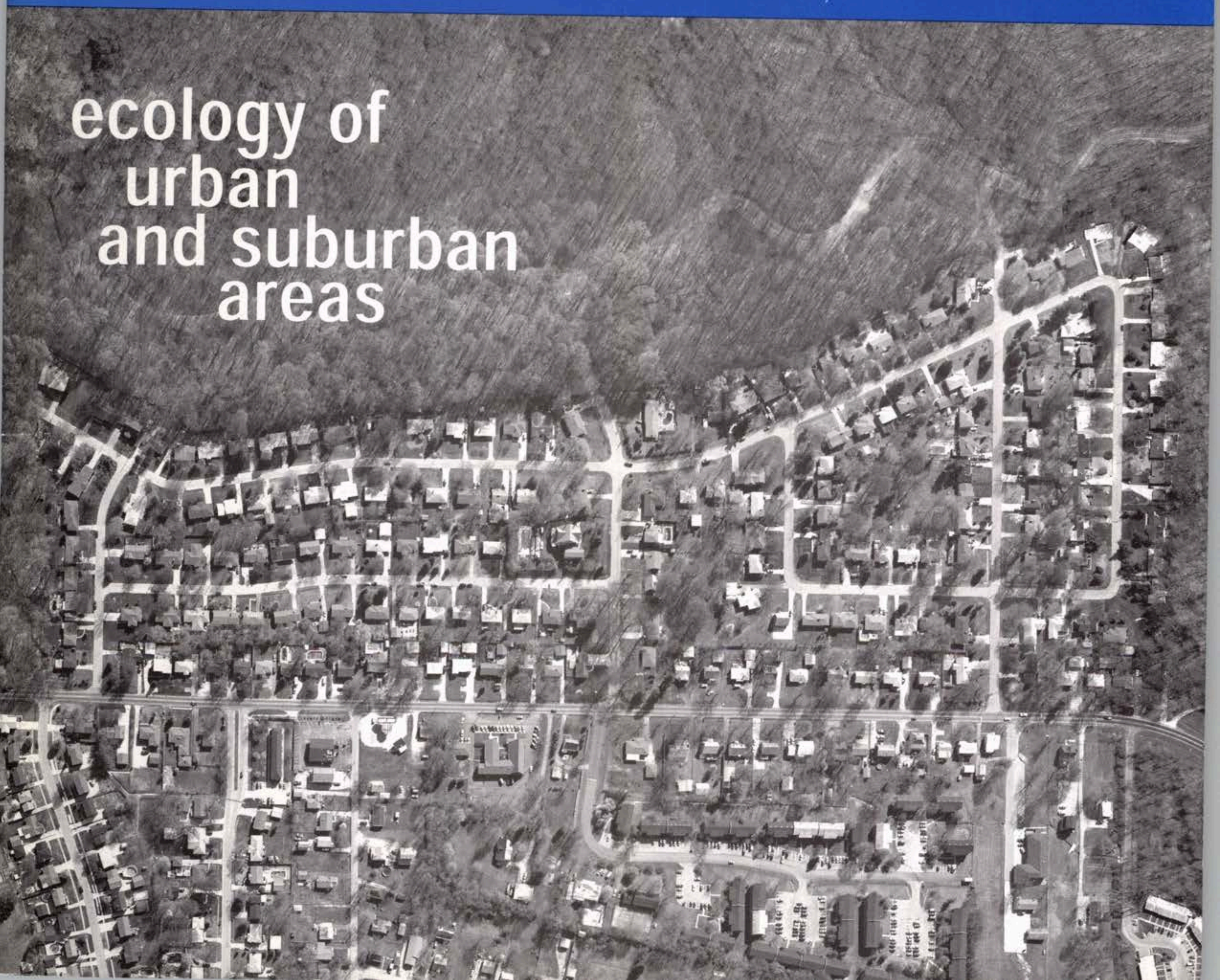

ecology of urban and suburban areas

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The Kentucky Institute for the Environment and Sustainable Development (KIESD) was created in July 1992 within the Office of the Vice President for Research, University of Louisville. The Institute provides a forum to conduct interdisciplinary research, applied scholarly analysis, pubic service and educational outreach on environmental and sustainable development issues at the local, state, national and international levels.

KIESD is comprised of eight thematic program centers: Environmental Education, Watershed Research, Environmental Law, Sustainable Urban Neighborhoods, Pollution Prevention, Environmental and Occupational Health Sciences, Environmental Policy and Management, and Environmental Engineering.

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UNIVERSITY of LOUISVILLE
dare to be great

Cover Photo: The orthophotographic imagery was created from Spring 2000 aerial photography at a scale of 1"=660' which was scanned and rectified to existing features and terrain elevation data from the LOJIC digital planimetric and topographic map database.

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The Excitement of Environmental Discovery

As dean of the College of Arts and Sciences, the University of Louisville' s largest academic unit, serving over 7,000 undergraduate majors and about 1,500 graduate students at both the masters and Ph.D. levels, I am at times awed at the responsibility of the College in exerting a significant impact throughout this academic community. Indeed, even beyond the students actually enrolled in the programs of the College, our faculty teach virtually all undergraduate students through our primary role in the university's general education curriculum. Nowhere is that responsibility more profound than in the opportunity the College provides to draw in every student, both undergraduate and graduate, to the excitement of research. If our students are to be successful in the world of work, they need to understand the importance of innovation and creative discovery, as well as to embrace and seek out the excitement of the research process.

Clearly, the natural environment as the vehicle for this journey of excitement has been shown to be both viable and attractive to students. Field research provides opportunities for students to apply basic knowledge and theory gained in the classroom to real world problems. Each of the projects described in the papers of this issue of Sustain has benefited from the engagement of undergraduate and graduate students.

The University has a long history of research on the environment dating back to some of the earliest scientific studies of the Ohio River. Recent additions to the faculty, particularly in the Department of Biology and in the Department of Geography and Geosciences, have reinforced our commitment to environmental research by expanding the breadth of work, which now includes diverse ecosystems ranging form small streams to large rivers as well as their surrounding forests. The Kentucky Institute for the Environment and Sustainable Development serves an important role by linking environmental researchers across academic disciplines and enabling them to address issues from a broad perspective that considers the needs of humans and ecosystems.

The impact of this research enterprise reaches beyond the campuses of this institution. As a fact of local economic impact, faculty involved in field research not only contribute to scientific advancement in their respective fields, but also provide important information to local planners who are charged with managing the natural resources of Kentucky and the region. The papers in this issue of Sustain showcase cooperative efforts between university scholars and the Bernheim Arboretum and Research Forest, the Jefferson County Memorial Forest, Metro Parks of Louisville, and the Metropolitan Sanitary District. Publication of research findings is not an end in itself, but the beginning of a dialogue that leads to more informed planning and management decisions.

Serving as Guest Editor for this issue of Sustain is Dr. Mark McDonnell who is an internationally recognized expert on the ecology of urban and suburban environments. His participation in this effort was brought about in part by the Liberal Studies Project of the College of Arts & Sciences, which is supported by a generous grant allowing us to invite scholars of his renown for extended visits to our campus. This issue is a clear reflection of the benefits which can arise through collaboration among international scholars, community agencies, and concerned citizens, all of whom care deeply about a sustainable environment.

James F. Brennan

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An Introduction to the Management and Restoration of Urban and Suburban Natural Areas

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Introduction

The preservation, restoration and ecologically sound management of urban and suburban natural areas are crucial to the maintenance of global, regional and local biodiversity. Throughout the world the number and size of urban areas are increasing (United Nations Centre for Human Settlements 1996, Cohen 1995). It has been predicted that by the year 2025 over 60% of the world's population will be living in urban settlements (Smith 1997). Unfortunately, our ecological understanding of the world has been obtained from studying natural areas specifically selected to minimize the presence of humans (Botkin 1990, McDonnell and Pickett 1993, Pickett et al. 2001). This situation has begun to change over the last decade as ecologists around the world have gained a new appreciation for the importance of ecological research in urban and suburban environments (Pickett et al. 2001, Paul and Meyers 2001). The research conducted in the greater Louisville metropolitan area described in this issue of *Sustain* provides important information for the ecologically sound management of the region's valuable natural resources.

Urban natural areas are ecosystems that persist due primarily to natural processes with minimal human intervention. In addition to their intrinsic value, they are especially important for providing unique examples of pre-urbanized ecosystems at a local scale and are refugia for indigenous plants and animals. This makes them valuable sources of propagules and organisms for future restoration projects. Patches of indigenous vegetation located in cities and towns are also vital living laboratories for environmental education providing many urban dwellers their only opportunity to experience and learn about the region's natural heritage. Urban and suburban natural areas in Kentucky and throughout the world are facing many serious problems that threaten their persistence (Dale et al. 2000, See also Carreiro this issue). The conservation, management and restoration of these valuable areas present a formidable challenge to Louisville and other cities around the world.

In this paper I will discuss a variety of topics related to the ecologically sound management and restoration of urban and suburban natural areas including: 1) the identification of achievable management and restoration goals and objectives; 2) the constraints of managing and restoring remnant patches of native vegetation in urban and suburban landscapes; 3) the need to view these patches as complex systems as opposed to a collection of things; 4) the necessity of using adaptive management and restoration techniques; and 5) techniques for monitoring vegetation. In the final section, I describe what we have learned from a long-term urban forest research and management project located in one of the oldest and densely populated cities in the United States, New York, New York.

Determining Goals and Objectives

The success of maintaining and restoring natural areas in urban and suburban environments depends upon having clear achievable goals. The goals and objectives of management and restoration projects in urban and suburban natural areas can address a variety of topics ranging in degree of difficulty and costs (Sauer, 1998, Lehvävirta and Rita 2002). These include maintaining, altering or restoring: 1) the physical conditions of a site; 2) plant community composition and structure; and 3) ecosystem functions such as nutrient cycling. The restoration of the physical conditions of a site should be one of the first objectives of any restoration project. Such projects might involve stabilizing an eroding slope or improving water quality.

Other goals may involve the establishment and persistence of certain species, populations and communities of plants. Due to the fact that individual organisms, populations and communities are dynamic in both time and space, it is more difficult to measure the success of these projects in relation to some ideal condition such as an undisturbed plant community. Alternatively, project goals could simply involve increasing the abundance of native species and reducing the cover of weeds, or increasing the number of individuals in a population of rare or uncommon plants. In the following sections, I discuss a number of issues that need to be considered when developing management and restoration strategies.

Urban Constraints to Management and Restoration

Urban constraints are imposed by the existing conditions within a natural area or site and the area surrounding the site (i.e., landscape context). The size and shape of a site will determine the amount of edge versus core interior habitat (Forman and Godron 1986, Forman 1995). Typically, urban natural areas in the center of cities are small to moderate size ecosystems with minimal core interior habitat. Conversely, these systems tend to have a large amount of boundary edges resulting in numerous categories of physical and ecological "edge effects." These "edge effects" can have both positive and negative impacts on ecological systems (Hansen and di Castri 1992, Matlack, 1993). Thus, management and restoration strategies need to reduce negative edge effects such as the invasion of non-native species while favoring those aspects of the edge that lessen harmful impacts on the site. For example, Cadenasso and Pickett (2000) found that the existence of a dense cover of vegetation at the edge of a forest reduced the input of seeds of non-native species.

The creation of a city also produces severe landscape fragmentation that results in isolating the remaining natural areas from similar habitats (Collinge 1996). This isolation can limit the maximum biological diversity that can be maintained at a site (Forman 1995), and may significantly constrain future management and restoration efforts. Patches of native vegetation in urban and suburban environments may be isolated from patches of similar vegetation, but they are typically close to a large number and diversity of potential weed invaders used in surrounding gardens and parks. One of the greatest challenges to managing and restoring remnant patches of native vegetation in urban and suburban environments is the control of invasive plant and animal species (Gilbert 1989, Soule¢1990).

In addition, the building of the city around remnant patches of natural areas can adversely influence and hydrologic processes that could also limit potential management and restoration goals. For example, if the flow of water through a river has been significantly reduced or is of poor quality, it may no longer be possible for some of the dominant organisms to sustain their population numbers (Paul and Meyer 2001, see Jack et al. this issue). Conversely, a remnant native forest may receive nutrient additions and other pollutants directly from adjacent developments or through air pollution (see Carreiro this issue) changing site conditions that may favor the invasion of non-native species causing the loss of native plants and animals.

The New York Botanical Garden Forest with the New York City skyline in the background. Note the Enid Haup Conservatory in the left of the picture.

In contrast to the relatively static landscape of inner cities, the dynamic landscapes of newly developed suburban areas where land conversion from agricultural or native plant communities to suburbs has only recently occurred, the size, shape and isolation constraints on management and restoration may not be as immediately severe. These remnant patches of native vegetation may experience all or some of the issues listed above, and in addition they may undergo transitional dynamics as the populations, communities and ecosystems adjust to the newly modified landscapes. Management and restoration efforts in these landscapes could be significantly enhanced by maximizing the size of the remnant patches that are set aside and by enhancing landscape connectivity between patches of native vegetation (Bennett 1999, Pirnat 2000). A realistic understanding of the constraints on managing and restoring patches of remnant vegetation located in urban and suburban environments will assist in developing achievable project goals and objectives.

Manage and Restore Systems Not 'Things'

Typically, natural areas in cities exhibit a variety of environmental conditions and a diversity of organisms. Plant and animal communities that are to be managed and restored exist due to multiple interactions between environmental conditions and living organisms. These components and their interactions form a dynamic ecological system (ecosystem) that is affected by the types, age and abundance of organisms and the changes in environmental conditions over time. For example, plants growing in shallow nutrient poor ridges are adapted to low nutrient conditions and can often outcompete exotic species in undisturbed habitats, but may be poor at resisting exotic invasions when soils become more

A trail through The New York Botanical Forest showing the railings used to reduce trampling impacts on the forest.

fertile due to human activities such as air pollution (see Carreiro this issue). Thus, effective management and restoration projects in urban and suburban natural areas must consider the past land-use history of the site, the current inputs of pollutants, and the nature and frequency of disturbance regimes. An understanding of the land use history is important because previous activities on the site may have significantly altered a variety of conditions such as soil fertility, hydrology, and the abundance of native and weed species. Each of these factors can directly or indirectly effect the ability to achieve restoration and management goals.

In addition, it is vital to have an understanding of the biological and ecological requirements of the plants and animals currently inhabiting a site as well as those that could be added to a site during the project. Because ecological systems are dynamic, it is important that the grand goals of the management and restoration projects focus on the systems and not just things (i.e. a single rare species or just the canopy trees) while realizing that evaluation schemes only measure 'things'. Even though the project goals should be focused on maintaining systems, many times we use single populations or suites of species to monitor the system's conditions (see Bukaveckas and McCandless this issue).

Although often overlooked, effective management of urban and suburban patches of remnant vegetation must make every effort to maintain natural disturbance regimes (see Bukaveckas and McCandless this issue) while also limiting the frequency and magnitude of new anthropogenic disturbances. For example, natural areas in cities tend to experience increased trampling impacts, premature death of plants due to vandalism and picking, aggressive weed invasion, arson lit fires and flooding to name a few. The impacts of these new disturbance regimes can be significantly reduced by appropriate management techniques.

The Role of Adaptive Management

Dynamic ecological systems require adaptive management and restoration strategies that rely on a close connection between the management and restoration efforts, a monitoring program, and the modification of methods and techniques in light of new knowledge (Pastorok *et al.* 1997, Hobbs and Norton 1996). This process is called adaptive management (Pastorok *et al.* 1997). Through the development of a regular monitoring program it is possible to adjust methods and techniques to maximize short- and long-term success of management and restoration projects. In addition, information gained from monitoring may, in some instances, alter the goals and objectives of a project. For example, the lack of success in establishing certain key species of trees may force the project to reconsider the composition of the target plant community. An effective adaptive management system should involve a variety of management and restoration techniques in order to develop a greater understanding of the dynamics of your system. From a scientific point of view, it is always useful to leave some areas unmanaged to serve as a reference point or control. Implementation of an adaptive management strategy requires not only post-treatment assessment, but also good record keeping of the methods and techniques used to manage and restore a site.

Techniques for Monitoring Vegetation

Vegetation monitoring falls into two broad categories: qualitative and quantitative methods. The appropriateness of each method is determined by the goals of the project, the knowledge and skills of the staff and the availability of time and money resources available for monitoring. The advantages and disadvantages of each approach are presented below.

Qualitatively monitoring vegetation involves such techniques as setting up photographic stations and taking pictures over time, or writing down notes on the condition of the vegetation at regular intervals. These exercises are not very time consuming and can provide a useful record of the response of the system to the management or restoration efforts. Another advantage is they require a minimum of staff training and are relatively inexpensive. The disadvantage of qualitative monitoring is that it is very difficult to pin down specifically what has changed in the plant community over time, especially over short periods such as a few months or years. It is possible to say the trees have grown or there is more ground cover, but it is virtually impossible to describe changes to individuals or populations of plants.

Quantitative monitoring of vegetation by definition requires the measurement of the amount, size or distribution of organisms and can involve quantifying the presence, abundance, size class distribution, structure and distribution of plants at a site. These types of measurements can be used over time to develop useful criteria for evaluating the success of a project such as the survivorship of plants or changes in plant community composition and structure.

The fields of ecology or forestry, and more specifically vegetation science, provide many methods and tools for quantifying individual plants, populations and plant communities (Kent and Coker 1992). It is beyond the scope of this paper to provide a universal prescription for quantitatively monitoring vegetation. I would encourage those interested in starting a monitoring program to look at the books cited above, take courses that cover vegetation sampling or find local experts who can help develop a method of monitoring consistent with the objectives and resources. It is especially important when designing a monitoring program that the products of the monitoring fit with the monitoring objectives as defined by the adaptive management program and available resources.

A permanent monitoring plot being prepared to be resampled in the NYBG Forest.

A Case Study

There is a 16 ha (40 acre) mixed hardwood – hemlock forest located in the center of the New York Botanical Garden, Bronx, New York. It represents the last remnant of the original forest that once covered New York City. Unlike other patches of forest in New York City, this patch has never been cleared for timber or significantly altered (McDonnell 1988, Rudnicky and McDonnell 1989). In the early 1980's, I had the opportunity to direct a project that studied the structure and function of this natural forest surrounded by intense urbanization, while also using it as a living laboratory to develop comprehensive management and restoration strategies for managing and restoring urban natural areas (McDonnell 1988). Although the level of research and monitoring conducted in the forest is beyond the resources of many municipalities, it provides an indication of the types of environmental monitoring that are possible when the expertise and resources are available.

Research efforts in the 1980's focused on developing baseline information on the plants, animals, and soils of the forest (McDonnell 1988). A historical reconstruction of the forest from vegetation survey maps provided insights into how the forest canopy changed between the mid 1930's and the mid 1980's (Rudnicky and McDonnell 1989). In addition, a long-term study to monitor future changes in the vegetation composition and structure of the forest was begun in 1986. Permanent forest reference plots were established in which the location of every tree was mapped; trees were tagged, and their condition evaluated. In addition, comparable data were collected for shrub and herb layers. Yearly census of the number of trees that die has provided one of the first estimates of tree mortality rates for forests in urban areas. One of the major goals of the vegetation studies in the forest was to develop a mechanistic understanding of the factors affecting tree regeneration. Important factors that we studied included soil seed banks, seedling demography, and the use of artificial seedlings to assess the impact of physical damage to seedlings. Research was also conducted on the physical, biological and chemical properties of the soil in the forest, (Pouyat and McDonnell 1991, Pouyat *et al.* 1994, Pouyat et al. 1995) and data have been collected on rates of forest floor decomposition and N mineralization rates (White and McDonnell 1988, Pouyat *et al.* 1995, Pouyat et al. 1997, McDonnell et al. 1997). The results of these studies provided value information as to pollution levels and the functioning of critical ecosystem processes such as decomposition and nutrient cycling.

The management objective for the forest was to maintain it as a hardwood-hemlock forest ecosystem in a state that is within the range of variability exhibited by similar forests in the region not exposed to urbanization (McDonnell 1988). The management goal is simply to promote the reproduction and growth of native species while minimizing changes caused by human activity. Over the past two decades significant progress has been made in reducing the negative impacts on this forest ecosystem.

Because the forest is a living system that is exposed to the extremes of weather as well as human impacts such as vandalism and trampling, it was important that appropriate trails, railings, benches, and signs to facilitate public use and access to the forest be developed and maintained. Our experience indicates that the best means of reducing vandalism and other human impacts is to repair quickly any damage that has occurred. Thus, in an urban environment considerable effort has to be expended in maintaining areas that have already been restored. In addition, we also learned it was critical to maintain a daily presence of a ranger in the forest whose role involved both education and stewardship.

Fifty years of unchecked erosion in the forest produced numerous areas that were devoid of vegetation. Using the information from the vegetation analysis as a template, the severely trampled areas of the forest were revegetated using native trees and shrubs. Small non-native trees and shrubs have been selectively removed by hand and a program was started to ringbark or girdle the large non-native trees which would result in their death, but it would not disturb the soil and forest floor which we felt would encourage non-native species invasions. In the eastern deciduous forest of the U.S., dead standing wood is important to the maintenance of insect communities that provide a food source for interior forest birds and animals. In the late 1980's, a program was pioneered to remove the crown of dead trees to reduce the load on the roots while leaving the main stem standing. As in all public parks, there is a trade off between ecologically sound practices and public safety.

Although the funding to study and manage this forest has declined over the years and I have moved on to another position, the forest is still intact and every effort has been made to follow the original management objectives presented above. One disturbance, the loss of virtually all of the forest's hemlocks, some of which were over 200 years old, due to the invasion of the aphid-like woolly adelgid (*Adelges tsugae*) (Onken et al. 2002), has significantly changed the structure of the forest, which is currently in a recovery phase.

In conclusion, after 20 years of experience with this unique forest in the Bronx, I propose the following general guidelines for managing urban natural areas to reduce human impacts and encourage natural ecosystem processes: 1) the integrity of the native patches of vegetation should be maintained by minimizing existing trails and preventing the creation of new trails, 2) the disruption of nutrient cycling processes should be prevented by keeping dead wood and organic material in the system, 3) the inputs of harmful chemicals, energy, water and organisms should be reduced when possible, 4) appropriate management of edges should be promoted to reduce impacts on patch interiors and, 5) native genetic diversity should be maintained by propagating and using plants from within the forest and the nearby natural landscape.

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Biographical Information

Dr. McDonnell received his doctorate from Rutgers University in 1983 and went on to work at The New York Botanical Garden's Institute of Ecosystem Studies as a research scientist for 10 years. From1993 until 1998, he was an Associate Professor in the Department of Ecology and Evolutionary Biology at the University of Connecticut were he also served as Director of The Connecticut State Arboretum. Dr. Mark McDonnell is currently the Director of the Australian Research Centre for Urban Ecology, a Division of the Royal Botanic Gardens Melbourne. In addition, he is an Associate Professor in the School of Botany, University of Melbourne. His interests range widely, and include the processes driving vegetation change, invasion of non-indigenous plants, landscape ecology, and the conservation and restoration of urban and suburban natural areas.

Recovery of a Suburban Forest in the Aftermath of a Tornado

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Center for Watershed Research and
Department of Biology
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Introduction

Jefferson County Memorial Forest (JCMF) is thought to be one of the largest forests in an urban county in the country. The forest covers over 5,000 acres (20 km^2) and is situated largely within the Louisville metropolitan area, only 20 minutes from the city center (see map: Figure 1). Throughout its history, the forest has been subjected to human impacts of varying cause and intensity. The property was logged prior to establishment of the preserve in 1948. It is currently used for recreational and educational programs and logging is no longer permitted. While direct human impacts have largely been mitigated through management, the forest's proximity to urban areas gives rise to stressors associated with air quality and heat island effects (see related article by Carreiro in this issue). These stress factors act in concert with the natural processes of forest vegetation dynamics and soil nutrient cycling to determine the direction and rate of change in the forest ecosystem. Forests change slowly owing to the long-lived tree species that dominate the ecosystem. Occasionally, this slow succession of change is disturbed by a discrete event that can dramatically alter the present and future appearance of the forest. Logging and intentional fires are examples of disturbances caused by humans whereas flooding, blowdowns and fires caused by lightning are natural disturbances that have shaped forest ecosystems throughout their history (Pickett and White, 1985). On May 28, 1996 a tornado struck a portion of the Jefferson County Memorial Forest with winds of 190-215 mph (305-345 km h^{-1}). This event provided an opportunity to document for the first time the effects of this type of disturbance on nutrient cycling for a forest in an urban setting. In this article, we provide a brief review of the processes by which disturbance effects forests and then present data on the response and recovery of a blowdown site at JCMF.

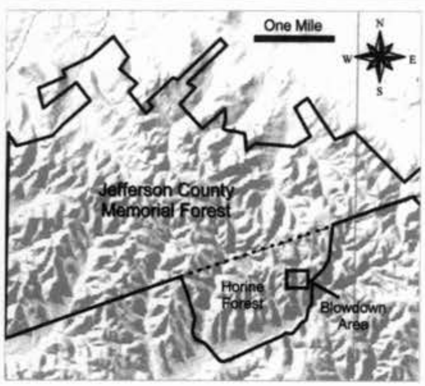

Figure 1. (Left) Map of the Jefferson County Memorial Forest (JCMF) showing location of the forest disturbance research site. (Right) Location of JCMF in proximity of the Louisville Metropolitan area.

Disturbance Effects on Forests

Forest ecologists have long been interested in the role that disturbance plays in altering plant-soil nutrient cycling. This interest stems from the concern that disturbance accelerates nutrient loss from forests and thereby diminishes their long-term productivity. Studies of nutrient cycling have largely focused on nitrogen because it is most often the element which limits tree growth and because nitrogen losses after disturbance often exceed those of other plant nutrients (Likens et al., 1977; Vitousek et al., 1982). Nitrogen loss from terrestrial ecosystems also has important implications for nutrient accumulation in downstream waterbodies such as lakes, rivers and estuaries (Fenn et al., 1998). Disturbance results in the full or partial loss of the forest canopy and this in turn leads to changes in the forest microclimate. Higher levels of solar radiation reaching the forest floor elevate soil temperature (Figure 2). Soil moisture content often increases since the canopy is no longer present to intercept rainfall and evapotranspiration through tree leaves is reduced. The combined

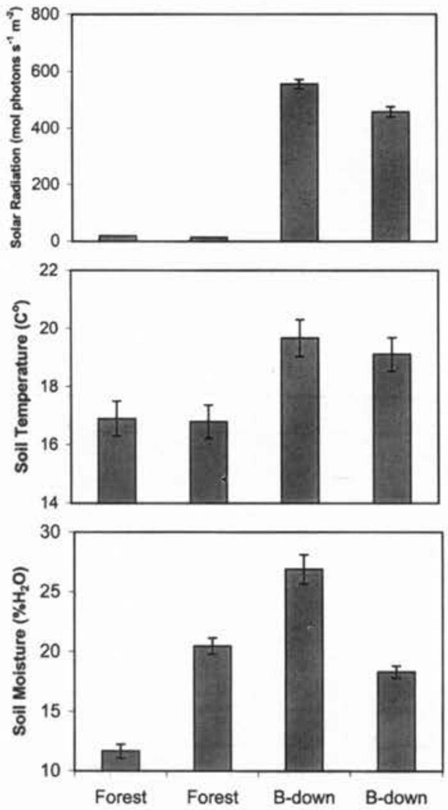

Figure 2. Solar radiation, soil temperature and soil moisture at 4 study plots located in undisturbed forest and blowdown areas of the Jefferson County Memorial Forest.

effects of warmer and wetter soil promote decomposition of organic matter that has accumulated on the forest floor as decaying leaves and other vegetative matter. Decomposition is the process by which nutrients stored in decaying organic matter are released and made available for plant uptake. In most forests, this is the dominant source of nutrient supply since external inputs from rainfall are usually small in comparison (Likens et al., 1977).

While disturbance is thought to stimulate decomposition, the loss of living vegetation results in lower nutrient uptake by plant roots. Since supply is out of balance with demand, nutrients accumulate in soil solution until rainfall washes them downstream. This mechanism, termed leaching, is the dominant process of nutrient loss from disturbed forests although other factors may also contribute (Table 1). Nitrogen is lost in the form of nitrate (NO_3), ammonium (NH_4) and dissolved organic nitrogen (DON) such that these solutes are often elevated in water draining from disturbed forests (Weston and Attiwill, 1996). As plants re-colonize disturbed areas, nutrient demand increases and leaching losses are diminished. The length of time required for plant communities to re-establish and facilitate nutrient retention depends on the nature and severity of the disturbance and the mass of nitrogen that has been lost from the soil (Pardo et al., 1995; Holmes and Zak, 1999). Species colonizing disturbed areas are often plants that can tolerate low nitrogen availability. Conditions of nitrogen stress may persist for many years after the disturbance while inputs from rainfall gradually replenish soil nitrogen levels.

Studies of disturbance effects on forest nutrient cycles have been conducted in various locations around the world but these have focused almost exclusively on forests which are remote from stress factors associated with proximity to urban centers. An exception is the study by McDonnell et al. (1993) documenting differences in decomposition rates, nitrogen dynamics and soil biota in forests along an urban to rural gradient emanating from the New York metropolitan area. Their research was among the first to demonstrate human effects on ecosystem processes for forests in urban settings (see also Pouyat et al., 1997; Pouyat and Carreiro, 2003).

A Disturbance Study at JCMF

Studies of disturbance effects on forest nutrient losses have largely focused on fire or harvesting effects. Studies of blowdown effects are rare and most have investigated vegetation responses rather than nutrient cycles (Peterson and Pickett, 1995). In 1997, one year after a tornado struck JCMF, we initiated a study to assess nutrient retention and loss from the blowdown site. We anticipated that if the disturbance had resulted in substantive nitrogen loss, the recovering for-

TYPE OF DISTURBANCE	MECHANISM OF NUTRIENT LOSS	RECOVERY TIME
BLOWDOWN	Leaching from soil	Short: re-sprouting from damaged trees; minimal soil disturbance.
FIRE	Combustion of organic matter	Moderate-Long: Leaching from soil dependent on intensity of fire and tree mortality.
LOGGING	Harvesting of organic matter Erosion Leaching from soil	Moderate-Long: dependent on harvest (selective vs. clearcut); high soil disturbance.

Table I. Effects of disturbance on nutrient loss and recovery from forested ecosystems.

Photo 1. Tornado impacts at Jefferson County Memorial Forest. (Left) Upper portion of the study catchment showing intact forest with mature canopy and minimal understory vegetation. (Middle) Transition between intact forest (upper slope) and blowdown area (lower slope). (Right) Disturbed forest lacks large trees except for standing dead timber. Forest consists of small saplings growing in high density.

est would exhibit signs of nitrogen stress. Conditions of low nitrogen availability would be reflected in high nitrogen retention within soils and low nitrogen content in plant tissues. Our study site was a catchment of approximately 185 acres (75 ha) in the Horine section of JCMF. The catchment is a south-facing slope bisected by a one-mile long first order stream. The stream flows continuously from February through July but only sporadically at other times. Approximately one quarter (50 acres) of the subcatchment was deforested by the tornado. Within the disturbed area there were no mature trees left intact although some larger trees remained standing as stumps that had been topped (See Photo 1). Our study was conducted between May and December of 1997 at which time the vegetation was dominated by saplings of less than one inch (2.54 cm) diameter and a number of herbaceous species that included Pale Jewelweed (*Impatiens pallida*), Spotted Jewelweed (*I. Capensis*), Pokeweed (*Phytolacca americana*) and White snakeroot (*Eupatorium rugosum*). The intact portion of the forest was dominated by northern hardwood species that included Sugar Maple (*Acer saccharum*), American Beech (*Fagus grandifolia*), Slippery Elm (*Ulmus rubra*) and Yellow Poplar (*Liriodendrum tulipifera*). Four study plots were established: two in the undisturbed forest and two in the blowdown. Drainage waters were collected at each of the four sites by means of collectors installed at various depths in the soil profile. Soil solution collectors were PVC pipes that had been cut lengthwise and buried in the soil to intercept rainwater moving through the soil profile. A drainage tube from the collector fed a small sampling bottle which stored water until collection (within 1-2 days after each rain event). Soilwater was analyzed for various fractions of dissolved nitrogen (NO_3, NH_4, DON) at the University of Louisville's Environmental Analyses Laboratory. The mass of nitrogen lost from the soil was calculated by multiplying the concentration of dissolved nitrogen by the volume of water collected.

Disturbance Effects at JCMF

Comparison of soil losses supported our hypothesis that blowdown areas were retaining more nitrogen than plots in the undisturbed forest. Average nitrogen losses over five rain events were lower at the two disturbed plots in comparison to the intact forest (Figure 3). Total nitrogen losses were nearly twofold lower from plots within the blowdown area. Differences in nitrogen loss were largely due to elevated nitrate concentrations in soilwater collected from forested plots (ammonium and dissolved organic nitrogen concentrations were generally similar). A comparison of nitrogen inputs from rainfall did not reveal any consistent differences among disturbed and undisturbed sites. Since nitrogen inputs to all plots were similar, lower losses at blowdown sites suggest that nitrogen was being retained more effectively at sites subjected to disturbance. High nitrogen retention is indicative of low nitrogen availability as might be expected if the disturbance had resulted in nitrogen loss. To further test this hypothesis we collected samples of the top 2 inches (5 cm) of the soil layer from disturbed and undisturbed plots. A subset of these samples was analyzed immediately to determine their nitrogen content while remaining samples were transferred to polyethylene bags. The bags were perforated to allow water to pass through and then buried below the

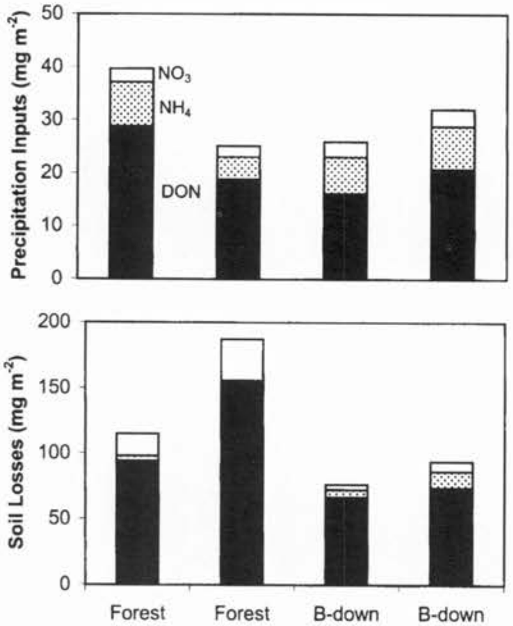

Figure 3. Dissolved nitrogen fractions in rainwater inputs (top panel) and soilwater losses (bottom panel) for undisturbed forest and blowdown areas at JCMF. Measured fractions include: nitrate (NO_3), ammonium (NH_4) and dissolved organic nitrogen (DON). Data are expressed as fluxes per unit area based on measured concentrations and volume of water collected. Differences in nitrate and total dissolved nitrogen fluxes between disturbed and intact forest plots were statistically significantly (one-way ANOVA; $p<.05$).

litter layer for a 28-day incubation period. At the conclusion of this period, the bags were retrieved and analyzed to determine the change in their nitrogen status. We found that soils collected from blowdown areas typically had a lower starting nitrogen content but showed a greater increase in nitrogen during the period of incubation (Figure 4). This finding showed that nitrogen was accumulating more rapidly at sites subjected to disturbance and was consistent with observed differences in nitrogen retention between disturbed and undisturbed sites.

Analyses of soil nutrient dynamics suggested that blowdown sites were experiencing nitrogen deficiency. This hypothesis was further supported by a comparison of plant tissue nitrogen levels in undisturbed and blowdown areas. Samples collected from randomly selected herbaceous plants were analyzed to determine their nitrogen content relative to carbon. Plants growing in the blowdown area were found to be highly deficient in nitrogen which accounted for only 0.6% of their biomass (Figure 5). By comparison, plants from the undisturbed sites averaged almost three times as much nitrogen (1.5%) in their tissues. This finding suggests that soil conditions in the disturbed forest are more severely nitrogen limited and favor growth by plant species that are capable of growing at low nitrogen availability.

Recovery at JCMF

Previous studies of forest nutrient dynamics have shown that immediately after a disturbance large amounts of nitrogen may be lost through leaching from soils (Likens et al., 1977). This initial pulse of nitrogen loss depletes soil reserves and results in lower nitrogen availability to support plant growth. As plant communities recover, root uptake of nitrogen increases demand and reduces leaching losses. Our study conducted one year after a tornado struck JCMF suggests that disturbance resulted in significant nitrogen depletion within the damaged forest, but that the intervening growing season allowed sufficient recovery time for plant communities to re-establish control over nitrogen loss. The rapid recovery of the plant community was aided by re-sprouting of fallen trees. New shoots were able to take advantage of surviving portions of existing root systems and exhibited rapid growth rates (See Photo 2). Though long-term survival of damaged trees may be low, their short-term contribution toward minimizing further nitrogen loss from the forest may be an important transition phase. In addition, herbaceous annuals were able to take advantage of the high light conditions resulting from the loss of the overstory trees and accounted for a large fraction of the plant abundance (>40 stems per m^2). Their ability to grow under low nitrogen conditions facilitates nitrogen retention until such time that a mature tree canopy develops. A similar pattern of response was observed by Peterson and Pickett (1995) in their study of three tornadoes that struck the Allegheny National Forest (NW PA) in 1985. Within two growing seasons, blowdown areas exhibited greater species richness, tree seedling density and plant cover than in the adjacent undamaged forest. Herba-

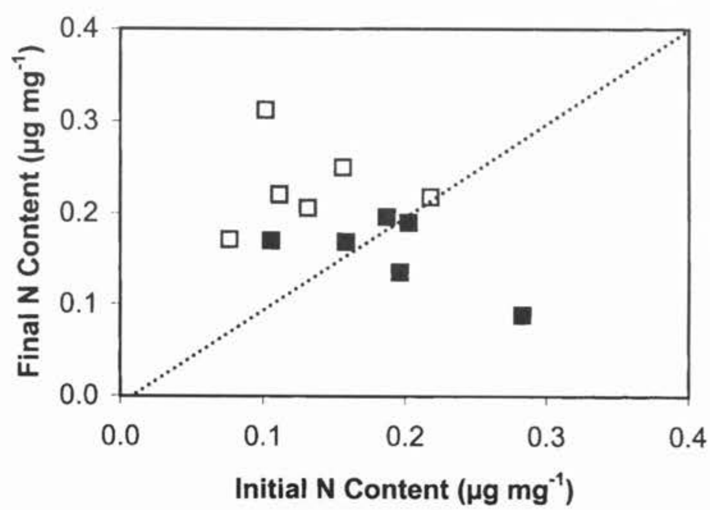

Figure 4. Intial and final nitrogen content of soil bags incubated for 28 days (Sept. 14 to Oct. 12, 1997) at undisturbed (closed symbol) and blowdown (open symbol) sites. Points above or below the 1:1 line represent bags that increased or decreased in nitrogen content (respectively). Nitrogen content expressed per mg of dried soil.

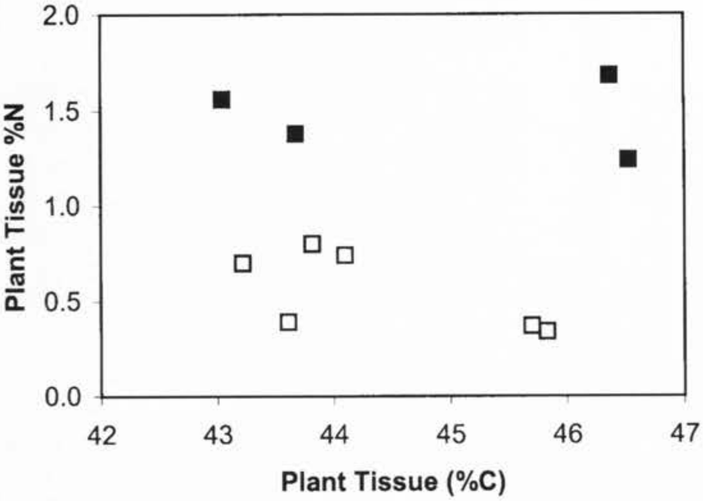

Figure 5. Plant tissue concentrations of nitrogen and carbon for herbaceous species collected from undisturbed (closed symbol) and blowdown (open symbol) forest plots. Values shown are based on 8 plants collected from each of four undisturbed sites and 3 plants collected from each of six blowdown sites. Differences in nitrogen content between forested and blowdown sites were statistically significantly (one-way ANOVA; p=.001).

ceous herbs and shrubs flourished during the first three years but had begun to decline by the sixth year when the young tree canopy became established.

Conclusions

Many types of forests depend on disturbances associated with fire, flooding and wind damage to maintain their vegetation composition. An important goal of ecology is to be able to predict the timing and direction of recovery to foster better forest management strategies. Forest regeneration is dependent upon a number of factors that include the severity and type of disturbance and the state of the community at the time of disturbance. The latter may be particularly important for forests situated in proximity to urban centers and their associated stressors (air pollution, heat island effects). Our initial studies at the site of a tornado impact have shown that vegetation recovery and nutrient dynamics of a suburban forest followed patterns similar to those observed in forests remote from urban centers. This resiliency may be attributed in part to the modest size of the damaged area relative to the size of the forest comprising JCMF. Natural areas in proximity to major urban centers are rarely contiguous and more often a mosaic of small patches distributed within an urban-suburban landscape. Fragmentation of natural areas isolates plant and animal communities and may slow post-disturbance recovery by impeding colonization of species capable of exploiting conditions found in damaged areas (Hale et al., 2001). Our results highlight the benefits of preserving natural areas at spatial scales exceeding those over

Photo 2. Disturbance and recovery at Jefferson County Memorial Forest. (Upper) Disturbed forest lacks overstory canopy resulting in a proliferation of weedy annuals and small saplings. (Lower) Forest recovery is due in part to re-sprouting from damaged and fallen trees whose roots remain intact.

which disturbances are likely to occur. Further monitoring at this site will serve to characterize forest change during regeneration and, combined with more detailed studies (see Carreiro this issue), provide a basis for assessing the effects of disturbance on forests in urban-suburban environments.

Acknowledgments

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Biographical Information

Dr. Paul Bukaveckas received his PhD from Indiana University and was a Post-doctoral Fellow at the Institute of Ecosystem Studies before joining the faculty at the University of Louisville. He is an Associate Professor in the Department of Biology and the Director of KIESD's Center for Watershed Research. His research interests focus on nutrient dynamics in terrestrial and aquatic ecosystems. He is currently involved in research projects on streams at the Bernheim Forest and large rivers throughout the Ohio Valley.

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Watershed Development and Fish Communities in Urban Streams

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Introduction

Urban land use can have significant negative effects on stream biotic communities, geomorphology and hydrology (Wang et al. 1997, 2000; see review by Paul and Meyer 2001) and there has been increased interest in the rehabilitation of urban watersheds. This interest has been driven by a number of factors such as concern for the possible negative ecological impacts of diminished biodiversity (e.g. Loreau et al. 2001) and a desire to improve the aesthetic qualities of urban stream environments. Sources of degradation in urban landscapes include increasing pavement and other impervious surfaces and the development of drainage networks for storm water (Walsh 2000). Many researchers have developed Geographic Information System (GIS) based databases in an attempt to link stream degradation with watershed characteristics such as riparian zone widths or "impervious surface." These models are often used to try to predict stream condition and prioritize stream reaches for more detailed evaluation of their ecological condition. These analyses can be extremely complex; for example, Wang et al. (2001) used a database that contained 63 different urban and other land use categories. Such sophisticated databases are time consuming and expensive to develop. Some stakeholder groups in watersheds, such as Water Watch or "Friends" groups, do not have the technical or financial resources to use such databases, even though they would be very useful to them. If a simpler GIS-based database approach could be used which was effective at predicting the condition of in-stream biotic communities, these citizen groups may be able to use such databases to help plan watershed protection activities and prioritize monitoring sites in their basins.

The Salt River basin contains Louisville, Kentucky's largest city, as well as some of the fastest growing counties in the Commonwealth (Figure 1). There are a number of groups in the basin (e.g. Salt River Watershed Watch, Kentucky Rivers Alliance) which have expressed concern about the pace of development in the formerly agricultural lands around Louisville and its impact on the streams of the watershed. However, the Salt River is a very large basin, draining 13% of Kentucky's land area and containing more than 4500 km of streams. A simple model using GIS-derived land use data to predict stream biological condition would be a very useful planning tool for these stakeholder groups. We developed a simple watershed classification system in which we calculated the percent developed land (PDL, as defined below) in 12 watersheds in the Salt River basin. We predicted that the PDL would be inversely proportional to stream quality as determined by an independently-derived measure of stream biotic integrity.

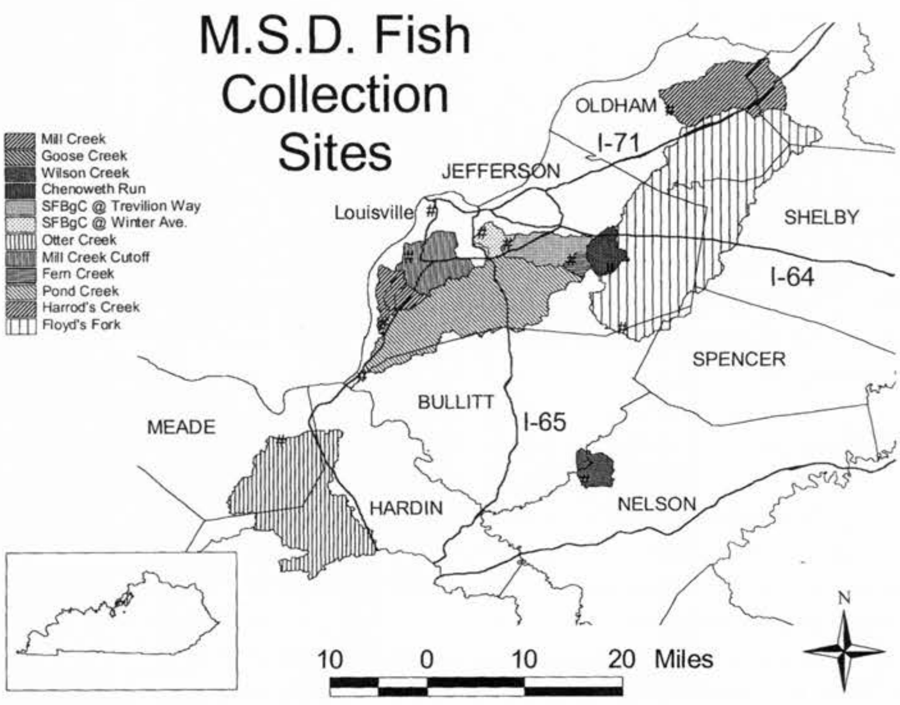

Figure 1: Location of study watersheds in the Salt River basin, Kentucky. Dark bars indicate sampling reaches within watersheds

There are a number of approaches which may be used to determine the biotic integrity of streams and validate the predictions derived from PDL in the watersheds. Bioassessment, techniques for inferring the ecological quality of a stream from the fish, invertebrates and algal communities found there, is perhaps the most common approach currently used by governmental agencies and other monitoring groups. Bioassessment has an advantage over traditional water chemistry sampling in that it tends to integrate in time and space events which may have historically affected a stream. For example, a chemical spill may travel quickly through a reach of stream and leave little trace of its passage in water samples taken after the event, but such impacts will often leave long-term clues in the structure of the stream communities. Invertebrates or fish sensitive to such a disturbance may disappear; algal biomass or diversity may drop or there may be other biological clues to the event that occurred. Many states (including Kentucky) have developed Indices of Biotic Integrity (IBIs) to quickly assess stream conditions. Such IBIs generate "scores" by using the results of several biometrics, each of which can provide information about the ecological state of the stream.

Fish communities have some particular advantages when used for bioassessments. Fish are long-lived and mobile, live and feed in a variety of ways and on a diverse group of prey, may integrate effects of lower trophic levels (such as in the familiar example of biomagnification) and they can be collected and identified fairly easily compared to some other groups in streams. Fish are also important to anglers and many other environmental interest groups so they may serve as a "charismatic" bioindicator of stream condition (Barbour et al. 1999.)

Because of these advantages, we assessed the fish communities in our study streams as our independent metric of stream condition. We then compared our calculated IBI to the PDL of each watershed to see if the predicted negative relationship between PDL and stream condition was supported.

Methods

The streams chosen were a subset of the Louisville Metropolitan Sewer District's long term monitoring stations (Figure 1). The watersheds had a wide range of land uses as determined by Geographic Information System (GIS) layers from the Commonwealth of Kentucky Office of GIS (KOGIS) (Table 1). These included forested lands, pasturage, suburban and urban land uses. The entire watershed upstream of each sampling point was used to determine the percent developed land (PDL) for the watersheds in the study. PDL includes residential, commercial and services, industrial, transportation and communication and "other urban land use" data layers as identified by KOGIS.

Stream	Watershed size (sq. km)	% PDL
Mill Creek Cutoff (MCC)	41	87
South Fork, Beargrass Creek (SFBC)	59.5	83
Beargrass Creek (BC)	43.5	82
Fern Creek (FC)	8.8	69
Chenoweth Run (CR)	30.3	55
Mill Creek (MC)	36.2	53
Pond Creek (PC)	209.7	52
Pennsylvania Run (PR)	16	33
Otter Creek (OC)	258	21
Floyd's Fork (FF)	282	14
Harrods Creek (HC)	176.7	7
Wilson Creek (WC)	18.1	4

Table 1: Stream and watershed characteristics. Streams are listed in order of decreasing percent developed land (PDL) in their watersheds. PDL includes residential, commercial and services, industrial, transportation and communication and other urban land uses as identified by the Kentucky Office of GIS.

Streams were sampled at low pool in August of 2001. We sampled the fish communities at one reach in each stream using the protocols outlined in *Methods for Assessing Biological Integrity of Surface Waters in Kentucky* (Kentucky Division of Water, 2002). Reaches were between 100 meters and 200 meters in bank length and consisted of at least two riffles, runs and pools each whenever possible. In some of the channelized or otherwise highly impacted urban streams there were only extensive bedrock sections or pooled areas with no distinguishable pool-riffle sequences. The upper and lower ends of the reach were blocked with a 0.3 cm mesh seine to prevent the escape of fish. A Smith-Root backpack electrofisher was used to sample the fish community. The electrofisher sends out pulses of electricity which temporarily stun the fish, making it easier to collect them with a dip net. The electrofisher was used in the main stream channel as well as in any important sub-habitats present such as root masses and undercut banks. A reach was considered thoroughly sampled when no new species were collected and all identified habitats had been sampled. Easily identified fish that were collected in large numbers were recorded in the field and released after voucher specimens were taken. Large specimens were also identified in the field, recorded and released. Fish for vouchers were anesthetized and preserved in the field. Fish were identified using literature from the Kentucky Division of Water (KDW) current master taxa list.

The KDW Fish Index of Biotic Integrity (IBI) is produced from a series of individual metrics (see Table 2) determined from the fish collected in the sampled streams. The IBI, an overall stream "score", is indicative of the ecological condition of the stream. This illustrates one of the strengths of IBIs in that a single low score on one metric will not necessarily cause the stream to score poorly overall. This limits the possibility that stochastic events affecting one indicator would not unduly influence the stream's over-all IBI. Scores are weighted by stream watershed area to compensate for the larger species pool expected in the larger streams.

Metric	What it measures	Predicted response to impairment
Native Species Richness (NS)	Total number of native species present in a sample	Decrease
Darter, Madtom and Sculpin Richness (DMS).	These groups are generally sensitive to pollution and are expected to decline with impairment.	Decrease
Intolerant Species Richness (INT).	Species across several taxonomic groups which are sensitive to impairment	Decrease
Water Column Species Richness (WC).	Combination of four older metrics; measures certain intolerant minnow groups, sunfish, carnivores and suckers	Decrease
Simple Lithophilic Spawning Species Richness (SL).	These species require clean gravels for spawning; sensitive to siltation	Decrease
Proportion of Insectivorous Individuals (% INST)	These taxa require healthy insect communities for their food	Decrease
Proportion of Omnivorous Individuals (% OMNI).	These are generalist feeders which are often tolerant of stream impairment	Increase
Proportion of Tolerant Individuals (% TOL).	These individuals are pollution tolerant and may invade impaired streams	Increase

Table 2: Kentucky Index of Biotic Integrity species metrics and predicted responses to degradation (adapted from *Methods for Assessing Biological Integrity of Surface Waters in Kentucky* (Kentucky Division of Water, 2002).

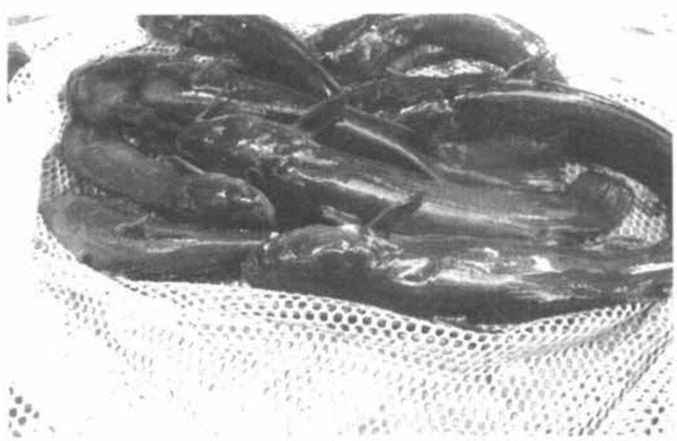

Figure 2. Some of the fish commonly collected in Salt River basin streams. From top to bottom: Redhorse, Longear sunfish, Yellow bullhead catfish. The redhorse is generally intolerant of organic pollution and sedimentation, while the bullhead is a tolerant fish often found in polluted streams in the basin.

Table Three. Metric and total IBI scores for sampled streams. Streams are listed as in Table One. See Tables One and Two for acronym keys.

Stream	NS	DMS	INT	WC	SL	%INS	%OMNI	%TOL	IBI
MCC	100	13	13	100	13	9	61	59	46
SFBC	100	8	8	11	100	5	9	88	41
BC	100	76	13	100	37	15	74	75	61
FC	100	36	37	100	37	31	71	61	59
CR	100	100	18	26	18	18	89	99	59
MC	100	15	15	100	15	10	50	41	43
PC	100	0	0	33	5	1	48	21	26
PR	100	59	27	100	52	26	100	79	68
OC	100	100	34.5	100	99.35	65.83	83.37	87.59	84
FF	100	100	21	100	100	50	50	50	71
WC	100	100	100	100	100	47	89	80	89
HC	100	100	100	100	100	76	78	72	91

Table 3: Metric and total IBI scores for sampled streams. Streams are listed as in Table One. See Tables One and Two for acronym keys.

Results

We collected a diverse fish assemblage from the streams in the study (Figure 2). The scores for the various metrics are presented in Table 3. These metrics are produced by taking the raw numbers of species or trophic groups (or raw percentages in some cases) and standardizing them into a single score for each metric. The scaling is affected by watershed size and by the composition of fish communities in good quality, "reference" streams found in the same physiographic region. Thus an individual urban stream's fish community is being indirectly compared to fish communities in good quality streams in this region of Kentucky.

As predicted, total IBI and individual metric scores were generally lowest in streams with the highest PDL and highest in the streams with the lowest PDL (Figure 3). The statistical analysis indicated that a little over 55% of the variation among sites in the Fish IBI could be explained by differences in PDL. While the overall trend did follow expectations, there were some interesting individual cases where IBI scores were lower or higher then might be predicted from the trend line (Figure 3). For example, both Mill Creek and Pond Creek scored lower than other streams with similar PDL in their watersheds, while Beargrass Creek's IBI was somewhat higher than watersheds with comparable PDLs. In case of Mill Creek and Pond Creek, these watersheds have had a long history of severe stream modifications and pollution problems (Jerry Terhune, personal communication) which may account for their lower IBI scores. The relatively high score of the Beargrass Creek site may be the result of the type of development in its watershed, which is largely residential, not industrial. There has been some effort above and below this site to restore the riparian zone, so the urban impacts may be some what reduced in this area.

Discussion

Overall, there does seem to be a fairly strong relationship between PDL in an urban watershed (as defined above) and the ecological integrity of the fish community as measured by the Kentucky Fish IBI. This is consistent with the conclusions drawn by several recent studies of the effect of urbanization on stream communities. Winter and Duthie (1998) found significant differences among periphyton and macroinvertebrate groups in urbanizing vs. rural sites in a watershed in Ontario, Canada. In the Cahaba River, several species of fish are becoming less common as a result of impacts from metropolitan Birmingham, Alabama (Onorato et al. 1998) and brown trout are much less common in the urban than in the non-urbanized portions of Valley Creek in Valley Forge, Pennsylvania (Kemp and Spotila 1997). Using the detailed GIS model described in the Introduction, Yang et al. (2001) were able to separate several components of the effects of urban land use on streams. They found the effects of urban development on the connectedness of impervious surface and on the quality of riparian buffers may be important individual drivers of stream condition above and beyond the "total impervious surface" or riparian buffer in a watershed.

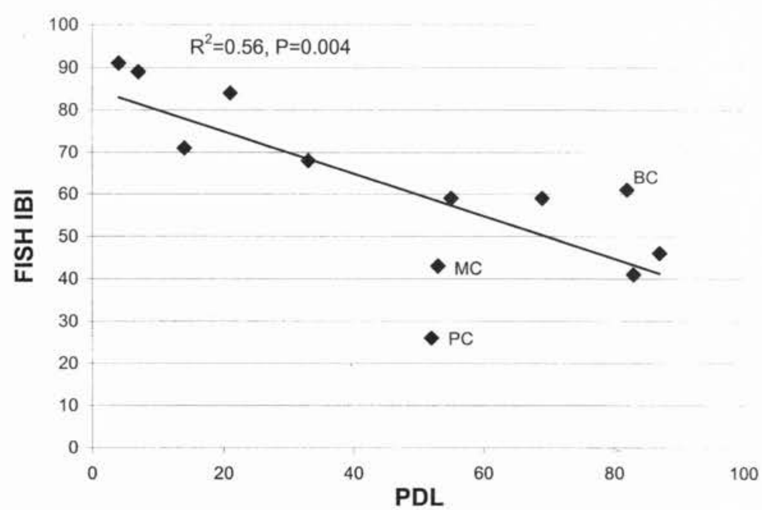

Figure 3: Fish IBI scores as a function of PDL, a measure of watershed development. Line shows best fit from linear regression. MC=Mill Creek, BC= Beargrass Creek,

However, there was a substantial proportional of the variation in the Fish IBI data (>40%) which could not be accounted for by PDL alone. There are several potential reasons for this. One weakness to our approach is the way we choose to define development; that is, combining all kinds of urban and suburban infrastructure into the PDL category. As we saw with Beargrass Creek, not all of these land uses are likely to have equivalent impacts on stream communities. This approach also pools data layers, such as industrial and residential, that more complex models keep apart as separate predictors of stream condition. We thus lost some of the potential resolution of the relationship between land use and stream condition; for example, in situations where there is the expected negative effect of development on stream biotic integrity, we are not able to attribute that effect to any one land use alone. Other studies which have been done which try to link land use with stream communities have been more precise in their distinctions among land use types or have used other variables such as impervious cover as predictors. While these approaches may provide better quantification of effects, we drew the same general conclusions about urbanization and stream biological condition in our watersheds from our less powerful but also less time and resource-intensive approach to this issue.

Another potential problem with our approach to categorizing development in the watershed is that it ignores historical effects, focusing only on current land use. This may lead in some cases to overestimates of likely stream quality given a certain level of development in a watershed. Harding et al. (1998) found in a comparative study of two North Carolina watersheds that land use in the 1950s was a better predictor of stream invertebrate and fish diversity than land use data from the 1990s. This "ghost of land use past" may be an important confounding variable in many land use studies. In this project, the lower than expected IBI scores of Pond and Mill Creeks may have been more the result of the serious historical water quality problems these streams have had in the past than the result of current conditions in their watersheds.

It is clear from our data that urbanization in the watersheds we sampled in the Salt River basin is correlated with a decrease in the ecological quality of the fish communities. GIS databases such as the one we used may be very valuable to watershed groups which are trying to protect or enhance the biological diversity of these systems. This database may be useful in economically identifying and prioritizing Salt River stream reaches for monitoring, assessment or immediate protection, preserving scarce resources for the implementation of plans to protect and improve the ecological quality of the urban streams of the basin.

There are a number of ways that local policy makers and managers can improve the quality of streams in the Salt River Basin. Improved protection of riparian zones, containment of storm water runoff and the use of watershed characteristics such as impervious cover to predict the success of instream restorations are tools that can be used to maximize the likelihood of restoration success and protect the remnant high quality areas which may still exist in some urban stream systems (Walsh, 2000). While sometimes expensive, such steps could pay a tremendous return in the form of urban streams whose quality makes them an asset to an urban community rather than a problem.

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Measuring A City's Effects on Nature

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Introduction

Never having lived west of the Hudson River before moving to Kentucky two years ago, I remember looking out the window of the airplane as it approached Standiford Airport and being struck by how green Louisville was compared with many crowded East Coast cities. It was late spring and I could see the prominent forested hills, which I know now are part of the widespread Knobs physiographic province, rising above the tree-lined streets in the southwestern part of the city. There were other large patches of green space, some consisting of lawns, others mostly treed. I immediately sensed that Louisville was a city proud of its parks with a citizenry committed to remaining connected with nature. Later I learned that this bird's-eye-view of the city had given me my first glimpse of the park and boulevard system designed by the 19th century landscape architect and visionary, Frederick Law Olmsted. I smile to think of the coincidence of starting my career as an urban ecologist conducting research inside one of his first public projects, Central Park in New York City designed in 1857, and of continuing my ecological experiments inside the urban forests that were his last creation, Louisville's Olmsted Parks, designed in the 1890's.

However, subsequent flights over Jefferson and surrounding counties also revealed that suburban development into farmland and forest appeared to be picking up speed. The 2000 U.S Census Bureau proved me right by listing Spencer County among the top ten fastest growing counties in the country (73% population growth between 1990-2000). The population in the other counties bordering Jefferson grew by 26% to 35% during that interval while Jefferson County itself only grew by 4.3%. Approximately 80% of the change in suburban county population was due to migration and not within-county births. In 1997, the total amount of developed land in the lower 48 states was 40 million hectares (98 million acres; USDA 1997 National Resources Inventory, 2000). However, much of this growth occurred over the last 50 years. Between 1960 and 1990, 12.5 million hectares (31 million acres) of rural land were converted to urban and suburban land use in the lower 48 (Dougherty, 1992). But in only five years between 1992 and 1997, an additional 4.9 million hectares (12 million acres) were developed, including 1.3 million hectares (3.2 million acres) of prime farmland (USDA 1997 National Resources Inventory, 2000), indicating that the pace of urbanization is accelerating. With perhaps a decade or two lag, trends in Kentucky and its "Golden Triangle" of Metropolitan areas, Louisville, Lexington and Cincinnati, Ohio, reflect those in the rest of the country. From 1982 to 1997, nearly 243,000 hectares (600,000 acres) of natural habitat and farmland in Kentucky were covered by cities and suburbs, with 40% of that acreage being developed between 1992 and 1997 (USDA 1997 National Resources Inventory, 2000).

Because awareness of the social, economic and environmental benefits provided by undeveloped natural ecosystems like wetlands and forests is increasing, natural areas in the Bluegrass region of Kentucky have been and are being purchased with governmental and private funds to prevent their being built upon. However, the long-term maintenance of these habitats, and of the plant, animal and microbial species they contain, is not ensured merely by posting signs declaring their existence as preserves. Saving the land from development is only a start. These habitats consist of living organisms, not museum specimens, and each species reacts to the physical, chemical and biotic conditions around them, conditions that are increasingly being altered by people both close to and far from any particular forest or field. As centers of commerce, industry and dense human settlement, cities cover soil with tall buildings and pavement, change the temperature, noise and lighting conditions around them, alter the chemistry of air and water, and attract new species from all corners of the globe either intentionally or as hitchhikers on cargo or vehicles (McKinney, 2002). In addition, road building and scattered development fragment large natural areas into smaller units that become isolated from each other. This combination of factors differentially increases or reduces the survival, growth and reproduction of plant and animal populations in remnant natural habitats within and near cities. In addition, organisms that inhabit the built environment also respond to the quilt-like pattern of different land cover types and the physical and chemical conditions that we create in cities. Some of these species, whether native or foreign, flourish within cities and suburbs and may eventually be regarded as troublesome pests (e.g., starlings, crows, Canada geese, rats, white-tailed deer, kudzu). Therefore cities and outlying suburbs exhibit within a small geographic area all of the same "global environmental change" phenomena currently disrupting ecosystems from the Amazon to the Alaskan tundra: natural habitat loss, habitat fragmentation, species extinctions, exotic species invasions, pest species irruptions, nutrient overload of aquatic and terrestrial ecosystems, atmospheric pollution, and climate warming (Vitousek et al., 1997a). Sustaining the needs of an exponentially growing human population, as well as our technologies, political systems and social choices, are the root causes of rapid environmental change at both global and local scales (Folke et al., 1997).

From an ecologist's perspective then, urban and suburban areas offer a multitude of possible questions, organisms and settings for sustaining a long-lived basic and applied research program. Yet the study of the ecology of urban areas in the United States is still, relatively speaking, in its infancy. For various reasons most American ecologists have preferred to examine the interactions between organisms and their environment in more pristine habitats far from human settlements. However, the need to provide environmental managers and regional planners with information on the causes and ecological consequences of land use change in the very places where 80% of the American people live (cities and associated suburbs) has become increasingly obvious to the scientific community (Grimm et al., 2000; McPherson et al., 1997). Ecological studies in and near cities can play a role in improving the quality of life of our growing urban population by identifying the important functions that natural and seminatural areas perform in fostering the physical and mental well-being of its citizens and in reducing costs to municipalities that must use technology to replace otherwise "free" ecosystem services that natural areas and their species provide (Daily et al., 1997; Epstein, 1997) (e.g., temperature mitigation, chemical purification of air and water, flood control, groundwater replenishment, pest control, control of human, animal and plant diseases, pollination in gardens and farms). In addition urban and suburban environments contribute to ecology as a science by providing novel combinations of physical-chemical settings and organismal interactions for testing generalizations and hypotheses generated in ecosystems far from direct human influence.

The usefulness of the urban-to-rural gradient approach

A great deal of the credit for the recent revived interest in urban-suburban areas among ecologists can be traced to the ideas laid out by Mark McDonnell (see article by McDonnell in this Sustain issue) and Steward Pickett (a native of Louisville) in a paper that appeared in the scientific journal, *Ecology*, in 1990, and in a subsequent book, *Humans as Components of Ecosystems* (McDonnell and Pickett, 1993a). In these works they introduced and developed the concept of the urban-to-rural land-use gradient and suggested that researchers could adopt the ecological gradient paradigm to structure their studies of urban effects on nature. To explain briefly, since many cities in the United States have grown more or less concentrically from an urban center, factors like human density, extent of impervious land cover, road density, traffic volume, fossil fuel use, and industrial complexes tend to increase steadily as one moves closer to the urban core. As a consequence, natural habitat remnants in the landscape become surrounded by diminishing degrees of urban influence as their distance from the city increases. Therefore one can compare, for example, the species and ecological responses of forests within a metropolitan region and relate them to the particular set of urban, suburban and rural land use variables surrounding them as a means of determining how closely those land use differences can account for particular ecological responses. These types of studies are comparative and correlative, taking advantage of the environmental variation caused by a scientifically unplanned and uncontrolled "experiment", urbanization, to understand how cities and sprawl affect specific organisms and natural communities. However since individual environmental factors and conditions caused by these different land uses do not necessarily vary linearly or in spatial synchrony with distance from the city, and because they sometimes interact to intensify or counteract each other, both the magnitude of the variation and of the forest responses are not always ordered linearly with distance from a city center. These landscape level gradients are complex and often indirectly ordered in real space. Therefore multivariate ordination techniques are often used to explore relationships between suspected causes and ecological responses. To go beyond correlational studies using aggregate land cover or land use attributes, and to identify proximate causes of the ecological effects, follow-up studies must be performed involving on-the-ground measurement of the responses and their suspected causes, and if possible their manipulation in lab or field experiments.

To summarize, urban-rural gradient studies can increase our understanding of how a particular city might affect the species composition and functional behavior of natural ar-

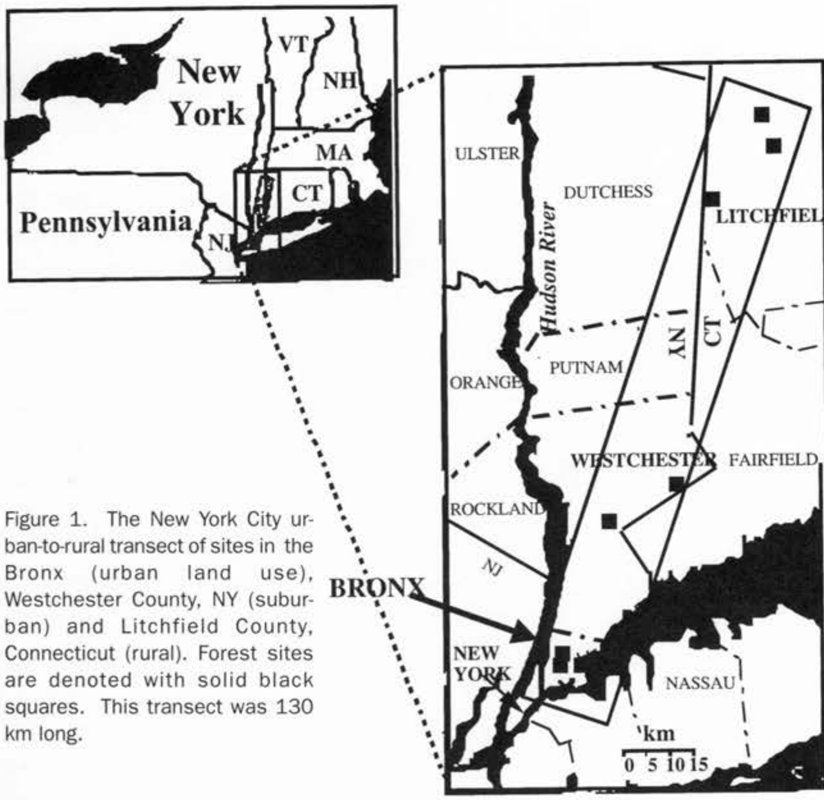

Figure 1. The New York City urban-to-rural transect of sites in the Bronx (urban land use), Westchester County, NY (suburban) and Litchfield County, Connecticut (rural). Forest sites are denoted with solid black squares. This transect was 130 km long.

eas presently within the city's boundaries by comparing urban forests (or other habitat types) with similar reference areas at increasing distances from urban influence. Managers and planners would find the comparative study of urban, suburban and rural natural areas particularly useful for predicting how parks and preserves currently far from the city may change once the sprawl front advances to surround them in the future. This may stimulate reevaluation of land purchasing priorities and development of adaptive management strategies that are tailored to a particular region (see McDonnell article in this issue of Sustain).

Comparing New York City and Louisville

The urban-rural gradient approach has been used successfully in the New York City Metropolitan area to understand how urban forests differ from those of similar tree species composition in outlying areas (McDonnell et al., 1997). I was fortunate to have been involved in these New York City studies for ten years before arriving in Louisville to establish another urban-to-rural gradient of forest sites that would allow us to begin comparisons among Metropolitan areas of different sizes and ages. Such comparisons would allow ecologists to determine which biotic or ecosystem responses are idiosyncratic to specific cities, and which responses are shared by natural communities in and near cities of different regions. While New York and Louisville differ in many ways, they do share some ecological similarities in being located within the Eastern Deciduous Forest biome and having many forest communities dominated by oaks. The urban-rural transect of forest sites in the New York City area (Figure 1) extends northeastward 130 km (85 miles) from the Bronx, NY (one of New York City's five boroughs), through suburban Westchester County, NY, to rural Litchfield County, Connecticut. In Louisville, KY the transect consists of Knob forest sites extending from Iroquois Park (Figure 2 and Figure 3A) southward 40 km (25 miles) through suburban Jefferson County Memorial Forest to the Bernheim Forest (Fig. 3B) in rural Bullitt and Nelson Counties. To maximize our ability to detect urban, suburban and rural land use effects on these forests, 30 x 30 meter plots were chosen within the forests using the following criteria: 1) dominance (≥ 50% of plot basal area) by the same tree species (red oak, *Quercus rubra*, in New York; chestnut oak, *Q. prinus*, in Louisville), and 2) location on the same or closely related soil series (Charlton-Hollis soils in New York, Tilsit soils in Louisville). By holding these factors constant we can increase our confidence that differences in matter, energy and species inputs into these forests and forest processes measured would be primarily due to the effects of land use and land cover surrounding the stands. Since the population of the Metropolitan Statistical Area (MSA) of New York is 21 million, our initial expectation was that the impact of such a large megalopolis on its respective natural systems would be at least an order of magnitude greater than that of Louisville with an MSA of 1 million.

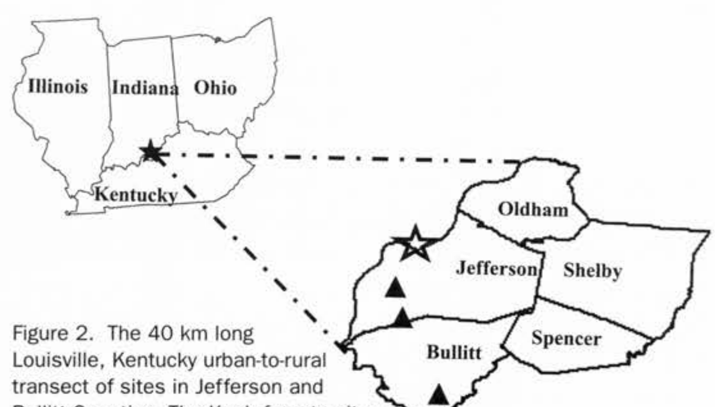

Figure 2. The 40 km long Louisville, Kentucky urban-to-rural transect of sites in Jefferson and Bullitt Counties. The Knob forests sites (solid black triangles) are Iroquois Park (urban), Jefferson County Memorial Forest (suburban), and the Bernheim Forest (rural). The star represents Louisville.

Both projects seek to understand the magnitude of the variation along the gradients in factors that can drive change in forest systems, chief among them temperature and air pollution, both of which can be linked directly to land cover and land use attributes of urban-to-rural areas. Both projects also focused initial experiments on measuring the ecosystem processes occurring in forest soils (leaf litter decomposition and soil nitrogen (N) cycling), which were expected to provide short-term responses (relative to tree successional responses, for example) to presumed variability in temperature and chemical deposition along the two gradients. A temperature differential was expected due to the well-established phenomenon known as the urban heat island effect, defined by comparing city temperatures with temperatures in neighboring rural areas. The temperature differential between urban and rural environments for near-surface air can be as great as 12° C (22° F; Oke, 1995) and this contrast intensifies with urban expansion. Westendorf et al. (1989) found the *maximum* air temperature difference between downtown Louisville and a nearby rural area to be 7.5° C (13.5° F). The mean monthly average temperature difference between urban and rural weather stations along the New York gradient varied from 2 to 3 ° C (about 4 to 6 ° F) year round, but most of that difference occurred within 15 miles of Manhattan's Central Park, thus defining the boundaries of the heat island (McDonnell et al., 1993b). Mean monthly temperature data, as well as temperature measurements within forest plots, are now being gathered for the Louisville gradient. We have analyzed a small temperature data set from July 15-30, 2002 and found a detectable gradient effect even inside shaded forest plots.

Figure 3.A. A view from the southern end of the Knob forest at Iroquois Park in southwestern Louisville. The forest is surrounded by tree-lined residential areas to the south and west, a busy avenue to the east and a golf course to the north. Downtown Louisville can be seen at the center horizon about three miles away.

Figure 3B. The forested Knob hills near Bernheim Arboretum and Research Forest about 25 miles to the south of downtown Louisville. These forests are surrounded by farms, distilleries and small residential clusters. Photos Margaret Carreiro.

The average daily mean air temperature difference 1 meter above the forest soil for three urban plots was 1.06° C (1.91°F) warmer than the mean of three rural plots, with suburban plots being intermediate; the difference at the soil surface was 1.04 ° C (1.87°F) warmer for urban plots (from temperature loggers recording every 20 min.). Since warmer temperatures increase the metabolic rate of many organisms, the urban heat island should extend the activity of plants, invertebrates like insects, "cold-blooded" vertebrates and microbes over a greater portion of the year. In fact, White et al. (2002) could detect the heat island effect on vegetation from space. Using satellite data to create a "greenness index", they found that, for the eastern deciduous forest biome in the U.S.A., urban vegetation experiences a growing season that was on average 7.6 days longer than that of paired rural vegetation nearby.

City air is also known to contain many pollutants derived from fossil fuel combustion. Some of these pollutants are injurious to people, animals and vegetation (e.g., ozone), while others can be used as nutrients by plants and microbes (Smith, 1990). One of these nutrient pollutants is nitrogen, which exists in various chemical forms in the atmosphere. These include NO_x gases (mostly from motor vehicle exhaust), and soluble and particle-associated ammonium (NH_4^+) and nitrate (NO_3^-; derived from NO_x gases). Thus, urban atmospheres contain high concentrations of nitrogenous compounds in gaseous, soluble and particulate form. A comparison of N emissions data for New York City and Louisville-Jefferson County shows that Jefferson County, with a population 8.7% the size and 6.8% the density of New York City, produces a surprisingly higher

County	Population 2000	Area km^2	Pop. Density km^{-2}	NO_x	NO_x km^{-2}	NH_3	NH_3 km^{-2}
New York City (5 cos.)	8,008,278	776	10,324	190,102	245	8685	11.20
Jefferson (Louisville)	693,604	986	704	52,934	54	1471	1.49

Table 1. Nitrogen emissions in 1999 in relation to county size and area for New York City, NY, and Jefferson County (Louisville), KY. New York City is composed of five counties (or boroughs), Bronx, New York, Richmond, Kings and Queens. NO_x, (NO and NO_2 gases) and NH_3 emissions are in units of metric tons per year and are only those that originate from area sources that are mobile and small stationary, *not large point sources*. Thus these values are more related to population size and factor out contributions from power plants that can vary greatly among cities. Source: Population and area data from US Census Bureau-2000; EPA AIRData-National Emissions Trends database http://www.epa.gov/air/data/nettier.html.

proportion of NO_x gases on an areal basis (22% that of New York City) than would be predicted from population density alone (Table 1). Louisville's annual mean NO_2 concentration is 67% that of a Bronx, New York City site, and its annual mean concentration of particulates in the 2.5 to 10 µm size class is nearly 1.5 times *greater* than that of the Bronx (36 vs. 25 $\mu g/m^3$; KDAQ, 2000; NYSDEC, 1998). These emissions and air quality data suggest that forests in moderately sized cities may receive N inputs that are almost as large as those in cities with an order of magnitude larger population.

The reason for this ecological interest in atmospheric N concentration is the fact that it is a plant and microbial fertilizer. The availability of inorganic nitrogen (N) to forests (typically as ammonium, NH_4^+, and nitrate, NO_3^-) is important to quantify, since it has been identified as the nutrient that most often limits plant growth in terrestrial ecosystems (Vitousek and Howarth, 1991). However, N in excess of plant and microbial uptake (a condition known as N saturation) may result in its exportation from the forest via leached soil water (see Bukaveckas and McCandless article in this issue of Sustain) that can contaminate aquatic systems, and may result in its microbial transformation to a gas (N_2O) that volatilizes and contributes to global warming (Aber et al. 1998).

Since there has been no prior information on atmospheric deposition of N to urban and suburban vegetation, both research groups chose to quantify the amount of N being deposited to forests along their respective urban-to-rural gradients in wet deposition (rainfall) and the amount captured by foliage in particulate form (dry deposition) to determine if urban forests receive more N as NO_3^- and NH_4^+ than their suburban and rural counterparts (Figure 4). Over an 11-week period during the 1996 growing season, Lovett et al. (2000) found that the mean amount of N (as NO_3^--N and NH_4^+-N combined) falling through the tree canopy during rain events and being deposited on the forest floor in urban forests was 1.5 to 2.5 times that in throughfall collected in suburban and rural forests along the New York City gradient (range for all sites 9.87 to 29.47 mmoles N m^{-2}). However, most of the difference in N was not due to the amount of N in the rainfall itself, but in the dry-deposited particulates that were captured by tree foliage between rain events. There was a 17-fold difference along this urban-rural gradient (urban >> suburban = rural) in the amount of N in this "net throughfall" component of precipitation with 70% of that nitrogen being in NO_3^- form. Carreiro and Tripler (unpublished data) have found the same trends in N deposition in Louisville area forests. Between May and October 2002 throughfall amounts of N were similar in magnitude and pattern to those along the New York gradient and the net throughfall N component to oak plots in urban Iroquois Park was five times greater than to rural stands (Bernheim Forest) just 25 miles from the city with Jefferson County Memorial Forest receiving intermediate amounts. Nitrogen in nitrate form comprised 80% of net throughfall N in Iroquois Park. Figure 5 A-C illustrates a typical pattern in N deposition in rainfall, throughfall and net throughfall from precipitation collected along the Louisville urban-rural gradient during the week of July 18 – 25, 2002. Such large localized N inputs may affect plant productivity and nutrient cycling processes in urban forests differently from forests a

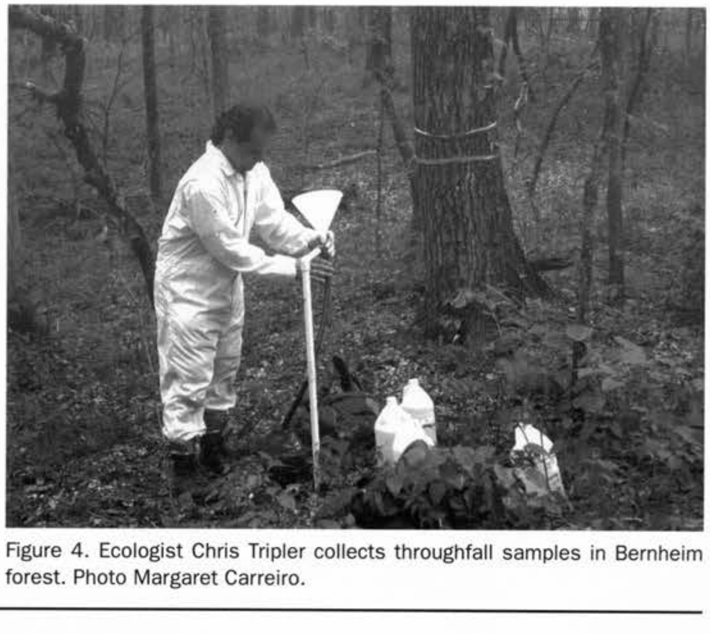

Figure 4. Ecologist Chris Tripler collects throughfall samples in Bernheim forest. Photo Margaret Carreiro.

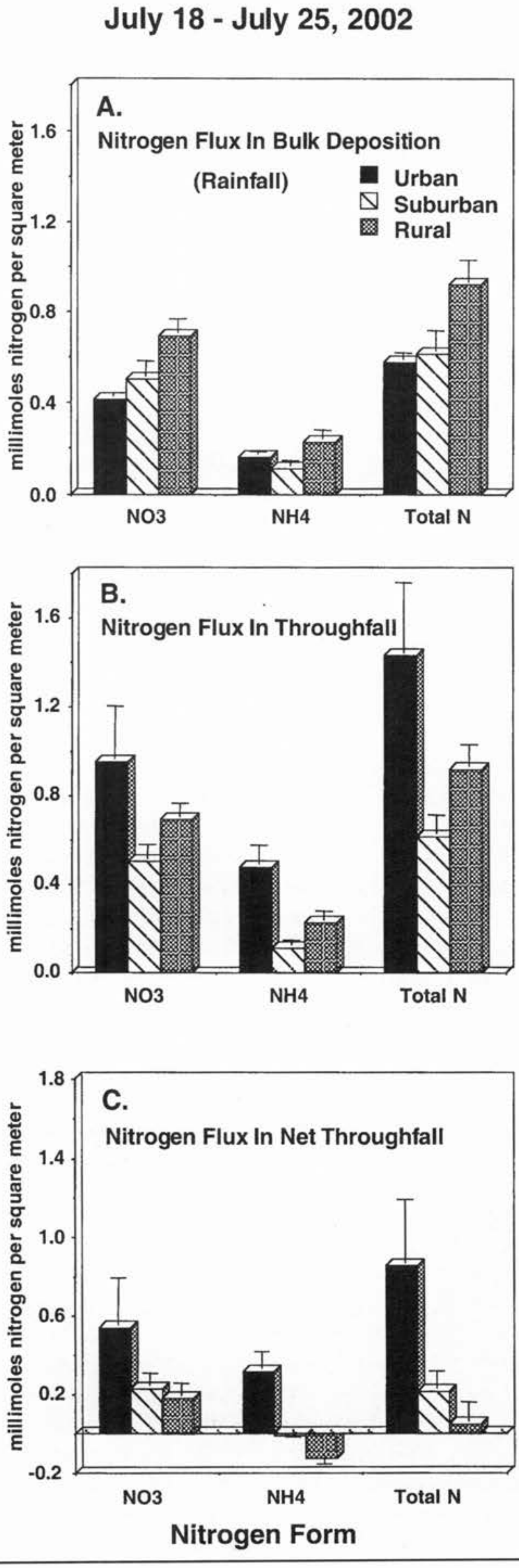

Figure 5. Nitrogen (N) deposition (millimoles N / m^2 ± S.E.) as nitrate (NO_3^-), ammonium (NH_4^+) and total N (NO_3^- and NH_4^+ combined) to oak forest plots in Iroquois Park (urban), Jefferson County Memorial Forest (suburban), and Bernheim Forest (rural) during the week of July 18-25, 2002. Mean precipitation ± S.E. for that week was Urban 1.37± 0.47 cm, Suburban 0.53 ± 0.07, and Rural 1.34 ± 0.14 cm so differences in N flux between urban and rural are not caused by rainfall volume.

A. Bulk deposition is rainfall collected in open areas close to the forest. It consists of rainfall and some particulate matter and therefore slightly overestimates the amount of N in pure rainfall. B. Throughfall is the precipitation collected from beneath the forest canopy and contains amounts of N present in rainfall and particulate N in "dust" washed from tree leaves. C. Net throughfall N is calculated by subtracting bulk deposition N from throughfall N and provides an estimate of dry deposited particulate N (the N that was captured in dust by leaves).

Since leaves normally absorb some nitrogen during canopy processing, net throughfall N is a slight underestimate of dry deposited N. Negative values in net throughfall indicate net canopy uptake. Note that the greater amount of N in urban throughfall is due to the particulate (net throughfall) contribution and not the contribution from rainfall. The patterns for this particular week were typical of the overall growing season trend from May to October, 2002 across this urban-to-rural gradient, but week-to-week variability in magnitude and sometimes in trend direction occurred.

short distance from the city. So if trends observed for both New York City and Louisville are applicable to other cities, urban forests may be receiving a large N subsidy in dry deposition during the growing season. Based on the large proportion of that N being in NO_3^- form, fossil fuel emissions, particularly those from motor vehicles, is the likely major source.

Ecosystem Effects of Atmospheric Nitrogen Deposition

So as forest trees perform an ecosystem service for people in cleansing the air of particulates that would be harmful to breathe, we must not forget that those materials ultimately enter forest soils where they can affect these systems over time. Whether this additional N benefits or harms a forest in the long run depends on the forest's plant species composition and soil type. Based on extensive ecological experimentation with N added as fertilizer to forests in the U.S.A. and Europe (Vitousek, 1997b; Tamm, 1991), we know that not all species of plants can take equal advantage of greater N availability for promoting growth. Therefore some plant species will be able to outgrow and reproduce faster than others in the presence of additional N. Whether those plants will be native species we wish to retain in the forest, or nitrogen-loving exotic species that can replace low-nitrogen tolerant natives remains to be seen for the Louisville area. But species replacements due to N enrichment have become apparent in other ecosystems around the world (Vitousek, 1997b).

Besides stimulating growth, additional N could also result in plant foliage containing higher concentrations of N. Having more N in foliage is a two-edged sword for many plants. On the one hand it permits them to photosynthesize more, but it may also increase their vulnerability to insect pests and fungal pathogens that can also grow faster on leaves that contain more N (Huber and Watson, 1974; Mattson, 1980; McClure, 1991). Adding N to the decaying wood, leaves and soil organic matter in the forest changes the rates at which microbes can break down these materials and circulate their constituent nutrients within the forest system. The decay of leaves of some forest plants like dogwood are accelerated, while tougher leaves of species like oak are slowed down by additional N (Carreiro et al., 2000). If the amount of N entering the forest from the atmosphere exceeds the amount plants and microbes can take up and use for growth, then a forest system may begin to "leak" excess N, especially as NO_3^- in soil water, or volatilize N back into the atmosphere. This would greatly alter the normal behavior of forests in the landscape since they will have become exporters of N to their surroundings rather than retentive N sinks. In addition, NO_3^- as it leaves the forest soil system takes along with it basic cations, such as magnesium and calcium, which are essential for plant growth and long-term forest productivity (Vitousek et al., 1997b; Aber et al., 1998). Concentrations of these basic cations in acidic Knob forest soils (Carreiro and Tripler, unpublished data) are not as high as those in the limestone-based soils of the Bluegrass plain that surrounds them due to their different geological origin. This difference in soils makes Knob forests more vulnerable to the harmful effects of both acidic deposition and N deposition than other forests in this region.

Back to the bird's-eye-view of urban and suburban landscapes

Both warmer temperatures and greater atmospheric N deposition close to cities provide examples of why natural habitat acquisition alone will not necessarily preserve the species and ecological benefits we wish to maintain for the local region as suburbs expand and become transformed into cities themselves over time. Natural habitats are not closed systems unaffected by the changes in land use and cover that surrounds them ("the matrix" in which they are embedded). In fact, small preserves are even more vulnerable to such matrix effects than large preserves because they have larger edge-to-interior ratios (perimeter length divided by area of the preserve). Therefore inputs of energy (like heat and noise), matter (like nitrogen deposition) and species (wildlife as well as humans and our unleashed pets) from external surroundings are likely to penetrate more deeply and likely to have a disproportionate effect on the species population dynamics and ecosystem processes than occur inside smaller preserves (Saunders, 1991; McKinney, 2002). In cities and suburbs these external surroundings consist primarily of commercial and residential areas and the range of human activities (traffic, fossil fuel emissions, landscaping) that occur within them.

To reduce our impact on small but important natural areas, we should be aware of opportunities to alter that matrix and our behaviors within it so as to reduce undesirable effects on these natural areas, which are repositories of our local cultural and biotic heritage, and to maintain their integrity for future generations to enjoy. For example, forests next to suburban developments are often exposed to small-scale but numerous disturbances caused by residential and commercial activity along their boundaries that can affect the regeneration of native species. Matlack (1993) found that most suburban activities penetrated more deeply into forests than even microclimatic edge effects. These sociological edge effect phenomena included trash dumps, piles of building rubble, campsites, woodpiles and piles of lawn clippings and raked leaves. He also found that most small forest patches of 0.6 to 20 hectares (1.5 to 50 acres) possessed numerous leaf litter-free paths, and that residential lawns often penetrated beneath the adjacent tree canopy. In addition to trampling effects, these disturbances increase the probability of introducing both exotic plant and invertebrate species like earthworms from gardens and lawns to forests that have not evolved with these species. Increased availability of exotic seed and plant cuttings, greater light penetration from disturbed edges and higher N deposition can interact to favor the spread into forests of undesirable low light-tolerant exotic species like garlic mustard, Japanese and shrub honeysuckles, privet, English ivy, creeping and shrub *Euonymus* species, Japanese grass and Japanese barberry. Some of these species may alter soil conditions to favor their own growth over time (Kourtev et al., 1998) to the disadvantage of many native woodland species.

We could also pay more attention to the plant species we choose to grow in our gardens and in our municipal plantings. Many of these species are exotics with the potential to escape their "civilized" settings either by wind or bird dissemination of seed. Some of these exotics (in addition to the ones listed above) have the ability to invade forested and non-forested natural areas (Mack et al., 2000). For a listing of exotic species often used in commercial landscaping and residential gardens, and that have been creating ecological problems in Kentucky, visit the website for the Kentucky Exotic Pest Plant Council (www.se-eppc.org/ky). Planting native tree, shrub and herbaceous species in our home gardens can also improve their quality as feeding and resting islands for native wildlife that people prefer viewing (hummingbirds, warblers, butterflies) rather than those considered pests (exotic starlings and English sparrows). Individual and communal decisions about our landscaping can therefore make the matrix that surrounds natural areas less harsh and more hospitable for many of the native species that we are trying to sustain in our parks and preserves. We may not be able to personally slow down species loss in distant locations like tropical rainforests, but there are many choices we as individuals can make to help preserve our own local species legacy.

Ecological studies of urban and suburban areas can provide a greater understanding of both the obvious and subtle ways that we alter the natural world within and around dense human settlements. Such studies can help us find ways to work with nature to improve the quality of our lives as we accommodate the needs of desirable species that are trying to share the land with us. Decreasing the friction in our relationship with nature can decrease the costly feedbacks on society of ignoring our connection to the land and its waters. We already know that people are in the driver's seat in terms of defining how our landscapes look and function. The biological and ecological condition of both our natural areas and our human communities are a reflection of our values and our civic commitment, as much as it is nature's. Individual, business and larger-scale political decisions about zoning, stringency of air and water quality standards, use of water-permeable paving materials, species used in plantings, and degree of energy and water use efficiency can help our landscapes, both the built and the natural, work together to create more environmentally sustainable and resilient cities for a changing future.

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Biographical Information

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Birds as Bioindicators of Urban Lead Pollution

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Introduction

The legacy of lead pollution is concentrated in urban areas due to the high concentration of older homes with lead-based paint, roads where lead was released from automotive exhaust, and large buildings that still burn recycled motor oil. Prior to 1978, lead was used as a pigment, resulting in bright long-lasting, white house paint with good surface adhesion. Lead was also added to gasoline as an anti-knock agent resulting in the release of 4-5 million metric tons of lead in the U.S. (NAS, 1993; Xintaras, 1992). Since lead additives have been phased out, lead release to the environment has declined substantially (Johnson et al., 1995). People and animals in the urban environment are exposed to lead from diverse sources: 1) soil which has been contaminated from industry, leaded gasoline or pulverized leaded paint, 2) naturally occurring (background) lead in the soil and 3) lead released into the air as a byproduct of burning recycled motor oil. Since the ban on the production of lead-based house paint and elimination of leaded gasoline additives, the major source of lead exposure in U.S. cities is from lead found in soil and dust particulates (Mielke and Reagan, 1998). Although natural (background) lead levels in Kentucky soils is known to be low (Karanthanasis, 1993), no systematic survey of lead sources in urban environments such as Louisville has been conducted. The age group most susceptible to lead poisoning is children under 6 years old. Blood lead testing conducted in 1998 showed that 12.9% of Kentucky children screened were above 10 :g dL^{-1}, a level that may result in behavioral and developmental problems (SOKE, 1999). It is probable therefore that there are areas of high soil lead in Louisville.

Louisville, like many cities has a series of parks scattered throughout the city. Our study sites, parks and a college campus, have been highly urbanized. The parks in Louisville were designed by Frederick Law Olmsted in the late 1800s, and are composed of 2,000 parkland acres (contained in 18 parks) and 15 parkway miles. This system was designed to be a series of interconnected green spaces throughout the city. Mr. Olmsted also designed the Belknap Campus of the University of Louisville used in this study. While the designs were innovative, they are listed on the National Register of historic places (Olmsted Park System, listed 05-17-1983); the parks were designed "as a place for quiet reflection and relaxation away from crowded city life, as well as for active recreation, and socializing where all people could enjoy themselves." (Louisville Olmsted Parks Conservancy). The vegetation in the Olmsted areas is dominated by mature trees, mostly oak and maple, and lawn grass, reflecting the needs of intended users, i.e. people not wildlife. The few shrubs and small trees present are generally not native. This leads to a habitat matrix very far removed from the natural habitat present before urbanization.

Birds as Bioindicators of Pollution

Bioindicators are species whose presence or physiological condition provides information about the overall health of the community or ecosystem in which they are found. To be useful, an indicator species should provide quantitative information that is relevant to one or more stressors that impact both ecological and human health. Studies that track the fate of pollutants typically rely on measurements of contaminants in the environment (water, air or soil samples) as well as contaminants contained in the tissues of organisms (see Figure 1). An important benefit of measuring pollutants in organisms vs. in the environment is that the former may provide a better index of human exposure and potential health effects. For example, bird tissues have been used to track contaminants in coastal ecosystems (Burger and Gochefeld, 1997; Gochfeld, 1997) and these data may provide a better proxy for human exposure than measuring pollutant levels in water directly. A variety of bird species are commonly found in urban environments. As individual birds move within their home range, they integrate pollution sources at scales relevant to humans. By contrast, soil sampling produces site-specific and often highly variable results.

Birds are relatively easy to collect and samples of blood and feathers can be obtained without causing mortality. Lead concentrations in bird blood are determined by the amount of lead recently absorbed through the gut and airways and that lead mobilized from stored sources such as bone

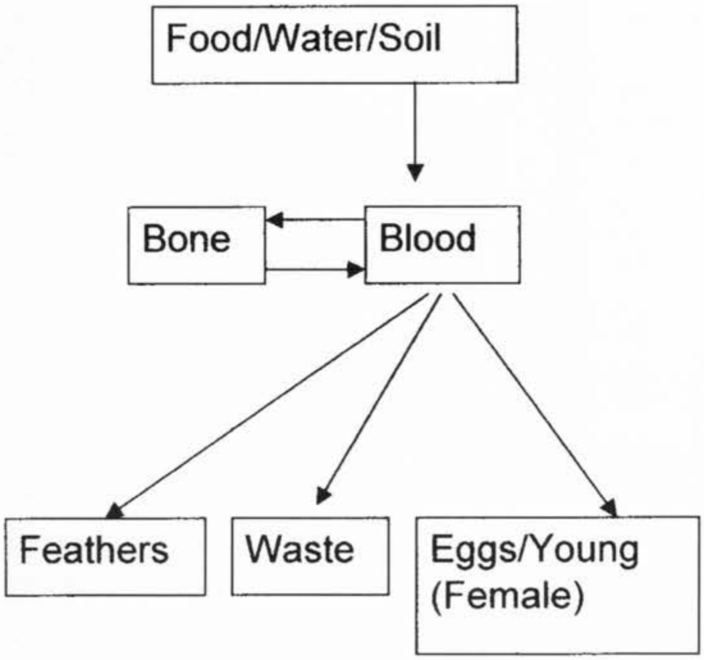
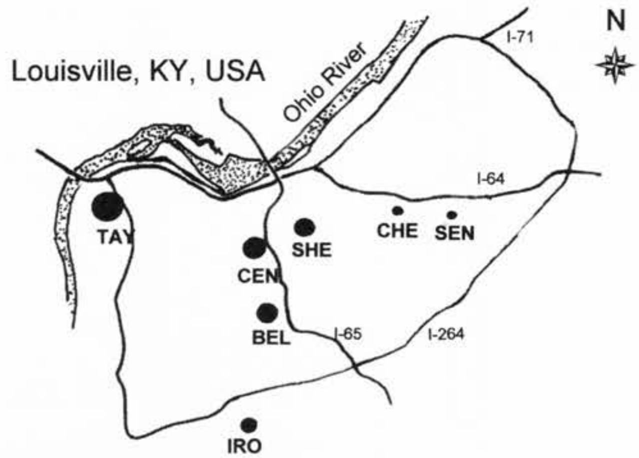


Figure 1. A conceptual model of lead intake and loss in birds. Ingested lead is partitioned from the blood to the main storage site of the bones. This stored lead can subsequently reenter the blood during times of calcium mobilization. Lead from the blood can also be removed through excretion, incorporation into feathers, or, in females, through deposition in eggshells and young.



Site	Soil lead (mg kg^{-1})
Taylor Park (TAY)	132
Central Park (CEN)	94
Belknap Campus (BEL)	86
Iroquois Park (IRO)	65
Shelby Park (SHE)	77
Cherokee Park (CHE)	31
Seneca Park (SEN)	27

Figure 2. Location of sampling sites within the Louisville metropolitan area (TAY-Taylor Park, CEN-Central Park, SHE-Shelby Park, CHE-Cherokee Park, SEN-Seneca Park, BEL-Belknap Campus, IRO-Iroquois Park). Circular symbols are centered over each park, and are scaled to represent the average measured soil lead in the park.

(Figure 1). For migratory species, lead burdens will be a composite of their local (breeding grounds) and distant (wintering grounds) exposure to lead. Results of bird biomonitoring may be used to identify areas where humans and wildlife suffer from lead effects provided that certain conditions are met. This approach was used by Eens et al. (1999) in passerine birds as bioindicators of point-source heavy metal contamination in Belgium, and in pigeons by Llacuna et al. (1995) in Spain. To use songbirds as monitors of non-point source lead contamination it is important that non-migratory species are used since their lead burdens will reflect local exposure. Second, different bird species occupy a variety of ecological niches (herbivores, insectivores) and these along with other factors such as age and sex are likely to effect levels of exposure. Third, species may differ in their relative abundance throughout urban, suburban and rural landscapes and thereby complicate the use of a single species approach. To address these issues and assess the feasibility of using birds as bioindicators on non-point source pollution in urban environments, we undertook a study of lead exposure among bird species found in urban and suburban areas comprising the Louisville metropolitan area.

Bioindicators of Lead in Louisville

The Louisville metropolitan area has a population of nearly 700,000 and is the 16th largest city in the United States, Figure 2. The study described here was designed to determine the suitability of birds as bioindicators of lead pollution in urban and suburban locations throughout the city. We did not intend to assess point-source contamination problems (such as industrial sites) but rather, focused on the hypothesis that lead levels in birds will be reflective of an urban to suburban gradient in levels of pollution (see article by Carreiro, this issue). Wooded parks provided a suitable habitat with a variety of bird species and comparable food resources across the urban to rural gradient. By choosing similar habitats and species across this gradient, we hoped to isolate the effects of varying lead exposure on lead levels in birds. Seven parks were chosen and these included: Taylor, Central, Shelby, Cherokee, Seneca, Iroquois and the Belknap campus of the University of Louisville (see map; Figure 2). Over a two-week period in 1997, samples of the top 5 centimeters of soil from four locations at each study site were collected. Samples were taken from bare soil spots using a clean, polypropylene coring device, placed into new zip-top bags and well mixed. Approximately 10 grams of soil were removed from each plastic bag and placed into a new, labeled, 10cm polycarbonate dish. An oven set at 60°C dried the soil to a constant weight (approximately 48 hours). A water-cooled 'wiley mill' with a stainless steel blade was used to pulverize the samples. Subsequently 0.1 to 0.2 g of soil powder was acid digested. The same method used for blood lead determinations was used to find total lead in soil acid extracts. As a quality control, standard soil samples (Environmental Resource Associates) were analyzed identically to the park soil samples. We found that soil lead levels ranged from 127 mg kg^{-1} in Taylor Park to 27 mg kg^{-1} in Seneca Park. There was also an apparent gradient in soil lead from northwest to southeast Louisville, roughly following from increasing to decreasing industrialization of the areas surrounding the parks sampled.

Is the level of soil lead contamination in Louisville parks cause for concern? Most parks were above background for the state of Kentucky (Wells et al., 1993), indicating low-level contamination. The current guideline for the action limit on residential soil lead is 400 mg kg^{-1} (EPA, 2001). The parks in this study were well below (range from 27 to 132 mg kg^{-1}) but since the values given are averages for each park, some individual samples were higher. The greatest soil lead was the area closest to the interstate in Taylor Park, at over 230 mg kg^{-1}. Mielke et al. (1997) found that when the median lead content in soil is less than 310 mg kg^{-1} then the median blood lead concentration in children will be below the CDC recommended action level (10 :g dL^{-1}). A study in inner city Washington DC (Elhelu et al., 1995) demonstrated the significant role that soil lead plays in children living there. For values of soil lead ranging from 130 to 444 mg kg^{-1} in seven wards of that city he found elevated pediatric blood lead. Reagan and Silberg (1989) suggested that soil lead as low as 50 mg kg^{-1} could pose a risk to small children.

Lead in Birds

At each site, songbirds were captured by mist netting and released unharmed after collecting a blood sample. Bird mist netting was conducted between daybreak and 11:30 am during May through June in 1996 and 1997. Issues of range and site-fidelity are important to consider. In our study we chose birds with relatively small ranges, and sampled during the breeding season when their site fidelity is highest. A total of 67 birds was captured (Table 1); the species used for this study were European Starling (*Sturnus vulgaris*), House Sparrow (*Passer domesticus*), American Robin (*Turdus migratorius*) and White Throated Sparrow (*Zonotrichia albicollis*). Other species that were caught but not used for blood collection included the Song Sparrow, Swainson's Thrush, Brown Headed Cowbird, Gray Catbird, Northern Cardinal, Oven bird, Carolina Wren and Common Grackle.

Our primary response variable was the concentration of lead in blood although we also measured the activity of a blood enzyme (delta-aminolevulinic acid dehydratase; ALAd) that is very sensitive to inhibition by lead and is often used as a surrogate measure for lead exposure. Between 100-200 :L of blood was removed by venipuncture, the volume scaled to body weight of the birds captured. Syringes and needles were treated immediately prior to use with an anticoagulant (heparin). An aliquot of blood used for lead measurement was stored in new, labeled polypropylene tubes at -80^{0}C. Blood was thawed, mixed and a subsample was analyzed for total lead content using a graphite furnace atomic absorption spectrophotometer. A standard blood sample from the College of American Pathologists was used to verify the consistency of the results. ALAd was determined within 4 hours of blood collection according to the method of Wigfield and Farrant (1981) using an incubation temperature of 42^{0}C to accomadate the higher basal temperature of birds.

Site	Robin	House Sparrow	Starling	White throated sparrow
Taylor	5	1	7	-
Seneca	-	1	6	1
Shelby	4	3	8	-
Central	-	3	6	5
Cherokee	1	3	-	-
Belknap	1	1	-	1
Total N	11	12	27	7

Table 1: A total of 67 birds were captured as part of this study. Blood samples were obtained from 57 individuals representing four species (House Sparrow, White Throated Sparrow, European Starling and American Robin). Ten individuals were migratory species and did not have blood drawn for analysis.

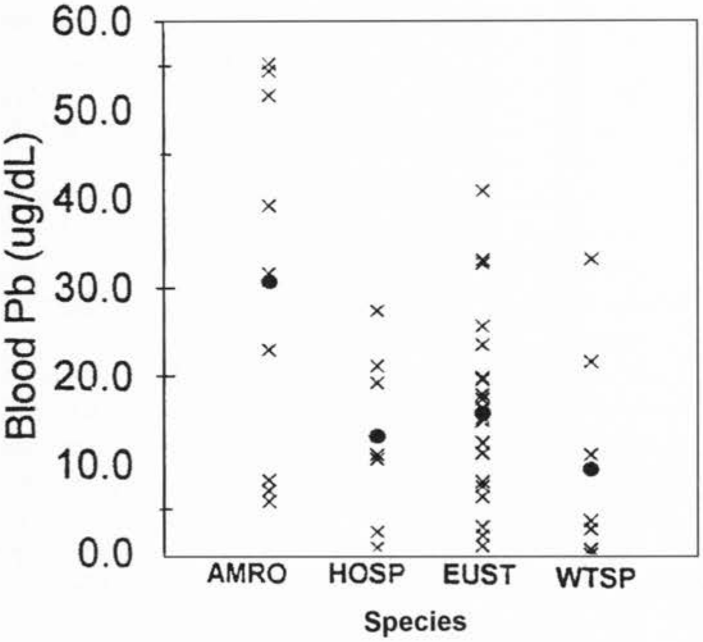


Figure 3. Blood lead concentrations of four common bird species found in municipal parks in Louisville, KY. Individual values as well as group means are shown, [X] and [Σ] respectively. Differences between species were found to be statistically significant. AMRO-American Robin, HOSP-House Sparrow, EUST-European Starling, WTSP-White-throated Sparrow.

Blood lead levels in individual birds varied by tenfold ranging from values near the limits of detection (< 3 µg dL^{-1}) to over 50 µg dL^{-1} (Figure 3). Average lead concentrations differed among the four species and these differences were found to be statistically significant (p 0.016). As a group, robins had the highest blood lead (average = 31 µg dL^{-1}) although birds with elevated blood lead levels included all four species. Males exhibited higher blood lead concentrations than females (Figure 4) and differences between the two sexes were found to be statistically significant (p=0.01). A possible explanation for this difference is that females actively select grit and food items that are high in calcium during their reproductive period (Jones, 1976; Ankney and Scott, 1980). Lower dietary intake of calcium among male birds has been shown to result in increased accumulation of lead and other metals (Scheuhammer 1996). In this study we deliberately sampled birds during their reproductive period to ensure small foraging ranges but it is possible that female body burdens of lead were reduced at this time due to egg-laying. Mobilization of stored lead through production of egg shells or young could cause a decline in female lead levels (Burger et al., 1999) and a prior study has shown that eggshells of migratory species are derived from local food sources (as opposed to winter feeding-grounds; Blum et al., 2001). These factors suggest that our estimates of lead levels in birds are likely to be conservative and yet we found that elevated concentrations were common for a variety of bird species collected throughout the Louisville metropolitan area.

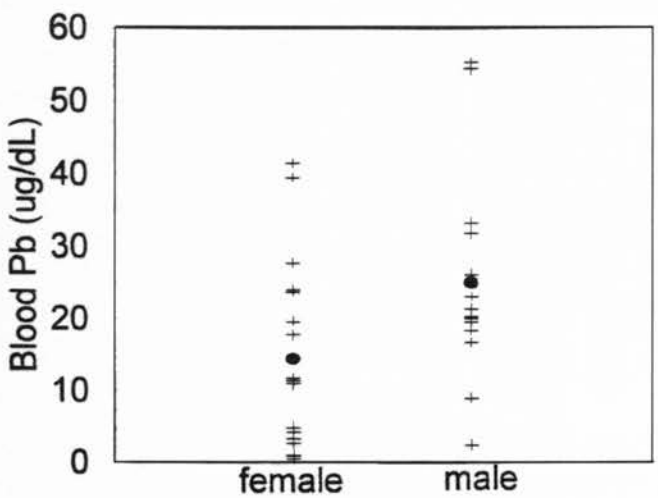


Figure 4. Blood lead concentrations in male and female birds collected in municipal parks in Louisville, KY (individuals denoted by [+] and average values by [Σ]). Blood lead concentrations in males were found to be significantly higher than those in females.

We relied on blood lead concentrations as our primary indicator but we also measured the activity of a blood enzyme (ALAd) that has been used in previous studies as a surrogate for lead exposure. The relative activity of this enzyme can vary from 1.0 (100% activity) to as low as 0.3 (30%) indicating a marked inhibition of heme synthesis. Starlings exhibited lower levels of enzyme activity (0.566 relative activity) than did robins (0.748 relative activity) suggesting that they experienced higher levels of lead exposure. This finding contradicted direct measurements showing that starlings had blood lead levels only half of those observed in robins. The correlation between blood lead and ALAd activity was low across the four species and among individuals within each species. Therefore, we concluded that enzyme activity levels were not a useful biomonitoring parameter in urban birds.

How are bird blood lead and soil lead related?

Statistical analyses to link blood lead in birds with lead levels in soils were complicated by unequal and small sample sizes for different species of birds collected at various sites. Therefore, we relied on a statistical procedure (randomization analysis; Manly, 1991) that entails re-sampling of data to determine the likelihood that differences in bird lead concentrations between low lead sites (soil <80 mg kg^{-1}) and high lead sites (soil >80 mg kg^{-1}) were statistically significant. The value distinguishing low vs. high sites (80 mg kg^{-1}) is somewhat arbitrary, though a value in this range is appropriate based on the most conservative published guidelines for residential areas. The difference in blood lead between birds caught at high and low soil lead sites (all species) was 11.30 $\mu g\ dL^{-1}$ and randomization analyses suggested a high probability that the observed mean was not due to chance ($p < 0.0001$). For this analysis all birds were randomly assigned to groups, equal in size to the two original groups (> and < 80 $mg\ kg^{-1}$ soil lead). Randomization analysis was used to test for a significant difference between the groups. The birds were randomly reassigned to two groups, a mean was calculated for each group, and then the difference in the means between the two groups was calculated. This procedure was repeated 50 times. The true difference of the means, 11.30 :g/dL, was never reached or exceeded. This indicates that bird blood lead is highly correlated with soil lead, when comparing low *vs* high soil lead sites.

Average blood levels of birds caught at high lead sites was greater than those caught at low lead sites. Interestingly no birds were caught at low soil lead sites that had high blood lead. Though we observed species- and sex- specific differences in blood lead concentrations, this preliminary study does not allow us to recommend a specific monitoring protocol.

Effects of Lead Exposure on Birds

It is unlikely that the blood lead levels measured in this study were sufficiently high to cause lethal or sublethal effects on bird populations in urban parks although we cannot discount the possibility that wild birds with higher lead may not have been caught due to lead's deleterious effect on motor function and survival. Avian blood lead concentration of 0.2 ppm is considered as a threshold for elevated lead exposure (USFWS, 1986). The biomonitoring approach necessitates the use of wild birds whose nutritional status and parasite loads are unknown. Esselink et al. (1995) suggested that body condition in birds is a very important factor when they are used for biomonitoring. He found that body fat and protein levels must be considered due to their effects on the concentration of heavy metals in tissues. Barus et al. (2000) demonstrated that intestinal parasites in birds accumulate lead and other heavy metals. We scored birds for visible intraclavical fat deposits, but found no relationship between blood lead and fat status (data not shown). In future studies it may be advantageous to assess additional bird health parameters and take fecal samples for an indication of intestinal parasite load so that these may be included as modifying factors in the development of a biomonitoring protocol.

Conclusions

We found that there may be an urban gradient in the lead found in soil, and consequently in the birds living on that soil. Overall the levels of lead found in the soil were not dangerously high. Also the birds captured exhibited low to moderate blood lead levels. If these birds are representative of the avian population as a whole in Louisville, Kentucky, the populations would not be expected to be greatly impacted by lead intoxication. Burger and Goschfield (1996) demonstrated in an experimental setting that the developmental disadvantage of lead to nestlings, namely retarded growth, was compensated by changes in adult care. It is less clear how chronic low levels of lead would affect overall fitness in songbirds. The birds that are present in urban environments have access to a greater abundance of food (Marzluff and Ewing, 2001) and therefore may attain higher body fat than birds of the same species in more natural environments. Edens and Melvin (1989) demonstrated that Japanese quail with high body fat were less sensitive to the physiological effects of lead than were birds of average body fat. In our study we found no correlation between body fat and blood lead. However we had no measure of fitness. It may be possible that fatter birds with the same blood lead would have shown less deleterious effects. It is also possible that our sampling techniques only captured those birds that were not greatly affected by lead. It may be possible that other birds of the same species with higher lead levels would have a decreased functional capacity, and may not have been caught, or that species sensitive to lead would be absent from the sites altogether. However, our results suggest that using birds to monitor for lead exposure in urban areas is an approach worthy of further study. The concentration of lead in blood samples taken from birds living in urban parks was correlated with soil lead concentrations. The species exhibiting the broadest range of lead contamination was the American Robin, though individuals exhibiting elevated blood lead levels were found in all four species we investigated. An enzymatic surrogate for lead exposure (ALAd) was not found in this study to be a good indicator of either blood lead or soil lead. While this enzyme is quite a sensitive indicator of lead, it is also sensitive to inhibition from other factors such as other metals and other factors leading to oxidative damage (Flora et al. 2003). Its use in wildlife studies of free living birds as an indicator of lead exposure should be re-evaluated.

The species composition of the birds captured for this study reflects the highly urbanized nature of the study sites, 2 non-natives including House Sparrow and European Starling being 2 of the most abundant species captured. It is possible that monitoring for areas of high urban lead using the non-native songbird species that inhabit these degraded sites, could lead to amelioration of the lead contamination. This in turn could improve habitat conditions for native species to reestablish themselves in these areas. Eeva and Lehikoinen (2000) demonstrated rapid recovery of passerine birds after the removal of heavy metal deposition. And presently, the use of birds as biomonitors has gained much more attention since 1999 due to the outbreak and subsequent spread of West Nile Virus.

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