PROCEEDINGS OF THE
14TH SYMPOSIUM
ON
THE NATURAL HISTORY OF
LOWER TENNESSEE AND CUMBERLAND RIVER VALLEYS

BRANDON SPRING GROUP CAMP

LAND BETWEEN THE LAKES

MARCH 25 AND 26, 2011

SPONS0RED BY

AUSTIN PEAY STATE UNIVERSITY

THE CENTER OF EXCELLENCE FOR FIELD BIOLOGY

AND

MURRAY STATE UNIVERSITY

WATERSHED STUDIES INSTITUTE

AND

U.S. DEPARTMENT OF AGRICULTURE, FOREST SERVICE

LAND BETWEEN THE LAKES NATIONAL RECREATION AREA

Austin Peay
State University

Center of Excellence
for Field Biology

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Sponsored by:

The Center of Excellence for Field Biology
Austin Peay State University, Clarksville, Tennessee

and

Watershed Studies Institute
Murray State University, Murray, Kentucky

and

U.S. Department of Agriculture, Forest Service
Land Between The Lakes National Recreation Area
Golden Pond, Kentucky

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PREFACE

At 1:00 p.m. on March 25, 2011, the 14th Symposium on the Natural History of Lower Tennessee and Cumberland River Valleys was convened at the Land Between The Lakes' Brandon Spring Group Camp by Dr. Steven W. Hamilton, Director of the Center of Excellence for Field Biology (CEFB). Following welcoming remarks by Dr. Hamilton on behalf of the CEFB and Austin Peay State University, Dr. Howard H. Whiteman, Director of the Watershed Studies Institute, welcomed guests on behalf of the Institute and Murray State University. Speaking on behalf of Dr. Bill Lisowski, USDA-Forest Service Area Supervisor for Land Between The Lakes (LBL), Wildlife Program Manager Steve Bloemer, welcomed attendees and provided updates on recent activities at LBL.

Following opening remarks from sponsoring organizations, Dr. Rebecca Johansen convened the Invited Papers Session, "Invasive Species of the Mid-South." Dr. Daniel Simberloff, Professor of Ecology and Evolutionary Biology at the University of Tennessee – Knoxville, presented a broad perspective on invasive species titled, "Biological Invasions – A Major Global Change and a Technological and Political Challenge." Dr. Michael Netherland, a Research Biologist for the U.S. Army Engineer Research and Development Center in Gainesville, Florida, followed with a more focused talk, "Biology and Management of Invasive Aquatic Plants in the Southeastern United States." Continuing on the aquatic invasive theme, Dr. Pam Fuller, Program Leader for the USGS Nonindigenous Aquatic Species Program in Gainesville, Florida, provided an overview of a system to track invasive aquatics with "USGS Nonindigenous Aquatic Species Database with a Focus on the Introduced Fishes of the Lower Tennessee and Cumberland Drainages." Closing out the Friday afternoon session was Dr. James Rhea, Entomologist with USDA Forest Service, Forest Health Protection in Asheville, North Carolina. Dr. Rhea's talk covered the threats to our forest communities in "Forests Under Attack by Non-Native Invasive Insects." Drs. Simberloff, Netherland and Fuller have published their reports in these proceedings. Dr. Rhea's presentation is summarized in an abstract. Dr. Johansen served as the editor for the Invited Paper Session.

On Saturday morning three contributed paper sessions were convened concurrently. Dr. Dwayne Estes moderated "Session I: Botany" and served as the botany editor for two papers and seven abstracts published in these proceeedings. Dr. Steven Hamilton served as moderator and editor for "Session II: Aquatic Biology" from which two short communications and eight abstracts are published here. Dr. Andrew Barrass moderated "Session III: Zoology" and edited one manuscript and nine abstracts from this session.

Editing and publishing of these proceedings follow the format of the prior thirteen proceedings published by the Center of Excellence for Field Biology. Contributed papers and abstracts were reviewed by CEFB staff for style, content and scientific merit. On occasion, external reviewers were asked to comment and critique contributed papers. Invited papers were reviewed by the session editor for style and content. Also, each manuscript was reviewed by one outside reviewer who could offer critical evaluation scientific merit. Thanks are extended to the external reviewers for their assistance and to all contributors for their fine work and patience while the proceedings were being completed. We hope the results are satisfactory both for the authors and for other readers. We apologize in advance for any oversights in the editing and publishing process.

ACKNOWLEDGMENTS

Jean Langley, technical clerk for the Center of Excellence for Field Biology, has been a critical component in the organization and execution of the meeting and publication of these proceedings. Without her diligent assistance and organizational skills, these results would be much diminished. We must also acknowledge the many undergraduate and graduate students who helped us prepare, set-up, operate and tear-down the symposium. Lastly, we thank Richard Lomax and his staff at the Brandon Spring Group Camp for the help and hospitality during this biennial event.

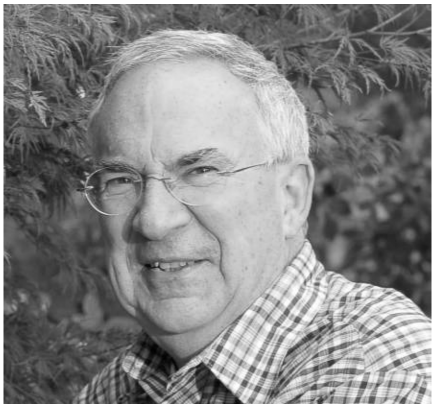
SYMPOSIUM REGISTRANTS

Following, in alphabetical order, is a list of those individuals who registered at the 2011 symposium. Institutional affiliation (when available), city (of the person's institution or home), and state are also given.

Tom Anderson – Murray State University (MSU), Murray, KY; Nathaniel Angus – Austin Peay State University (APSU), Clarksville, TN; Robert Arndt – United States Forest Servic (USFS), Golden Pond, KY: Andrew N. Barrass – APSU, Clarksville, TN; Tom Blanchard – University of Tennessee – Martin (UTM), Martin, TN; Stephanie Brandt – Kentucky Fish and Wildlife Service (KFWS), Frankfort, KY; Matt Bruton – APSU, Clarksville, TN; Angelo Bufalino – St. Louis, MO; Willodean Burton – APSU, Clarksville, TN; Brianna Cassidy – MSU, Murray, KY; Julie Cassidy – U.S. Coast Guard, Harding, KY; Edward W. Chester – APSU, Clarksville, TN; Brett Davis – MSU, Murray, KY; Hal De Selm – University of Tennessee (UTK), Knoxville, TN; Kathy DeWein – APSU, Clarksville, TN; Britny Edwards – APSU, Clarksville, TN; Dwayne Estes – APSU, Clarksville, TN; Ashlie Farmer – APSU, Clarksville, TN; Hannah Farrar – APSU, Clarksville, TN; Robert Ferree – APSU, Clarksville, TN; Michael Flinn – MSU, Murray, TN; Angel Fowler – APSU, Clarksville, TN; James Fralish – Southern Illinois University, Carbondale (SIUC), IL; Daniel French – Lambuth College, Jackson, TN; Michael Fulbright – APSU, Clarksville, TN; Pam Fuller – United States Geological Survey (USGS), Gainesville, FL; Claire Fuller – MSU, Murray, KY; Courtney Gorman – APSU, Clarksville, TN; Mark Gudin – Tennessee Wildlife Resources Agency (TWRA), Nashville, TN; Caitlin Gussenloven – UTM, Martin, TN; Cabrina Hamilton – Western Kentucky University (WKU), Bowling Green, KY: Debbie Hamilton – Allensville, KY; Steven W. Hamilton – APSU, Clarksville, TN; Iin Handayani – MSU, Murray, KY; Angelica Harris – APSU, Clarksville, TN; Megan Hart – APSU, Clarksville, TN; Barry Hart – Cumberland City, TN; Kate He – MSU, Murray, KY; Lisa Henning – APSU, Clarksville, TN; Kaitlyn Hill – APSU, Clarksville, TN; Mark Hoger – APSU, Clarksville, TN; Terry Hopkins – APSU, Clarksville, TN; Thomas Hulsey – WKU, Bowling Green, KY; Rebecca Johansen – APSU, Clarksville, TN; Shaniava Johnson – APSU, Clarksville, TN; Brittny Jones – APSU, Clarksville, TN; Jeff Kampman – MSU, Murray, KY; Kristen King – APSU, Clarksville, TN; George Kipphat – MSU, Murray, KY; Clea Klagstad – APSU,, Clarksville, TN; Robert Knapp – MSU, Murray, KY; Morgan Kurz – APSU, Clarksville, TN; Jean Langley – APSU, Clarksville, TN; Katie LeBlanc – MSU, Murray, KY; Todd Levine – MSU, Murray, KY; Matthew Lipscomb – Lambuth, Jackson, TN; Bill Lisowsky – USFS, Golden Pond, KY; Bommanna Loganfathan – MSU, Murray, KY; Joe Martin – APSU, Clarksville, TN; Seth McCormick – APSU, Clarksville, TN; Albert Meier – WKU, Bowling Green, KY: Michael Netherland – US Army Engineer Research and Development Center, Gainesville, FL; Chris O'Bryan – APSU, Clarksville, TN; Nathan Parker – APSU, Clarksville, TN; Andy Radonski – United States Fish and Wildlife Service (USFWS), Benton, KY; Elizabeth Rakes – USFS; Kirk Raper – MSU, Murray, KY; Judy Redden – UTM, Martin, TN; Rusty Rhea – USDA Forest Service, Asheville, NC; Andrew Riggs – APSU, Clarksville, TN; Michelle Rogers – APSU, Clarksville, TN; Nissa Rudh – MSU, Murray, KY; Dayle Saar – MSU, Murray, KY; Joe Schiller – APSU, Clarksville, TN; Josh Schulte – APSU, Clarksville, TN; A. Floyd Scott – APSU, Clarksville, TN; Daniel Simberloft – UTK, Knoxville, TN; Beth Slade – APSU, Clarksville, TN; Jimmy Smith – Tennessee Department of Environment and Conservatoin, Nashville, TN; Shanora Solomon – UTM, Martin, TN; Coy St. Clair – MSU, Murray, KY; James Stewart – UTM, Martin, TN; Brittney Suther – APSU, Clarksville, TN; Matthew Thomas – KFWS, Frankfort, KY; Tom Timmons – MSU, Murray, KY; Elizabeth Torkelson – Greenbriar, TN; Jack Torkelson – Greenbriar, TN; Emily Towery – APSU, Clarksville, TN; Matthew Wagner – APSU, Clarksville, TN; Michael Washburn – MSU, Murray, KY; Brenda Webb – Florence, AL; David Webb – Florence, AL; Sara Wigginton – WKU, Bowling Green, KY: Preston Wright – Lambuth, Jackson, TN; Kiah York – APSU, Clarksville, TN; David White – Hancock Biological Reserch Station, Murray, KY; Howard Whiteman – MSU, Murray, KY; Amanda Whitley – APSU, Clarksville, TN; David Whithers – Nashville, TN; Dawn Wilkins – UTM, Martin, TN; Prasanthi Yernari – MSU, Murray, KY.

SYMPOSIUM SPEAKERS

Invited Speakers



Daniel Simberloff
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Michael D. Netherland
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Pam Fuller
U.S. Geological Survey
Gainesville, Fla.

James “Rusty” Rhea
USDA Forest Service
Asheville, N.C.

Contributed Papers

Session I: Botany – (from left) Dwayne Estes, Hal De Selm, Courtney Gorman, Matt S. Bruton, Kate He, Dayle E. Saar, and Edward W. Chester.

Session II: Aquatic Biology – (from left) Todd Levine, Jeff Kampman, Angel Fowler, Coy St. Clair, Brianna Cassidy, Ashlie Farmer, Jimmy Smith, and Sara Wigginton.

Session III: Zoology – (from left) Andrew Barrass, Morgan Kurz, Kirk Raper, Cabrina Hamilton, Chris O'Bryan, Seth McCormick, Andy Riggs, Rebecca Johansen, Tom Anderson, and Brett Davis.

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INVITED PAPERS

INVASIVE SPECIES OF THE MID-SOUTH

Friday, March 25, 2011

Moderated by:

Rebecca Johansen
The Center of Excellence for Field Biology
Austin Peay State University

BIOLOGICAL INVASIONS – A MAJOR GLOBAL CHANGE AND A TECHNOLOGICAL AND POLITICAL CHALLENGE

DANIEL SIMBERLOFF

Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, TN 37996

ABSTRACT. The impacts of introduced species are varied, sometimes subtle, and often idiosyncratic. Nevertheless, most impacts fall into one of a few categories, such as predation, herbivory, disease, and hybridization. Effects of an introduced species can ripple through an entire ecosystem, as when an introduced pathogen or phytophage eliminates a dominant tree species or an introduced nitrogen-fixer fertilizes an entire ecosystem. The sheer number of invasions and variety and depth of effects often induces pessimism, a sense that fighting them can at best delay an inevitable onslaught. However, many invasions have been either eliminated or at least controlled. These successes are often unpublicized and include striking, clever uses of cutting-edge biology, but also some surprisingly effective simple, crude approaches. There is no scientific or technical reason why these approaches cannot be adapted to many more invaders, and often cost is not an impediment. The biggest problem is marshalling the political and social support to implement an effective, comprehensive operational structure, including early-warning/rapid-response capability. Another hindrance is posed by a few ecologists and other critics who have recently disparaged the threat posed by introduced species, arguing that most are innocuous, and actions against them are simply a form of xenophobia. They are wrong – introduced species are one of the great global changes – but critics can attract a following among those who profit from introducing species and resource managers who want to avoid dealing with them.

INTRODUCTION

Biological invasions occur when a species arrives (usually aided by humans) in a new region, spreads, and establishes populations that persist unaided in non-anthropogenic habitats. Many introduced species die out, and only a fraction of those that survive become invasive. Many introduced ornamental plant species, for instance, persist in gardens but do not spread into natural habitats. However, for some groups of species, a large fraction of introduced species become invasive. For example, of freshwater fish, mammal, and bird species introduced from Europe to North America or vice-versa, more than one-fourth has become invasive (Jeschke and Strayer 2005). Biological invasions have caused or contributed to the extinction of many native species and threaten the existence of many others, as well as the integrity of many native ecosystems. In some areas – especially islands – introduced species comprise a large fraction of the biota. In the Hawaiian Islands, most freshwater fish species, almost half of all plant species, 40% of bird species, and 25% of insect species are non-native.

IMPACTS

Biological invasions have many different kinds of impacts, and the impact of a particular invasion can depend on which other native and introduced species are present. Impacts are idiosyncratic and run the gamut from affecting one native species to modifying entire ecosystems, but certain impacts are especially common.

i. Ecosystem transformation

Some invaders (usually plants) modify habitat structure, thereby affecting many native species. In the Florida Everglades, Australian paperbark (*Melaleuca quinquenervia*) has replaced diverse grass- and sedge-dominated prairies with nearly monospecific forests of paperbark trees (Schmitz et al. 1997). By contrast, in Tierra del Fuego, North American beavers (*Castor canadensis*) introduced in 1946 now number over 100,000 and have converted forests of southern beech (*Nothofagus* spp.) to grass- and sedge-dominated meadows (Lizarralde et al. 2004). Invasive plants can change an entire ecosystem simply by overgrowing native species, as the tropical Pacific alga *Caulerpa taxifolia* has overgrown seagrass meadows over thousands of hectares in the Mediterranean, with myriad follow-on effects on the animal community (Meinesz 1999). Similarly, South American water hyacinth (*Eichhornia crassipes*) now blankets substantial lake and river areas in Africa (Matthews and Brand

2004a), the southeastern United States (Schardt 1997), and Asia and Australia (Matthews and Brand 2004b), smothering native submersed vegetation while decaying water hyacinth causes dissolved oxygen to decline, affecting native animal species.

In late 19th century England, hybridization between native cordgrass (*Spartina maritima*) and introduced North American *S. alterniflora* produced a new species (*S. anglica*) that invaded tidal mudflats, trapped sediment, increased elevation, and converted mudflats to dense salt marshes with different animal species (Thompson 1991). *Spartina anglica* had similar impacts when it was subsequently introduced to New Zealand and Washington State (U.S.A.).

Invasive plants can affect entire ecosystems by changing nutrient or water regimes. For instance, on the young, nitrogen-poor volcanic island of Hawaii, firetree (*Morella faya*), an Atlantic nitrogen-fixing shrub, has increased soil nitrogen so that other introduced plants that previously could not thrive are replacing native plants that had evolved to tolerate nutrient-poor soil (Vitousek 1986). In Israel, eucalyptus trees were deliberately introduced from Australia to drain swamps (Calder 2002), while Mediterranean salt cedars (*Tamarix* spp.) have dried up a marsh in California (McDaniel et al. 2005).

Invasions can also affect entire ecosystems by modifying the fire regime; Australian paperbark trees have replaced native grassland in Florida chiefly because paperbark ignites easily and burns hotter than the plants it replaces. The opposite shift can also occur by an introduced species that changes a fire regime: forest can be converted to grassland. African molassesgrass (*Melinis minutiflora*) and tropical American tufted beardgrass (*Schizachyrium condensatum*) in Hawaii have replaced native woodland by increasing fire frequency and extent (D'Antonio and Vitousek 1992).

Introduced pathogens and insects can affect an entire ecosystem by attacking the dominant native plant. In the early twentieth century, Asian chestnut blight fungus (*Cryphonectria parasitica*) invaded eastern North America, almost totally eliminating American chestnut (*Castanea dentata*), which in some areas had comprised 30% of canopy trees (Williamson 1996, Freinkel 2007). Loss of chestnut led to changes in forest structure and greatly affected nutrient cycling, because chestnut wood decomposes slowly, while the leaves decompose very rapidly (Ellison et al. 2005); leaves of oaks (*Quercus* spp.) that largely replaced chestnut decompose slowly. Because chestnut blight invaded before the era of modern ecology, its impacts were not well documented at the time, but at least seven moth species host-specific to chestnut went extinct (Opler 1978).

ii. Predation and aggression

The most obvious impact of introduced species is predation on native species. The brown tree snake (*Boiga irregularis*), which hitchhiked to Guam from New Guinea in cargo after World War II, has caused the global extinction or local extirpation of nine of the twelve native forest bird species and two of the eleven native lizard species (Lockwood et al. 2007) of Guam. Recent introduction and spread of the Burmese python (*Python molurus bivittatus*) in south Florida threatens many native animal species, which have no prior experience with such a large top carnivore (Reed and Rodda 2009). The python has preyed on large alligators in Florida. Another large introduced predator that has devastated native species is the Nile Perch (*Lates niloticus*), introduced with disastrous consequences to Lake Victoria in the 1950s to found a fishery that would provide food and jobs to local communities (Pringle 2011). Lake Victoria houses a great evolutionary species radiation of hundreds of cichlid fish species. Predation by the Nile Perch has already globally extinguished about half of these, and others survive only in captivity (Lockwood et al. 2007). The Nile Perch fishery is dominated by wealthy European exporters, while native fishing communities are devastated socially and economically (Pringle 2011).

Although large predators like the brown tree snake, Burmese python, and Nile Perch are highly visible to the public, even more consequential are invasions by small mammalian predators such as the ship rat (*Rattus rattus*), Norway rat (*R. norvegicus*), Pacific rat (*R. exulans*), small Indian mongoose (*Herpestes auropunctatus*), and stoat (*Mustela erminea*) on islands that formerly lacked such species. Native bird species had evolved the behavior of nesting on the ground and are highly susceptible to such interlopers. For example, invading rats have globally extinguished at least 37 species and subspecies of island birds throughout the world (Atkinson 1985), while the small Indian mongoose has been wholly or partially responsible for the endangerment and extinction of island birds, mammals, reptiles, and amphibians (Hays 2011).

Several predators introduced for biological control of previously introduced species (see below) have greatly reduced invasive populations of their target species. Unfortunately, several predators introduced for biological control have inadvertently caused extinction of non-target species. For instance, the small Indian mongoose (*Herpestes auropunctatus*) caused or contributed to the extinction of several island species of birds, mammals, and frogs in the Pacific and Caribbean after it was introduced as a biological control agent for introduced rats (Hays 2011). Similarly, the rosy wolf snail (*Euglandina rosea*) of Central America and Florida, introduced to many Pacific islands to control the introduced giant African snail (*Achatina fulica*), has caused the extinction of over 50 species of native land snails (Cowie 2002). The fish *Gambusia affinis* from Mexico and Central America, introduced to Europe, Asia, Africa, Australia, and many islands to control mosquitoes, preys on native invertebrates and small fishes and is implicated in the extinction of several fish species (Pyke 2008).

Introduced animal species can attack native species, whether or not they eat them. For instance, the South American red imported fire ant (*Solenopsis invicta*), invading the southeastern United States and California, attacks other ant species, and this aggression has caused great declines in populations of native ant species in disturbed habitats (Tschinkel 2006). The South American Argentine ant (*Linepithema humile*), an invader of North America, Europe, and Japan, reduces native ant populations by attacking them (Holway and Suarez 2004). Even without attacking, some animal invaders behave in ways that devastate native species. The Eurasian zebra mussel (*Dreissena polymorpha*), expanding in eastern North America, settles on native freshwater bivalve species and stitches their valves together with byssal threads, so they suffocate or starve, threatening several species with extinction (Ricciardi et al. 1998).

Introduced plants can inhibit other species by producing or sequestering chemicals. For instance, in prairies of western North America, Eurasian diffuse knapweed (*Centaurea diffusa*) and European spotted knapweed (*Centuaurea stoebe*) dominate native plants partly by producing toxic root exudates (Callaway and Ridenour 2004). The African crystalline ice plant (*Mesembryanthemum crystallinum*) retains salt, and decomposing fallen leaves salinate the soil to the detriment of native salt-intolerant plants in California (Vivrette and Muller 1977). Plant invaders can also dominate a native species by interfering with a necessary symbiont of the latter. For example, many plants have mutualistic arbuscular mycorrhizal fungi; the fungal hyphae penetrate the plants' root cells and aid in taking up soil nutrients. Garlic mustard (*Alliaria petiolata*), from Europe, Asia, and North Africa, lacks mycorrhizal associates but produces root exudates that are toxic to arbuscular mycorrhizal fungi found in North American soils (Callaway et al. 2008), aiding its invasion of many North American woodlands and floodplains.

iii. Herbivory

Invasive introduced herbivores often damage the flora of their new homes, particularly on islands. For instance, goats (*Capra aegagrus hircus*) introduced to St. Helena in 1513 eliminated at least half of the native plant species before botanists had the opportunity to record them (Cronk 1989). Invading European rabbits (*Oryctolagus cuniculus*) have devastated plant populations on many islands, killing shrubs and seedling and sapling trees by stripping bark. Having destroyed vegetation, rabbits often also exacerbate the situation by causing extensive erosion through digging (Thompson and King 1994).

Invasive introduced herbivores greatly damage crops and forests. The destruction of over 600,000 ha of French vineyards by the North American grape root louse phylloxera (*Daktulospharaira vitifoliae*) had drastic economic and social impacts, including abandonment of entire communities that had depended on viticulture (Pouget 1990). The South American cassava mealybug (*Phenacoccus manihoti*) often destroys more than half the cassava crop in Africa (Norgaard 1988); in the United States, the Russian wheat aphid (*Diuraphis noxia*) caused $600 million damage in only three years (Office of Technology Assessment 1993). The European gypsy moth (*Lymantria dispar*) caused a similar amount of damage to forests of the eastern United States in one year (Office of Technology Assessment 1993). The Asian balsam woolly adelgid (*Adelges piceae*) has effectively eliminated the previously dominant Fraser fir tree (*Abies fraseri*) in high elevation forests of the southern Appalachians (Rabenold et al. 1998), and the related Asian hemlock woolly adelgid (*Adelges tsugae*) is killing most hemlock (*Tsuga*) trees in eastern North America (Ellison et al. 2005).

iv. Pathogens

Introduced pathogens have often modified entire ecosystems, as in the chestnut blight example discussed above. An animal pathogen example is rinderpest (*Morbillivirus*), a viral disease of ungulates introduced to southern Africa from Arabia or India in cattle in the 1890s that devastated many native ungulate populations. The geographic range of some ungulate species in Africa is still affected by rinderpest. Because ungulates often determine vegetation structure and dynamics, rinderpest affects entire ecosystems (Plowright 1982).

Many introduced pathogens have affected particular native species or groups of them without changing an entire ecosystem. For example, avian malaria, caused by *Plasmodium relictum capristranoae*, was introduced to the Hawaiian Islands with Asian birds and vectored by introduced mosquitoes. It contributed to the extinction of several native Hawaiian birds and continues to restrict many Hawaiian birds to upper elevations, where mosquitoes are absent or scarce (Woodworth et al. 2005). Crayfish plague (*Aphanomyces astaci*), introduced to Europe with the North American red signal crayfish (*Pacifastacus lenusculus*) and also vectored by the subsequently introduced Louisiana crayfish (*Procambarus clarkii*), has devastated native European crayfish populations (Goodell et al. 2000). Likewise, the European fish parasite *Myxosoma cerebralis*, which causes whirling disease in salmonid fishes, infected North American Rainbow Trout (*Oncorhynchus mykiss*) that had been introduced to Europe. Infected frozen Rainbow Trout shipped to North America allowed the parasite to reach a trout hatchery in Pennsylvania, from which infected trout were sent to many western states. Sport fisheries of Rainbow Trout were completely destroyed in some areas (Bergersen and Anderson 1997). Introduced plant parasites can also affect agriculture. For instance, parasitic witchweed (*Striga asiatica*) from Africa arrived in the southeastern United States after World War II and inflicts great losses on grass crops (including corn); it has been the focus of a long eradication campaign (Eplee 2001).

Forest industries based on introduced tree species as well as natives are also threatened by invasive pathogens and insects. For example, Chilean plantations of North American Monterey pine (*Pinus radiata*) totaling ca. 2,000,000 ha are threatened by the recently arrived pathogenic oomycete *Phytophthora pinifolia* (Durán et al. 2008).

Introduced vectors can spread native pathogens as well as introduced ones. The native trematode *Cyathocotyle bushiensis*, a parasite of ducks, has recently reached new areas along the St. Lawrence River as its introduced intermediate host, the Eurasian faucet snail (*Bithynia tentaculata*), has spread (Sauer et al. 2007). The interaction between introduced parasites or pathogens and vectors can be complex but nonetheless devastating. Chinese Grass Carp (*Ctenopharyngodon idella*) infected with the Asian tapeworm *Bothriocephalus acheilognathi* were deliberately introduced to Arkansas in 1968 to control introduced aquatic plants. They subsequently invaded the Mississippi River, where the tapeworm infected the native Red Shiner (*Notropis lutrensis*), a bait fish. Fishermen or bait dealers then introduced infected Red Shiners to the Colorado River, and by 1984 they had arrived in a Utah tributary, the Virgin River. In the Virgin River, the tapeworm killed many Woundfin (*Plagopterus argentissimus*), a native minnow already threatened by dams and hydrological projects (Moyle 1993).

v. Competition

Introduced species are often said to "outcompete" natives, but cases demonstrating resource competition are scarce. A well-studied example is that of the introduced North American gray squirrel (*Sciurus carolinensis*) in Great Britain, where it forages for nuts more efficiently than the native red squirrel (*S. vulgaris*). This competition has led to the decline of the red squirrel (Williamson 1996); however, in addition the gray squirrel carried a disease, *Parapox* virus, to which they are resistant and red squirrels are highly susceptible, exacerbating the decline of the native species (Gurnell et al. 2004). The North American gray squirrel species has recently invaded the Piedmont in Italy and is spreading, raising fears for the future of the red squirrel on the European continent (Bertolino et al. 2008).

vi. Hybridization and introgression

A subtle but nevertheless frequently devastating impact of introduced species can occur when the invader is sufficiently closely related to a native species that they can mate, particularly if the offspring are fertile so that backcrossing (introgression) occurs in the native population. In Australia and New Zealand, where most of the native species separated evolutionarily from those of other regions very long ago, hybridization with invaders is less frequently a problem (though cases exist; see below). However, many species in Europe and North America are more closely related to one another, and within single continents the degree of relationship may be even greater, so that hybridization is more likely. A native species may be so changed by hybridization and introgression with an invader that one can view the native as having undergone “genetic extinction.”

This impact is especially likely when the invading population greatly outnumbers the native, so that native individuals are more likely to mate with the invaders. For instance, a narrowly distributed fish in Texas, *Gambusia amistadensis*, was hybridized to extinction by interbreeding with *Gambusia affinis* introduced for mosquito control (Hubbs and Jensen 1984). Several native fish species of the American West currently on the United States Endangered Species List are threatened at least partly by hybridization with Rainbow Trout, native to the West but introduced to many new regions for sport fishing. Hybridization has been a factor in 38% of all native fish extinctions in North America over the 20th century (MacIsaac 2011). The rusty crayfish *Orconectes rusticus*, native to the Ohio River Valley, has been commonly used as bait and has been introduced in much of eastern North America through the sport fishing industry; it threatens the existence of the native northern clearwater crayfish *Orconectes propinquus* by hybridization (Perry et al. 2002).

The North American mallard (*Anas platyrhynchos*), widely introduced as a game bird, has hybridized with several native congeneric species. It is a threat to the genetic integrity of the endemic New Zealand grey duck (*Anas superciliosa superciliosa*) and the Hawaiian duck (*Anas wyvilliana*), as well as, possibly, the yellowbilled duck (*Anas undulata*) and the Cape shoveller (*Anas smithii*) in Africa (Rhymer and Simberloff 1996, Matthews and Brand 2004a). In Europe, populations of the white-headed duck (*Oxyura leucocephala*) are found only in Spain, where they are threatened by hybridization and introgression with invading North American ruddy ducks (*Oxyura jamaicensis*) (Muñoz-Fuentes et al. 2007). The ruddy duck had been introduced years earlier to Great Britain as an amenity but crossed the English Channel and spread through France to Spain.

Introgression from one species can threaten another species even if the two do not hybridize! In the early 20th century, the White Sucker (*Catostomus commersoni*), a fish native to areas east of the North American Continental Divide, was introduced to the Colorado River (west of the Continental Divide), where the Flannelmouth Sucker (*Catostomus latipinnis*) and the Bluehead Sucker (*Catostomus discobolus*) are native and commonly occur sympatrically without hybridizing. Both species are now hybridizing with the introduced White Sucker, leading to introgression between the two native species, producing a hybrid swarm with genes from all three species (McDonald et al. 2008).

Improved molecular genetic techniques have shown that hybridization and introgression between introduced and native species are much more common than had been realized. Hybridization between introduced and native species has even led to the formation of new species, as in the case of *Spartina anglica* discussed above. Similarly, Oxford ragwort (*Senecio squalidus*), a hybrid of two species from Italy, was introduced to the Oxford Botanical Garden ca. 1690 and escaped. A doubling of chromosome number in a hybrid between *Senecio squalidus* and *Senecio vulgaris* (groundsel) produced the new polyploid species *Senecio cambrensis* (Welsh groundsel) (Ashton and Abbott 1992).

Hybridization can threaten species even when there is no introgression. The European mink (*Mustela lutreola*) is threatened by habitat destruction and pollution in much of its range, while North American mink

(*Neovison vison*) has been raised in Europe as a furbearer. American mink have escaped in several regions and established invasive populations. American mink males become sexually active well before European mink males and mate with female European mink. The females eventually abort the hybrid embryos, so there is no introgression, but these females are removed from the breeding population for a season, a major handicap for a small, declining population (Maran and Henttonen 1995).

vii. Chain reactions and invasional meltdown

Impacts of biological invasions can result from chains of interactions between species: species A affecting species B, then species B affecting species C, species C affecting species D, and so forth. An example is the case described above involving introduced Grass Carp, an introduced tapeworm, and the introduced Red Shiner ultimately affecting the native Woundfin minnow. A more complex chain involves the devastation of European rabbit populations in Great Britain by New World *Myxoma* virus, described above. Caterpillars of the native large blue butterfly (*Maculina arion*) in Great Britain develop in underground nests of a native ant, *Myrmica sabuleti*. This ant does not nest in overgrown areas, and grazing and cultivation had produced open areas for many centuries. However, changed land use patterns and decreased grazing in the 20th century caused rabbits to be the primary species maintaining open habitat suitable for the ant. When the virus devastated rabbit populations, ant populations crashed and the large blue butterfly was extirpated from Great Britain (Ratcliffe 1979).

A remarkable chain reaction began when landlocked Kokanee Salmon (*Oncorhynchus nerka*), introduced to Flathead Lake, Montana in 1916, replaced most native Cutthroat Trout (*Oncorhynchus clarki*) and became the main sport fish. The Kokanee dispersed far from the lake, and their spawning populations became so large that bald eagles (*Haliaeetus leucocephalus*), grizzly bears (*Ursus arctos horribilis*), and other predators congregated to feed on them. Between 1968 and 1975, opossum shrimp (*Mysis relicta*), native to deep lakes elsewhere in North America and in Sweden, were introduced to three lakes in the upper portion of the Flathead catchment to bolster Kokanee production. However, the shrimp drifted downstream into Flathead Lake and precipitated a drastic decline in populations of cladocerans and copepods on which the shrimp preyed. Because the Kokanee also fed on these prey, their populations fell rapidly, causing a sharp decline in local bald eagle and grizzly bear numbers (Spencer et al. 1991). The important take-home message of these chain reactions is that impacts of biological invasions are often complex, idiosyncratic, and would not have been predicted even by experts.

Invasive species may facilitate one another's existence or exacerbate one another's impacts, a phenomenon called "invasional meltdown" (Simberloff and Von Holle 1999, Simberloff 2006). The introduced faucet snail (*Bithynia tentaculata)* that vectors a native trematode parasite of ducks and thereby expands the trematode's range was discussed above. *Bithynia* also vectors a recently arrived European trematode (*Leyogonimus polyoon*), also lethal to ducks (Cole and Friend 1999). In sum, the introduced snail and the introduced trematode combine to generate higher duck mortality than either would have produced alone. In other instances of invasional meltdown, introduced animals pollinate introduced plants or disperse their seeds. For example, ornamental figs (*Ficus* spp.) introduced to Florida had never invaded natural areas because they could not produce seeds in the absence of their host-specific, pollinating fig wasps. However, the fig-wasps of three fig species eventually arrived ca. 20 years ago, and the figs became fertile, one species becoming invasive (Kauffman et al. 1991). A similar event occurred in New Zealand, where an ornamental Australian strangler fig, unable to reproduce, invaded native forest once its fig wasp pollinator arrived in 1996 (Towns et al. 2011).

The red-whiskered bulbul (*Pycnonotus jocosus*), introduced to the island of La Réunion from Asia via Mauritius, exacerbates invasions by several introduced plants, including *Rubus alceifolius*, *Cordia interruptus*, and *Ligustrum robustrum*, by dispersing their seeds (Baret et al. 2006). In Hawaii, the Asian common myna (*Acridotheres tristis*), introduced to control pasture insects, disperses the Caribbean weed *Lantana camara* throughout the lowlands and into native forests (Davis et al. 1993). Also in Hawaii, introduced pigs eat and then disperse several invasive introduced plant species, and their rooting and defecation also disperse several introduced invertebrates; the pigs are partly sustained by feeding on introduced European earthworms (Stone 1985).

Introduced plants can modify habitat so as to precipitate a meltdown process with expanded and/or accelerated impacts. The previously mentioned nitrogen-fixing *Morella faya* (firetree) from the Atlantic has invaded nitrogen-deficient volcanic regions of Hawaii. Firetree is essentially fertilizing large areas that previously had no nitrogen-fixers. Other introduced plants established elsewhere in Hawaii had been precluded from invading these previously nutrient-deficient areas, but the nitrogen fixation by firetree now facilitates their entry (Vitousek 1986). Soil under firetree also contains elevated populations of introduced earthworms, which increase the rate of nitrogen burial from firetree litter, thereby multiplying the effect on the nitrogen cycle (Aplet 1990). Introduced pigs and the introduced Japanese white-eye, *Zosterops japonicus*, disperse seeds of the firetree (Stone and Taylor 1984, Woodward et al. 1990). These introduced species act together to cause the replacement of native vegetation.

An important invasional meltdown in North America involves the introduced zebra mussel (Johnson 2011), which filters prodigious amounts of water, and invasive Eurasian watermilfoil (*Myriophyllum spicatum*), which is inhibited by cloudy water. The increase in water clarity wrought by the mussel aids the milfoil, while the milfoil aids the mussel by providing a settling surface and also helps the mussel disperse to new water bodies when plant fragments are transported on boat propellers or in water (Simberloff and Von Holle 1999).

Cases of invasional meltdown might have been predicted when one introduced species is later reunited with a subsequently introduced coevolved species. The fig species and their coevolved wasps are a good example. However, sometimes species that had never been found together facilitate one another when both are introduced, as with the Asian myna bird and New World *Lantana camara* in Hawaii. In such cases, the interaction and enhanced impact of the invasion would have been difficult to predict.

viii. Lags

Introduced species may remain innocuous for decades or even centuries before suddenly increasing in numbers and range to become highly invasive. The case of the hybrid cordgrass *Spartina anglica* is a prime example. The introduced parent species, North American *Spartina alterniflora*, was introduced to Great Britain at least by the early nineteenth century and had hybridized occasionally with the native *Spartina maritima*, but the hybrids were sterile until chromosome number of one doubled ca. 1891, yielding a highly invasive weed (Thompson 1991). Giant reed (*Arundo donax*) was introduced from the Mediterranean region to California in the early nineteenth century as a roofing material to stem erosion, but it remained restricted in range and its impact was minor until the mid-twentieth century, when it spread widely, becoming a fire hazard and damaging wetlands (Dudley 2000). The Caribbean brown anole lizard (*Anolis sagrei*) was introduced to Florida in the nineteenth century, but it remained in extreme south Florida until the 1940s, when its range began to expand. The spread accelerated in the 1970s and ultimately encompassed most of Florida (Kolbe et al. 2004).

Many invasion lags are mysterious. For example, there is as yet no cogent explanation for why giant reed in California took over a century to become invasive. In other cases, a change in the physical or biotic environment is the obvious explanation for a sudden explosion of an introduced species that had formerly been restricted. The sudden invasion by long-present figs in south Florida described above was triggered by the arrival of pollinating fig wasps. In other cases, demography is the explanation. For instance, trees have such a long life cycle that they cannot increase their population size rapidly even in a suitable environment.

The rapid development of molecular tools has enabled biologists to determine the sources of invasions with greater accuracy. With these tools some sudden expansions after a lag phase have been attributed to the introduction of new genotypes to a previously restricted population. The brown anole population long present in Florida was augmented in the twentieth century by new individuals from different parts of the native range, so that the population in Florida now has far more genetic diversity than that of any native population. The dramatic range expansion of this species may be due to introductions to new sites combined with the arrival of new genotypes better adapted to the range of environmental conditions found in Florida (Kolbe et al. 2004). A sudden northward range expansion of the European green crab (*Carcinus maenas*) along the North American Atlantic coast was caused by the arrival of new, cold-tolerant genotypes in the established population (Roman 2006).

The fact that some biological invasions entail lag times, and that we lack an understanding of why this is so for many of them, greatly complicates the prediction of impact and also decisions on whether an introduction that appears innocuous should be managed or eradication attempted. Is a "harmless," restricted population a ticking time bomb waiting to explode across the landscape?

MANAGEMENT OF INVASIVE SPECIES

The enormous economic and ecological damage inflicted by biological invasions has spawned many measures to combat them. The best approach is to prevent introductions in the first place. If an introduced species succeeds in establishing a population, it may be possible to find it quickly and to eradicate it. If an established population has already spread widely, it may still be possible (at great expense) to eradicate it, and it is likely that populations can at least be maintained at low enough levels that impacts are lessened.

i. Exclusion

Introductions are either planned or unplanned, and prevention entails somewhat different approaches for the two classes. In either case, the three primary tools are laws, risk analyses, and border security. For planned introductions, such as a new sport fish or ornamental plant, the minimal law would be a "black list" of species not permitted entry under any circumstances. Typical black list laws, such as the Lacey Act in the United States, have not been very effective because they tend to be reactive rather than proactive. That is, species are rarely put on black lists until they have already entered a country and become invasive. A better approach is a "white list" of species approved for introduction after a risk analysis that considers the traits of the species intended for introduction and the outcome in regions where it has been introduced. New Zealand and Australia use versions of white lists in permitting or excluding planned species introductions. Key to the effectiveness of white lists is accuracy of the underlying risk analyses, and risk analyses for introduced species have not proven very accurate, although much current research is devoted to improving them (e.g., Kolar and Lodge 2002). The underlying problem is that the interactions of an introduced species with all the species in the recipient community are complex and numerous; understanding them all would in most cases take years of study. Worse, introduced species can evolve, and evolution is an inherently unpredictable process.

Preventing unplanned introductions requires identifying and constricting the pathways by which they enter (Ruiz and Carlton 2003). Many marine species are accidentally transported in ballast water, while insects often hitchhike on ornamental plants or agricultural produce. The Asian longhorned beetle (*Anoplophora glabripennis)*, a devastating forest pest, came to North America and Austria in untreated wooden packing material from Asia, while the Asian tiger mosquito (*Aedes albopictus*) reached the United States in water transported in used tires. Snails have been inadvertently introduced worldwide on ceramics and paving stones. When the risks associated with these pathways have been recognized, technology may exist to constrict them, but it is often costly. For instance, agricultural products can be fumigated or frozen.

Finally, whatever laws and regulations exist for both planned and unplanned introductions, inspections at ports of entry are necessary to enforce them, and many detection technologies are available, including sniffer dogs and highly accurate X-ray equipment. However, detection will never be perfect, so penalties for violating regulations must be severe enough to deter potential scofflaws.

ii. Eradication

An effective early warning-rapid response system is important for eradication of established populations, as expense of eradication increases rapidly once an invader starts to spread. No nation has adequate monitoring to detect new invaders reliably, but informed citizens, including biologists, have frequently detected newly established populations. For example, the killer alga (*Caulerpa taxifolia*) that has ravaged the northwest Mediterranean was noticed within six months of arrival in California by a diver who had read about the European invasion. The Asian longhorned beetle invasion in Chicago was quickly found by a citizen gathering firewood who had seen news reports about the invasion in New York. Both of these species were successfully eradicated by well-organized projects lasting approximately four years.

Many other invaders have been successfully eradicated, particularly on islands. For example, rats have been eradicated from increasingly large islands – the largest so far is 113 km^2. Populations of goats and pigs recently have been eradicated from Santiago Island (585 km^2) in the Galapagos (Cruz et al. 2005). Insects frequently have been eradicated, including an African mosquito vector of malaria (*Anopheles arabiensis*) from a large area of northeastern Brazil (Davis and Garcia 1989). Plants with a soil seed bank present a particular challenge, but even these have sometimes been eradicated. For example, sand bur (*Cenchrus echinatus*) was eliminated from the 400-ha Laysan Island in the Hawaiian chain (Flint and Rehkemper 2002). Certain features distinguish successful eradication from the many failed attempts: (i) Sufficient resources to finish the project; the expense of finding and removing the last few individuals may exceed that of quickly ridding a site of the majority of the population. (ii) Clear lines of authority, so that an agency can compel cooperation. Eradication is, by its nature an all-or-none operation and can be subverted by a few individuals who do not cooperate (for instance, by preventing access to private property or forbidding use of a pesticide or herbicide). (iii) The target organism must be studied well enough that a vulnerable point in its life cycle is identified. (iv) If the eradication succeeds, there must be a reasonable prospect that reinvasion will not occur quickly.

Many methods have been used in these campaigns: males sterilized by X-rays for fruitflies, chemicals for *Anopheles gambiae* and for rats, hunters and dogs for goats. The Judas-animal technique has been applied to goats, pigs, and a few other animals that congregate. An individual of the invasive population is captured and fitted with a radio transmitter, then released to join others in the population. Hunters on foot or in aircraft then detect the radio signal and kill the newly located individuals. Some eradication projects that would likely have succeeded have been halted short of their goal by public objections to killing vertebrates or using chemicals. A notable example is the cessation, because of pressure from animal-rights groups, of the campaign to eradicate the gray squirrel before its spread in Italy (Bertolino and Genovesi 2003).

iii. Maintenance management

When invasive species cannot be eradicated, three main technologies can limit their populations: mechanical or physical control, chemical control, and biological control. Invasive plants often have been maintained at low levels by herbicides. For example, water hyacinth in Florida was greatly reduced and subsequently managed by use of the herbicide 2,4-D, combined with some mechanical removal (Schardt 1997), while in some parts of South Africa, a combination of mechanical and chemical control keeps populations of lantana well controlled (Matthews and Brand 2004a). A South African public works program, Working for Water, has used physical, mechanical, and chemical means to clear thousands of hectares of land of introduced plants that use great amounts of water, such as mesquite (*Prosopis* spp.) and several species of *Acacia* (Matthews and Brand 2004a). In the Canadian province of Alberta, Norway rats (*Rattus norvegicus*) have been maintained at very low levels for many years by a combination of poisons and hunting by the provincial Alberta Rat Patrol (Bourne 2000).

However, long-term use of herbicides and pesticides often becomes problematic. The target species may evolve resistance. This has happened with many insect pests of agriculture and recently arose with the herbicide used to control the aquatic Asian weed *Hydrilla verticillata* in Florida (Puri et al. 2007). Also, chemicals are often so costly that they cannot be used over large areas, particularly if the target invader is a conservation rather than agricultural or human health problem. Finally, chemicals often have impacts on non-target species, including humans. The decline of raptor populations as eggshells thinned because of DDT residues is an example (Lundholm 1997). Many newer herbicides and pesticides have few if any non-target impacts when used properly but may be very expensive.

These disadvantages of pesticides have provoked great interest in biological control – deliberately introducing a natural enemy (predator, parasite, or disease) of an invasive introduced pest. The classic example is the introduction of the Australian vedalia ladybeetle (*Rodolia cardinalis*) in 1889, which controlled the Australian cottony-cushion scale (*Icerya purchasi*) on California citrus (Evans and Snyder 2011). Only a minority of well-planned biological control projects actually succeed in controlling the target pest, but many of those that have succeeded are highly effective and have conferred low-cost control in perpetuity. For instance, a devastating invasion of water hyacinth in the Sepik River catchment of New Guinea was controlled by introducing two South American weevils that had been successfully used to this end in Lake Victoria, *Neochetina eichhorniae* and *N. bruchi* (Matthews and Brand 2004b). On the island of St. Helena, a tropical American scale insect (*Orthezia insignis*) that had threatened to eliminate the endemic gumwood tree (*Commidendrum robustum*) has been controlled by a predatory South American lady beetle, *Hyperaspis pantherina* (Booth et al. 2001).

Biological control projects often entail introducing herbivorous insects to attack terrestrial and aquatic weeds. In what was probably the first successful biological control project for a weed, a Brazilian cochineal insect (*Dactylopius ceylonicus*) effectively eliminated the smooth prickly pear (*Opuntia vulgaris*) in India (Doutt 1964). The same insect was introduced to South Africa in 1913 to attack the same plant with the same result (Doutt 1964). As noted above, in Lake Victoria, Africa, invasion by South American water hyacinth was controlled by introducing two South American weevils, *Neochetina eichhorniae* and *N. bruchi* (Matthews and Brand 2004a); the same weevils were introduced to control water hyacinth in tropical Asia (Matthews and Brand 2004b). The South American alligatorweed flea beetle (*Agasicles hygrophila*) has greatly reduced the invasion of South American alligatorweed (*Alternanthera philoxeroides*) in Florida (Center et al. 1997) and Asia (Matthews and Brand 2004b). One of the best known biological control successes was the introduction of the South American cactus moth (*Cactoblastis cactorum*) to Australia, where it ended a massive invasion of prickly pear cactus (*Opuntia* spp.) (Zimmermann et al. 2001).

Pathogens and parasites are also used in biological control of introduced hosts, often to great advantage. For instance, an introduced South American parasitic wasp, *Epidinocarsis lopezi*, has partially controlled the South American cassava mealybug in Africa, mentioned above (Norgaard 1988). However, because biological control agents and their targets are living organisms, they can evolve and thereby affect the interaction between them. The *Myxoma* virus was introduced from the Americas to control rabbits in continental Europe (where the European rabbit is native) and Great Britain and Australia (where it is introduced). Initially the project was a great success, with rabbit mortality over 90%, but eventually the virus evolved to be more benign, and in Great Britain and Australia rabbits also have evolved increased resistance to the virus, so each successive epizootic has lower mortality (Bartrip 2008). In Australia, this decline led to tests of a Chinese virus, the calicivirus, which causes rabbit hemorrhagic fever. As tests were conducted, the calicivirus escaped, then was deliberately released throughout Australia, triggering a massive epizootic (Frank et al. 2010).

A danger is that biological control agents can attack non-target species and even cause extinctions – the cases involving the rosy wolf snail, small Indian mongoose, and mosquitofish have been mentioned earlier. Biological control introductions of herbivorous insects have sometimes devastated non-target native species. The cactus moth that was so successful in Australia was introduced to control pest prickly pear on the island of Nevis in the West Indies. From there, it dispersed northward through the West Indies, arrived in Florida, and is now spreading further north and west. In Florida, it threatens the native semaphore cactus (*Opuntia corallicola*).If this moth reaches the American Southwest and Mexico, it may threaten other native *Opuntia* species and also affect economically important markets for ornamental and edible *Opuntia* (Zimmermann et al. 2001). In general, problems of this sort have been associated with introduced biological control agents such as generalized predators that are not specialists of the specific target host. However, even species that are restricted to a single host genus, such as the cactus moth, can create problems.

CONTRARIANS, PESSIMISM, AND POLICY

Although there is widespread recognition that invasive introduced species constitute a major conservation and economic problem, several commentators, including biologists, have argued that the extent of the problem is overblown and that attention and resources are being squandered in dealing with introduced species. These concerns stem from several directions.

Some critics wish to maintain a particular introduced species targeted for removal or oppose specific management activities. Such objections most frequently arise when mammal and bird invaders are targeted. As noted above, the campaign to stop the spread of the North American gray squirrel in Europe was halted because of objections from animal rights advocates. Animal rights advocates (e.g., Welch 1973) have similarly objected to campaigns in Florida, California, and other states to eradicate damaging populations of South American monk parakeets (*Myiopsitta monachus*). Hunters also object to removal of game animals. Probably the best known example is the long controversy in Hawaii over removal of introduced Asian pigs, European wild boar, and their hybrids, which greatly damage forests (and spread seeds of non-native plants) but are prized by hunters, including native Hawaiians (Burdick 2005). A similar controversy partially stymied a project to remove non-native mountain goats from Olympic National Park, Washington (Houston et al. 1994). Even some ecologically damaging non-native plants have passionate advocates, leading to controversies over plans to remove non-native eucalypts in California (Williams 2002), *Casuarina* (Australian "pine") in Florida (Ellis 1999), strawberry guava (*Psidium cattleianum*) in Hawaii (Tummons 2008), and buckthorn (*Rhamnus* spp.) in Illinois (Helford 2000).

Other opponents of eradication or management of invasive species see the entire enterprise as a displaced form of xenophobia (Wolschke-Bulmahn and Gröning 1992, Pollan 1994, Pauly 1996, Gould 1998, Sagoff 1999, Subramaniam 2001; cf. Simberloff 2003). Although some evidence suggests a xenophobic cast to certain arguments and activities of the late nineteenth and early twentieth centuries, there is scant indication that motives associated with the rise of modern invasion biology in the 1980s are other than ecological (Simberloff 2005, Coates 2006).

A small but vocal contingent of ecologists argues that most introduced species do not become highly invasive and that the attention paid to managing invasions is therefore out of proportion to the scale of the problem (Sax et al. 2002, Brown and Sax 2004, Gurevitch and Padilla 2004, Sax and Gaines 2008, Davis 2009). To an extent, this argument is a red herring, because management activities are already targeted narrowly at those introduced species known to be invasive or perceived as likely to become invasive. Also, to an extent this argument is simply based on inaccurate data (Clavero and García-Berthou 2005, Simberloff 2005). There is no indication that concern with introduced species has lessened because of this argument.

However, there is a danger. In general, agencies charged with managing any problem, especially ones that may be costly to manage, will seek reasons not to act, and the perception of a controversy, even one involving a small minority of persons with expertise, provides a pretext for inaction. The actions with respect to global climate change of many Republicans in the Bush administration and in Congress is sad testimony to this fact. Similarly with biological invasions, reporters are eager to describe controversy. The article by Sax and Gaines (2008) was highlighted in a New York Times article entitled "Friendly invaders" (Zimmer 2008), and Davis's views are favorably cited in a Scientific American feature entitled "Alien invasions? An ecologist doubts the impact of introduced species" (Borrell 2009). Davis's 2009 book was reviewed in Nature with the title "The end of the invasion?" (Marris 2009). Surely such descriptions in the popular press will not help to change the woefully inadequate response to invasions that characterizes almost all governments.

Finally, pessimism can stymie policies and management activities targeting invasions. Quammen (1988) captured this view succinctly when he suggested that the forces causing invasions – increased trade and travel – so outweigh the actions we can marshal against them that it is probably hopeless to combat them. The world will still be green and there will still be lots of species in most places a few centuries from now, but it will be the same species everywhere, the global weeds, both animal and plant weeds. Such an attitude leads to the question, "Why try?". Indeed, Davis (2009) advocates simply "learning to love them" as a generic response to invasions, while Mark Gardener, head of restoration at the Charles Darwin Research Station in the Galapagos says, "It's time to embrace the aliens" (Vince 2011). There is little doubt that policymakers will note this view, and it will not encourage them to act aggressively and quickly to deal with invasions.

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BIOLOGY AND MANAGEMENT OF INVASIVE AQUATIC PLANTS IN THE SOUTHEASTERN UNITED STATES

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ABSTRACT. Aside from natural lakes in Florida and some regions of Louisiana and Arkansas, the majority of public water bodies in the southeastern United States are regulated reservoirs that serve multi-purpose functions in providing flood control, power generation, water storage and domestic water supply, and recreational pursuits such as boating, fishing, and hunting. In most cases, these southeastern reservoirs do not have well-established native plant communities and have been particularly prone to introduction and spread of various invasive aquatic plants such as hydrilla (*Hydrilla verticillata),* Eurasian watermilfoil (*Myriophyllum spicatum*), water hyacinth (*Eichhornia crassipes*), alligatorweed (*Alternanthera philoxeroides*), and giant salvinia (*Salvinia molesta).* Given the broad spectrum of uses noted above, the attitudes of stakeholders towards allowing invasive plants to expand in a given system can range from zero tolerance to those who propose the need for at least 30 percent submersed plant coverage for habitat enhancement. The role of invasive plants in providing beneficial ecological services is a current and lively topic of debate between and within various resource management agencies and stakeholder groups in the southeastern U.S. Not surprisingly, attitudes towards invasive plants as well as subsequent management efforts to control these plants, tend to be complex, and over time they have evolved on a somewhat regional basis. In Tennessee and Kentucky, the aquatic plant currently showing the greatest ability for spread and establishment is the monoecious biotype of hydrilla. The biology of the hydrilla biotypes is discussed in the context of their ability to rapidly spread and predominate in large areas of southern waters. Key physiological, morphological, and reproductive traits of these invasive plants are described in terms of the potential for conferring competitive growth advantages or for allowing these plants to exploit resources in previously non-vegetated areas. The recent invasion of crested floating heart (*Nymphoides cristata)* in Florida and the Santee Cooper Lakes in South Carolina, serves as an illustrative example of a new species introduced from the aquarium trade that shows potential as a highly invasive aquatic plant. Lack of knowledge of the basic biology and management of this plant, as well as its potential to become invasive throughout the Southeast confounds our ability to develop a regional response plan. Other examples provided to demonstrate the complexity of invasive species and management efforts include utilizing hydrilla to support snails that provide preferred foraging habitat for the federally endangered snail kite on a large multi-purpose public lake in Florida. Hydrilla also provides a substrate for a toxic cyanobacterium that has been linked to deaths of waterfowl and raptors in several southeastern reservoirs. In addition, large unmanaged systems provide a significant propagule source and serve as "gateways" for spread of invasive species to other regions. This is discussed in the context of Kentucky Reservoir and the Ohio River, and the potential for spread of monoecious hydrilla into thousands of smaller glacial lakes in the upper Midwest.

INTRODUCTION

Invasive aquatic plants are prevalent throughout the Southeastern United States; however, there are fairly distinct differences in the distribution of these plants from the natural lakes of the peninsula of Florida, to southern coastal states, to interior regions such as Tennessee and Kentucky. Southeastern states such as AL, AR, GA, KY, LA, NC, SC, TN, and TX, are all rich in water resources; however, the majority of large water bodies in these states are regulated reservoirs. States such as Alabama and Georgia report no natural public lakes, while Tennessee recognizes Reelfoot Lake (formed via earthquakes in 1811 and 1812) as it's only natural lake. This has significant implications as reservoirs do not tend to support the same level of extensive native plant communities as natural lakes in Florida or the glacial lakes of the northern tier states. Environmental stresses such as extreme water level fluctuations, high flows that result in scouring of sediments, significant turbidity, and an abundance of generalist herbivores (e.g., turtles, muskrats) can prevent establishment of submersed and emergent native plant communities in many southeastern reservoirs (Smart et al. 1996, Netherland 2008). Given the values associated with aquatic vegetation (e.g., habitat for invertebrates and fish, substrate stabilization, nutrient sequestration, improved water clarity, improved fishing conditions and increased waterfowl usage), there have been numerous efforts to establish native plants in reservoirs (Smart et. al. 1996, Doyle et al. 1997, Toth 2007). While invasive plants face the same environmental constraints as the native plants, they often have adaptations that allow them to better tolerate the extremes associated with reservoirs. This has led many stakeholders and some resource management agencies to consider the potential benefits or ecological services afforded by invasive plants in areas that may support limited native plant communities (Rybicki et al. 2007, Hershner and Havens 2008, Hoyer et al. 2008).

Aquatic Invasive Plant Threats to Kentucky and Tennessee

While floating invasive plants such as water hyacinth (*Eichhornia crassipes*), water lettuce (*Pistia stratiotes*), and giant salvinia (*Salvinia molesta*) are recognized for their potential to rapidly spread in both natural lakes and man-made reservoirs, it is likely these plants will continue to be constrained by winter temperatures that preclude widespread and long term establishment in the Tennessee and Kentucky regions. Water hyacinth has been reported as problematic in the backwaters of the Tennessee Tombigbee waterway in northern Mississippi**,** but there are few reports of ongoing management issues with floating plants in Tennessee and Kentucky. Likewise, there are also few emergent invasive plants that have demonstrated a rapid spread or required significant management in this region. The northward expansion of plants such as alligatorweed (*Alternanthera philloxeroides*) is of concern due to the reduced efficacy of the biocontrol insect (*Agasicles hygrophila)* in more temperate areas. It is interesting to note the US Army Corps of Engineers Jacksonville District harvests the alligatorweed flea beetles from Florida in the spring and on request ships them to resource managers in more temperate regions to aid in biological control of this plant. The potential for southern migration of invasive plants that cause problems in high-flow northern waters such as flowering rush (*Butomus umbellatus*) should also be tracked (Bellaud 2009). The current spread of an invasive haplotype of phragmites (*Phragmites australis*) throughout the Great Lakes states and in southern coastal regions suggests this plant may have a broad range of environmental tolerance that would allow it to be invasive in the Tennessee and Kentucky regions (Whetstone 2009).

The invasive submersed plant Eurasian watermilfoil (*Myriophyllum spicatum*) was problematic in the Tennessee Valley Authority (TVA) reservoirs as early as the 1960s**,** and much of the early literature describing efficacy of drawdowns and large-scale herbicide applications for control of this plant came from TVA reservoirs (Smith 1971). To give a perspective of the scale of infestation, from April through June 1969, over 17,000 acres of Guntersville and Nickajack reservoirs were treated with 2,4-D to control Eurasian watermilfoil. As herbicides and drawdowns had reduced Eurasian watermilfoil in the TVA reservoirs, the introduction of the dioecious biotype of hydrilla resulted in a rapid expansion to a few thousand acres on Guntersville Reservoir in the late 1980s (TVA & USACE 1992). It was noted that hydrilla predominated in areas formerly colonized by Eurasian watermilfoil. Hydrilla has been steadily managed on some TVA reservoirs during the past several years; however, the TVA has significantly reduced herbicide treatments. Current treatments by TVA are confined to areas in the vicinity of public facilities, and treatments along residential and commercially developed shorelines are largely funded by private entities. The recent introduction and establishment of the monoecious biotype of hydrilla on Lake Guntersville (and other TVA reservoirs) is of concern due to the potential for this biotype to better withstand high flow and disturbance events in the fall and spring.

Hydrilla has often displayed aggressive growth followed by rapid and extensive expansion (Fig. 1). The ability of hydrilla to grow under low light conditions, exploit use of available carbon through C-4 like photosynthesis, and create environmental extremes in the surface canopy (high temperatures, low dissolved oxygen, low light conditions, and large fluctuations in pH) has been well-documented (Haller and Sutton 1975, Van et al. 1976, Van et al. 1978, Bowes et al. 1979, McFarland and Barko 1987, Langeland 1996, Spencer and Ksander 2001, and Rybicki and Carter 2002). Hydrilla also has several other physiological and life-cycle attributes (e.g., rapid internode elongation, tuber formation, and tuber dormancy) that allow it to grow and persist in areas of reservoirs that have not historically supported submersed plant populations (Langeland 1996, Netherland 1997, Hoyer et al. 2008). It is hypothesized that as hydrilla grows and expands, it tends to clarify the water column by filtering of particulates, uptake of nutrients via the plant and associated epiphytic algae, and reduced wave action and sediment stabilization. The increased light penetration essentially provides a feedback loop that allows hydrilla to further expand in deeper water.

Figure 1. Hydrilla has numerous physiological attributes that allow it to rapidly expand and form dense surface canopies on reservoirs throughout the southeastern United States.

A Second Invasion by Hydrilla – Biotype Differences

It is significant to note that the Tennessee and Kentucky region is likely seeing a second invasion by a monoecious biotype of hydrilla. The original invader was a dioecious biotype (the female plant only) that is highly invasive in the southern states. Much of the literature describing hydrilla physiology and biology is based on the dioecious biotype. A second introduction of a monoecious biotype (plants with male and female flowers) of hydrilla into the U.S. has resulted in the northern spread of hydrilla (Madeira et al. 2000) and a fairly distinct zone of distribution between the biotypes (Table 1). The two biotypes are starting to overlap in areas such as northern Alabama, north Georgia, South Carolina, and Tennessee.

Table 1. Spread of hydrilla through the decades (1960 to 2010). The monoecious biotype of hydrilla is denoted in italics. *Indicates states with both biotypes.

Decade Hydrilla Introduced	**Total Number of States**	**States**
1960-1969	1	FL (1959)
1970-1979	6	AL, CA, *DE*, GA, LA
1980-1989	13	*CT*, *MD,* MS, *NC,* SC, TX, *VA,CA*
1990-1999	17	AR, *PA,* TN, *WA*
2000-2009	28	*AL*, GA**, ID, *IN*, KY, *KY*, MA, ME, NJ, NY, OH,* OK, *SC*, TN*, WI, WV*

There has been speculation that the monoecious biotype of hydrilla is outcompeting the dioecious form in areas where they co-exist in Lake Guntersville. The biology of the two hydrilla biotypes is quite different, and it has been hypothesized that the monoecious biotype may be better suited to thriving in the conditions characteristic of high flow reservoirs in the Tennessee and Kentucky region (Netherland 2005). Monoecious hydrilla typically dies back completely in the fall and relies on synchronous sprouting of tubers and turions to recolonize an area. In contrast, dioecious hydrilla likely functions more like Eurasian watermilfoil in relying heavily on reduced but persistent rootcrowns for recovery in the spring. While drawdowns are effective for Eurasian watermilfoil in the areas exposed to drying (Madsen 2009) they are ineffective in controlling either biotype of hydrilla as reflooding will stimulate high rates of tuber sprouting (Netherland 1997). Ongoing research suggests that sprouting of monoecious hydrilla tubers is much more synchronous than for dioecious biotypes, and the timing of sprouting in late May and early June could coincide with reduced flows and turbidity following spring rains (Rob Richardson, North Carolina State University, personal communication). This could result in conditions that favor the establishment and spread of monoecious hydrilla over dioecious hydrilla and Eurasian watermilfoil. Research on the different phenology of monoecious and dioecious hydrilla tubers and the role of synchronous sprouting is ongoing. The competitive interactions between monoecious and dioecious hydrilla in AL, AR, TN, KY, and NC deserves research attention as resource managers in this region likely are not making a distinction between the two biotypes.

Hydrilla and Ecological Services

As hydrilla has shown the ability to successfully establish in southern reservoirs, there has been increased discussion on how and whether to utilize this invasive plant for various ecological services (primarily as submersed habitat in areas where native SAV can not readily establish). The debate over whether there is a “right amount of an invasive plant”, and if so, how one maintains this desired level is particularly relevant to hydrilla. The positive relationship between hydrilla and bass fishing has long been touted by anglers and has also received research attention (Maceina and Reeves 1996, Slipke et al. 1998, Sammons et al. 2005). More recent research has focused more on hydrilla and its potential for overall ecosystem services (Rybicki 2007, Hershner and Havens 2008, Hoyer et al. 2008).

Many aquatic plant managers have been reluctant to accept the premise that hydrilla can provide an overall benefit to the aquatic ecosystem because many systems are likely to grow hydrilla to excessive levels or growth will reach nuisance levels in areas where hydrilla is not desired (e.g., marinas, docks, boat ramps, etc.). Given the ongoing expansion of monoecious hydrilla, it is especially important that resource managers understand the implications of a new biotype proliferating in large reservoirs. One of the dilemmas in making the decision to utilize an invasive plant for ecosystem services in a multi-purpose system is determining when management efforts are necessary. Hydrilla has consistently shown the ability to rapidly go from light and scattered infestations (e.g., high quality habitat) to dense growth that can impede navigation, result in significant changes in water quality, and influence fish community structure and size (Dibble et al. 1996). Paradoxically, most managers prefer to avoid or cannot fund large-scale management efforts due to the cost and high volume of herbicide required; however, stakeholders often view sustained small-scale management in an even more negative light as they are more likely to see herbicide applications (albeit small-scale) on a more frequent basis. Managers should strongly consider that control efforts will likely be required when utilizing an invasive plant for ecosystem services.

Nymphoides cristata and the Invasive Plant Dilemma

The invasive plant *Nymphoides cristata* provides an example of a little known species that shows significant potential to be highly problematic in the southern U.S. There has been very little research on basic biology and management of this nymphaeid plant. This creates a dilemma on whether to divert research funds and efforts from known problem plants (e.g., hydrilla or giant salvinia) to species that scientists suspect may cause future problems. *Nymphoides cristata* is widely available in the aquarium trade (marketed as “Snowflake”) and was first reported in a natural water body of Florida in 1996 (Burks 2002). Within Florida it was recently moved from a category II, plant of increasing numbers, to a category I, invasive exotic that is altering native plant communities by displacing native species by changing community structures or ecological functions (FLEPPC, 2009). While the plant was originally thought to be a "Florida problem", a recent infestation in Lake Marion, SC, has increased from an initial find of approximately 20 hectares in 2006 to over 1500 hectares in 2010. This rapid rate of expansion, general inability to achieve sustained control with registered herbicides and a very low feeding preference by triploid grass carp suggest this plant may be a significant future problem in numerous southeastern reservoirs, particularly those stocked with grass carp for hydrilla control. Continued sales of *N. cristata* in the ornamental aquarium trade and the establishment of disjunct populations in lakes and reservoirs throughout the Southeast suggest increased research efforts are justified. In essence, given the current distribution of the plant and widespread presence in the aquarium trade, prevention may be very difficult. Unfortunately, trying to predict which invasive aquatic species will become the next hydrilla is particularly challenging. The results from current management efforts on Lake Marion and future documentation of spread to nearby waters will enlighten resource managers regarding the future course of this plant.

Three Examples of Challenges Associated with Hydrilla

The decision on whether to manage hydrilla or other aquatic invasive plants can be quite complex and confounded by numerous issues. Below, three examples provide current situations involving hydrilla and how decisions on management and ecological services can become highly complex and in a sense, political. Aside from Eurasian watermilfoil, the Tennessee and Kentucky regions have not historically been impacted by significant invasive aquatic plant problems; however, with the current spread of monoecious hydrilla, this may change.

Hydrilla as a Substrate for a Toxic Cyanobacterium

The introduction of non-native aquatic plants can have unforeseen ecological consequences. For example, avian vacuolar myelinopathy (AVM), a neurologic disease that has caused death of herbivorous waterbirds and raptors since it was first diagnosed in 1994, has been linked to hydrilla (and other invasive plants such as egeria) in several southeastern reservoirs ranging from Texas to North Carolina (Wilde et al. 2005). In this case, the leaves of hydrilla support an epiphytic stigonematalan cyanobacterium that produces a novel neurotoxin that is highly toxic to birds that feed on the vegetation. American coots and other herbivorous waterfowl, concentrate their feeding on dense vegetation, and their populations generally increase on reservoirs with abundant aquatic plants. As these birds are impaired by AVM, birds of prey (especially eagles) readily consume the sick and dead birds (Wilde et al. 2005) and subsequently contract AVM.

Strom Thurmond Reservoir, a US Army Corps of Engineers Reservoir in South Carolina, provides a good example of the dilemma that resource managers face in managing hydrilla. A recent survey indicated that the 28,700 ha reservoir contained over 3,200 ha of hydrilla (Fig. 2). Hydrilla density has steadily increased along shoreline areas of this deep reservoir providing a habitat that is intensely fished by bass anglers. As hydrilla coverage and density increases there is more substrate for the toxic cyanobacteria and increased contraction of AVM by water birds and birds of prey is likely. Research on this issue is ongoing, but regardless of the science, different stakeholder groups (e.g., anglers, waterfowl hunters, bird watchers, aquatic resource managers) are likely to have very different opinions regarding the best course of action and strategy for controlling hydrilla. Moreover, it is also likely that issues with AVM will increase as hydrilla continues to spread to new sites throughout the southeast.

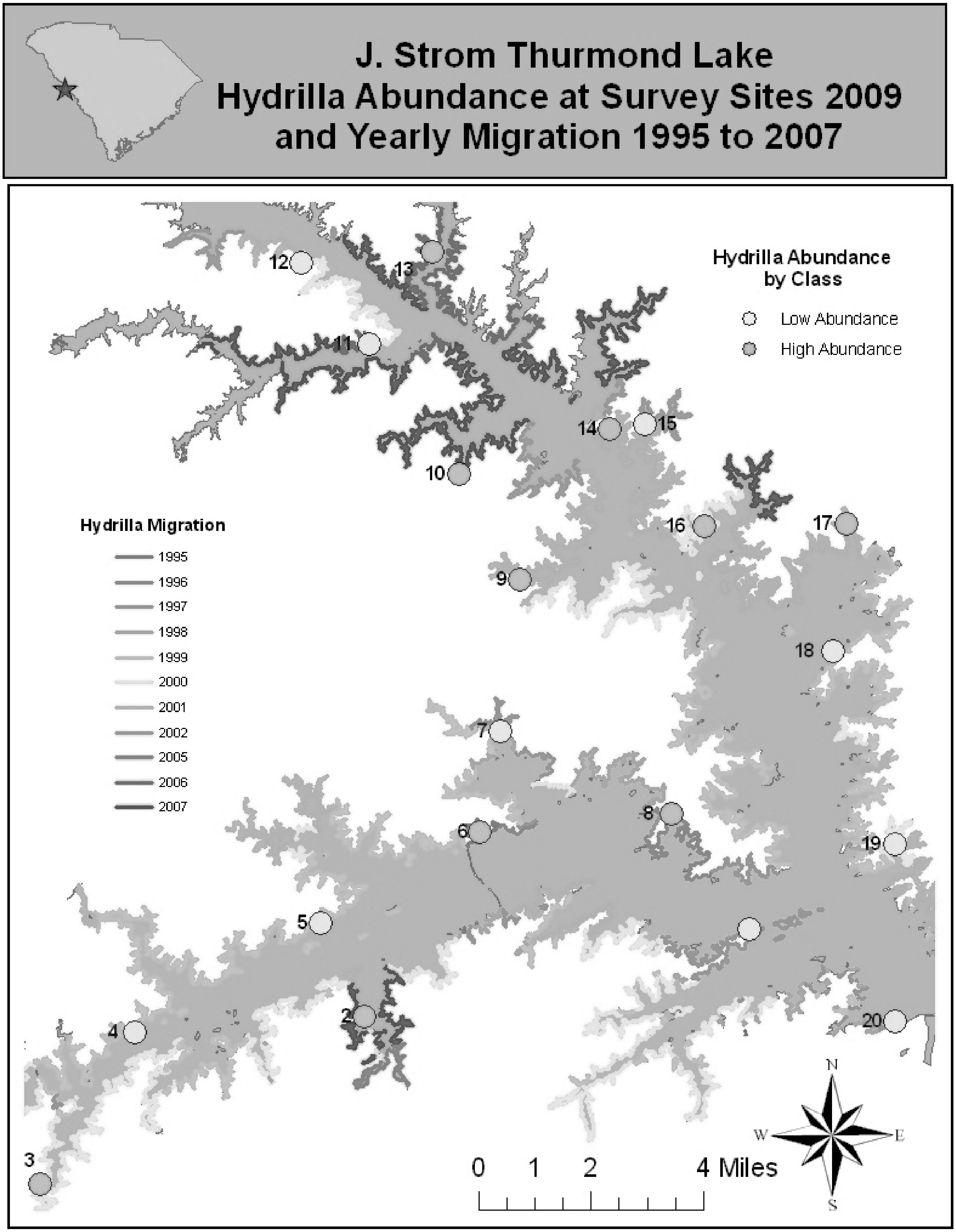

Figure 2. Hydrilla coverage and and change in distribution over a 12-year period on Strom Thurmond Reservoir

Hydrilla Utilized to Increase Numbers of an Invasive Snail to Support an Endangered Raptor

There is an ongoing debate in Central Florida regarding the promotion of hydrilla abundance on the 9,500 ha Lake Tohopekaliga to increase the population of an invasive snail (*Pomaceae insularum*) to support the federally endangered snail kite (*Rostrhamus sociabilis plumbeus*). The snail kite is a wetland-dependent raptor that typically displays an extreme form of dietary specialization, feeding almost exclusively on a single species of native apple snail (*Pomacea paludosa*) (Cattau et al. 2010). The invasive snail has become much more prevalent on Lake Tohopekaliga and has been linked to increased usage of the lake by high numbers of adult snail kites. The invasive snail is significantly larger than the native snail, and while adult snail kites can handle these larger snails, the juveniles have significant problems in successfully handling the larger invasive snails. The reduced ability of juvenile snail kites to utilize invasive adult snails has been suggested as leading to an ecological trap that attracts breeding adult snail kites while simultaneously depressing juvenile survival (Cattau et al. 2010).

As the adult snail kites have concentrated on Lake Tohopekaliga, the debate regarding the status of hydrilla control efforts has become a central focus. While the science regarding linkages between increased hydrilla presence and increased foraging success by juvenile snail kites (presumably utilizing smaller individual invasive snails) remains somewhat anecdotal, the current management strategy on Lake Tohopekaliga has been altered to promote more hydrilla in an effort to increase snail numbers and improve capture of invasive snails by both adult and juvenile snail kites. Lake Tohopekaliga can support abundant hydrilla with thousands of contiguous hectares covered by surface mats. The potential consequence of promoting high numbers of this invasive snail, a generalist herbivore, (Gettys et al. 2008), has received limited attention during this debate. As noted earlier, there can be unforeseen consequences when utilizing invasive species for ecological services.

A New Wave of Hydrilla Invasion?

While the debate over the potential for hydrilla to provide beneficial ecological services in southeastern and mid-Atlantic reservoirs will likely continue, the movement of hydrilla into the thousands of kettle lakes in the upper Midwest and Northeast could represent a major disturbance to systems that already support diverse and abundant native submersed species. While it remains unknown whether hydrilla will aggressively displace the native vegetation, as well as compete with other established invasive species (Eurasian watermilfoil, curlyleaf pondweed), states such as Indiana, Wisconsin, Maine, and Massachusetts currently have hydrilla eradication programs. In essence, this region of the country is currently working under a precautionary principle when it comes to nascent hydrilla infestations. Infestations in large systems such as the Ohio River present a real threat as a gateway for hydrilla movement into the natural lakes of the upper Midwest and Northeast. As hydrilla continues to proliferate in reservoirs and rivers, the ability to keep it out of the natural glacial lakes will become more difficult. Policies will vary from state to state, and the ability to maintain an eradication program in a state next to a state that has no management policy will create future challenges for aquatic managers.

Management Tools

There are numerous management tools available for control of invasive aquatic plants. Technical aspects and the pros and cons associated with various management techniques such as harvesting, triploid grass carp, biological control, cultural controls, and herbicides are well described and discussed (Gettys et al. 2009). Grass carp are widely stocked in southeastern reservoirs for low-cost hydrilla control, and yet the long-term and sometimes near total vegetation control provided by grass carp is often considered highly undesirable by many stakeholder groups (Netherland 2008). Aquatic herbicides are a widely used tool for hydrilla control; however, there is also strong opposition to this approach by various stakeholder groups (Netherland 2008, Netherland 2009). Given the current pace of research and the low amount of research funding dedicated to invasive aquatic plants, the likelihood that novel approaches to hydrilla control will be available in the near term is low. Stakeholders are increasingly requesting resource managers to allow invasive plant growth for ecosystem benefits, without allowing these plants to grow to nuisance levels that impede various uses. This obviously creates a difficult challenge for resource managers.

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USGS NONINDIGENOUS AQUATIC SPECIES DATABASE WITH A FOCUS ON THE INTRODUCED FISHES OF THE LOWER TENNESSEE AND CUMBERLAND DRAINAGES

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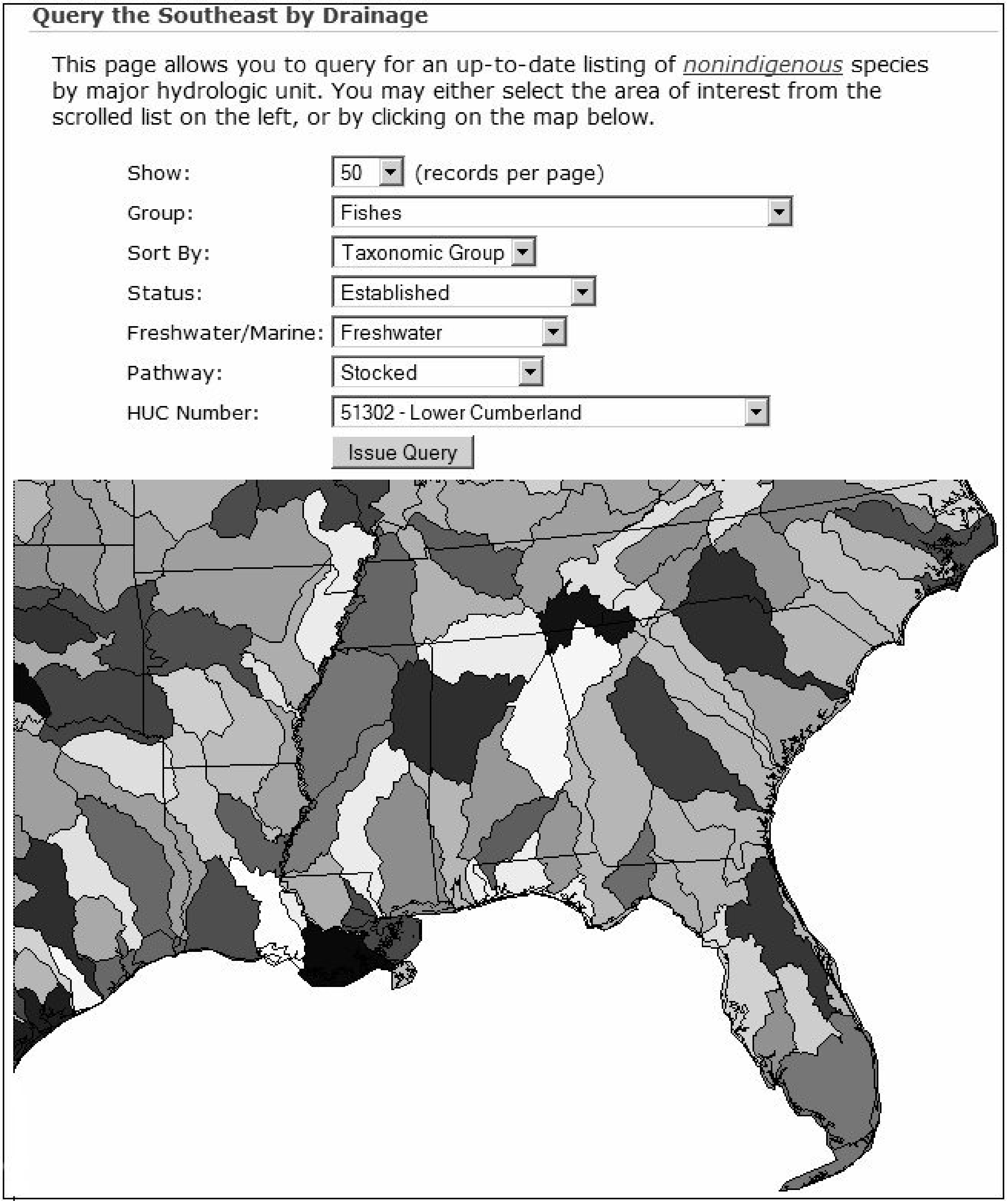
ABSTRACT. The Nonindigenous Aquatic Species (NAS) database (http://nas.er.usgs.gov) functions as a national repository and clearinghouse for occurrence data for introduced species within the United States. Included is locality information on over 1,100 species of vertebrates, invertebrates, and vascular plants introduced as early as 1850. Taxa include foreign (exotic) species and species native to North America that have been transported outside of their natural range. Locality data are obtained from published and unpublished literature, state, federal and local monitoring programs, museum accessions, on-line databases, websites, professional communications and on-line reporting forms. The NAS web site provides immediate access to new occurrence records through a real-time interface with the NAS database. Visitors to the web site are presented with a set of pre-defined queries that generate lists of species according to state or hydrologic basin of interest. Fact sheets, distribution maps, and information on new occurrences are updated as new records and information become available. The NAS database allows resource managers to learn of new introductions reported in their region or nearby regions, improving response time. Conversely, managers are encouraged to report their observations of new occurrences to the NAS database so information can be disseminated to other managers, researchers, and the public. In May 2004, the NAS database incorporated an Alert System to notify registered users of new introductions as part of a national early detection/rapid response system. Users can register to receive alerts based on geographic or taxonomic criteria. The NAS database was used to identify 23 fish species introduced into the lower Tennessee and Cumberland drainages. Most of these are sport fish stocked to support fisheries, but the list also includes accidental and illegal introductions such as Asian Carps, clupeids, various species popular in the aquarium trade, and Atlantic Needlefish (*Strongylura marina*) that was introduced via the newly-constructed Tennessee-Tombigbee Canal.

INTRODUCTION

NAS database

The United States Geological Survey's Nonindigenous Aquatic Species (NAS) Database tracks the distribution of species introduced outside their historically native range. This includes species introduced from other countries, as well as species moved within the United States from one state to another, and from one drainage to another (even within the same state). The NAS database focuses on freshwater species and partners with the Smithsonian Environmental Research Center, which tracks estuarine and marine species in their complementary database, the National Estuarine and Marine Invasive Species Information System (NEMISIS).

Web-based queries. The NAS on-line database [http://nas.er.usgs.gov] provides a number of standardized queries to users. Users can query species by state; by 2-, 6-, or 8-digit hydrologic unit codes (HUCs) (Fig. 1); by county; and by a set of text fields (Fig. 2) that includes taxonomic group, state, species name, status, pathway, habitat (freshwater vs. marine), and origin (exotic or native transplant). All queries result in a list of species that match the requested criteria and contains links to more information about each species including collections, a species fact sheet, a HUC-based map, and a point-based distribution map. These maps are a core part of the NAS program and will be explained in more detail below.

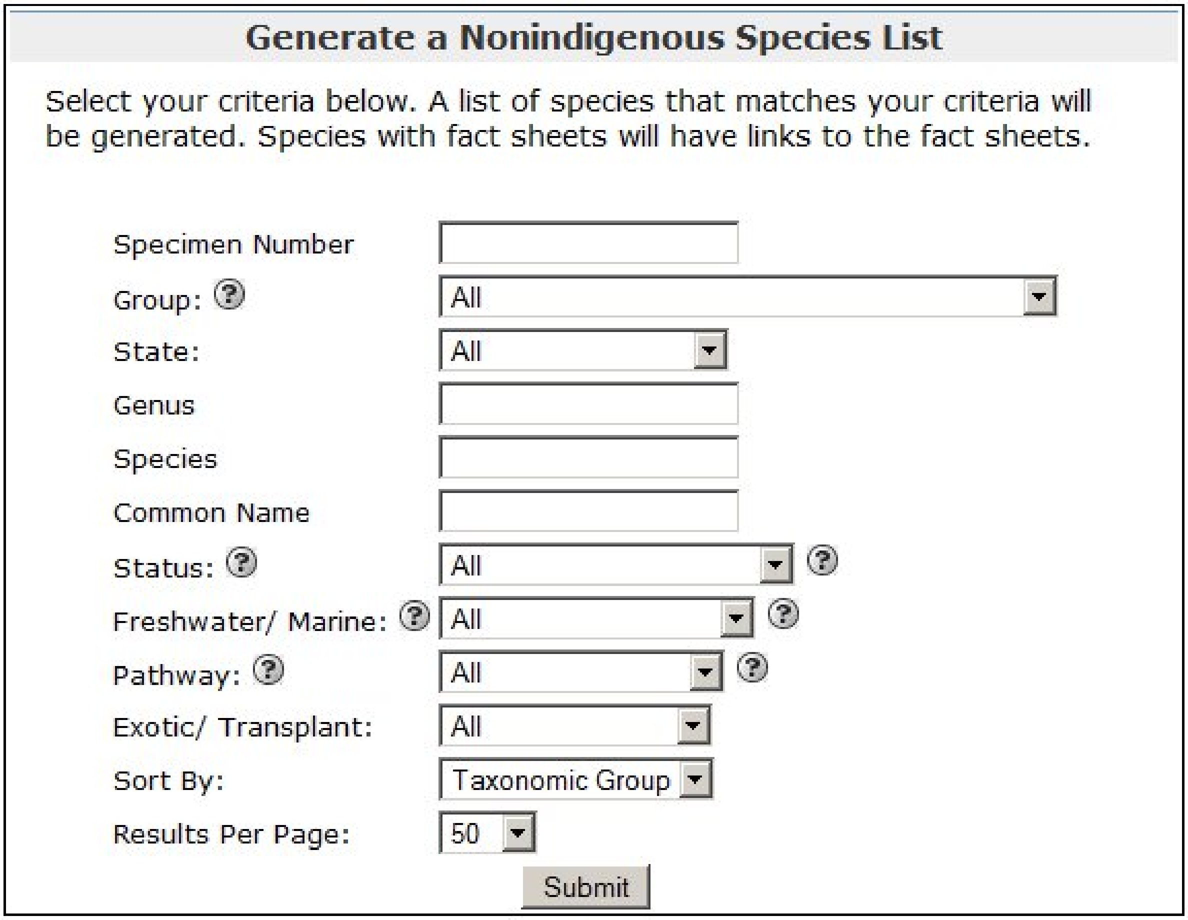


Query the Southeast by Drainage

This page allows you to query for an up-to-date listing of *nonindigenous* species by major hydrologic unit. You may either select the area of interest from the scrolled list on the left, or by clicking on the map below.

Show: 50 (records per page)
Group: Fishes
Sort By: Taxonomic Group
Status: Established
Freshwater/Marine: Freshwater
Pathway: Stocked
HUC Number: 51302 - Lower Cumberland

Figure 1. Example of a web-based query. This is a query by 6-digit hydrologic unit in the Southeast. User can select a taxonomic group, status, freshwater or marine habitat, vector, and HUC. The result will be a species list that matches the criteria requested.



Generate a Nonindigenous Species List

Select your criteria below. A list of species that matches your criteria will be generated. Species with fact sheets will have links to the fact sheets.

Specimen Number
Group: All
State: All
Genus
Species
Common Name
Status: All
Freshwater/ Marine: All
Pathway: All
Exotic/ Transplant: All
Sort By: Taxonomic Group
Results Per Page: 50

Figure 2. Example of a web-based query. This is an advanced text query. The user can select a taxonomic group, state, species, population status, habitat, pathway, exotic or native transplant. The result will be a species list that matches the criteria requested.

Source Analysis. In order for a management agency to address the influx of invasive species, it is important to know the source areas of introduction. The NAS database can be used to look at the proportion of introduced species that are coming from other countries via importation and the proportion that are native to other areas of the United States but have been moved outside of their native range. An analysis of the entire USA reveals that 61% of the fish species introduced into non-native areas are native to other regions of the nation (Figure 3). The remaining 39% were imported from foreign countries. However, the pattern is much different for the state of Florida. Only 18% of the fish species introduced into that state are native to the U.S. while 82% are from foreign countries. The high percentage of exotic species is due to Florida's role in raising aquarium species in aquaculture facilities and because of the warm climate that allows many releases to survive long enough to establish or be recorded. Comparing the state of Tennessee and the lower Tennessee and Cumberland River drainages to the Unites States reveals more similarity (Figure 4). Another way the source analysis can be used is to look specifically at source locations of foreign species. Figure 5 shows a comparison of origins of introduced foreign fishes and reptiles. The majority of reptiles come from either Asia or South America, whereas fishes are more diverse in origin.

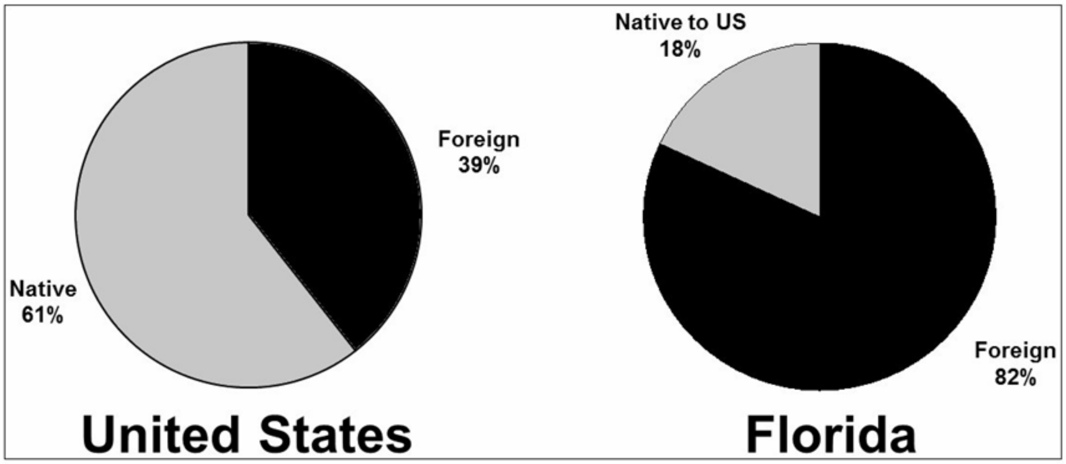


Figure 3. Origins of fishes introduced into the United States vs. Florida. “Native to U.S” depicts species that are native to some part of the United States but introduced outside their native range. For example, these may be species native to the east coast that are introduced to the west coast. “Foreign” are species that are from foreign countries.

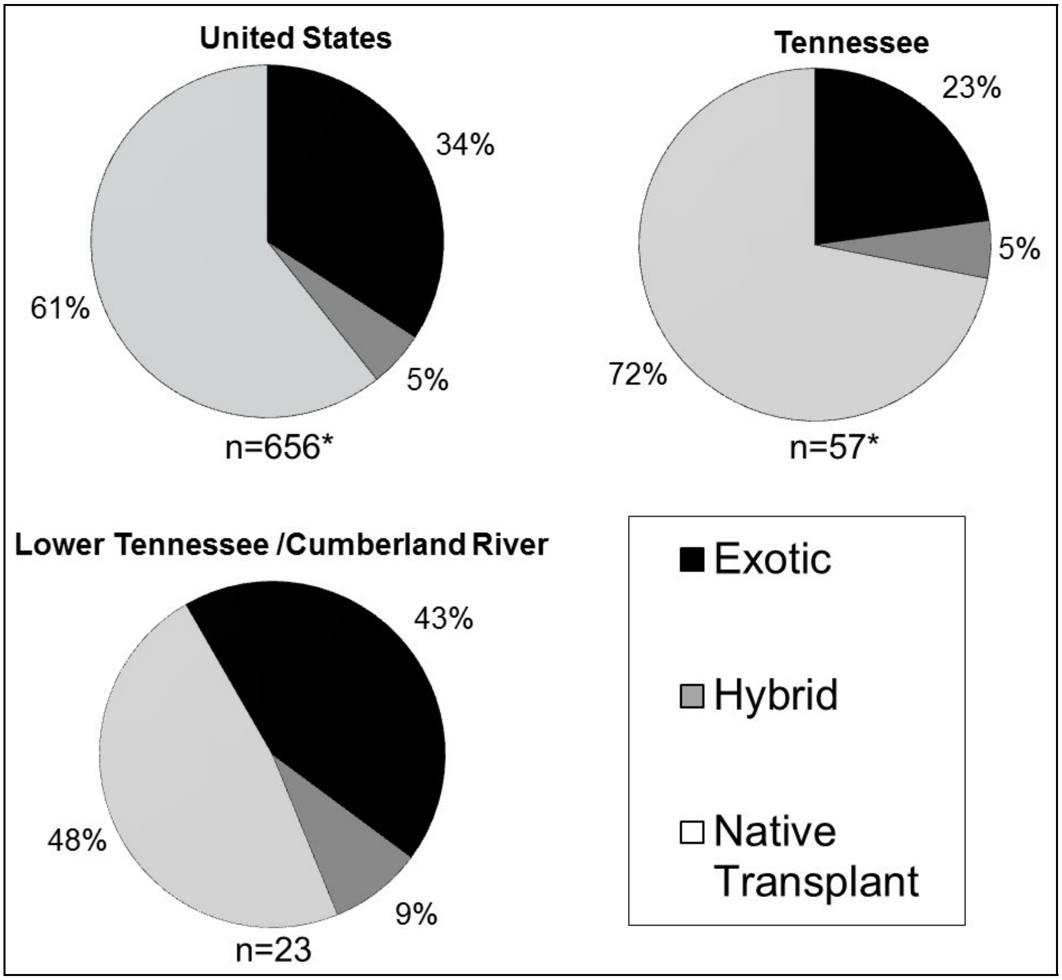


Figure 4. Origins of introduced fish species (*and subspecies) in three different areas.

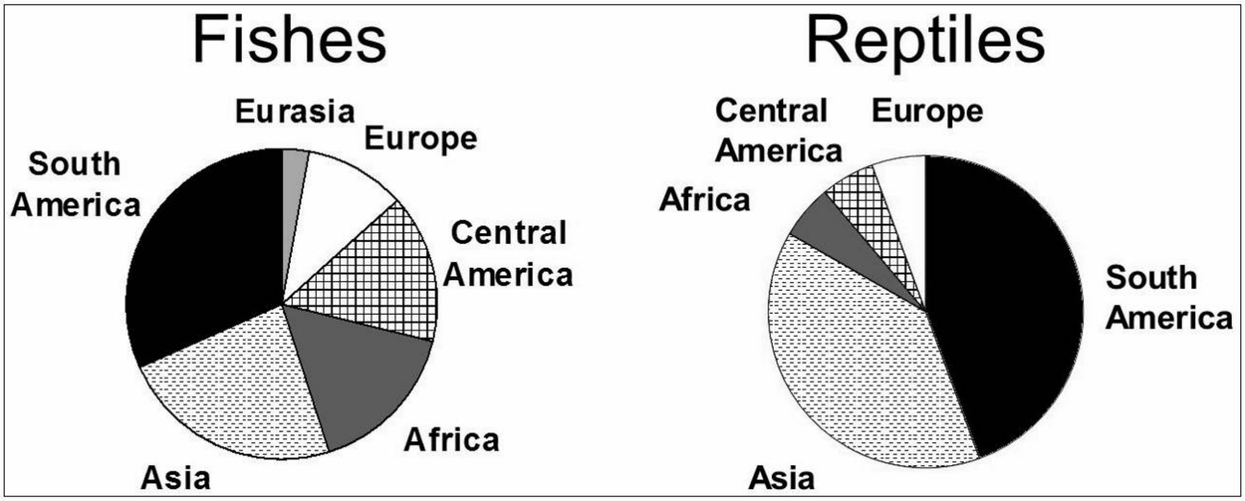


Figure 5. Origins of foreign species. This example shows how a user can look at specific taxonomic groups to define the origin of the imports.

Spatial Analysis. Because the data in the database are geo-referenced, a user can look at spatial patterns. Figure 6 shows the invasion "hot spots" in the country for introduced fishes. South Florida, Lake Mead on the Colorado River, and the Hawaiian Islands are the areas with the most fish introductions. The Atlantic slope and the West are both fairly high, whereas the Mississippi, Missouri and Ohio basins are all fairly low. Combined with other data (e.g., environmental or population data, distributions of imperiled species), analyses of spatial trends of introductions can assist natural resource managers in targeting areas for further research or mitigation programs.

Vector Analysis. The previous two sections demonstrate how the NAS database can summarize where species are introduced from and to where they are introduced. The remaining information needed for management is to know how they are introduced. The NAS database can be used to address this as well. Different areas (or taxonomic groups) can be compared to one another (Figures 7 and 8), and examination of vectors can help management agencies to better design and implement effective strategies and programs (e.g., permitting regulations, education) for reducing introductions.

Patterns Through Time. Because each of the collection records in the NAS database includes a date component, it can be used to look at how patterns change over time. The simplest way to do this is to look at the number of species introduced over time (Figure 9). All three geographic scales (nation, state, and drainage) show an increase in the number of introductions over time, with the greatest jump occurring after 1950. Another way to look at the temporal component is to combine it with another variable such as origin (Figure 10) or pathway (Figure 11), to visualize changes in the importance of different introduction sources or vectors over time.

Mapping. The NAS program produces two types of maps. The first is a map of species distribution by hydrologic units (also called hydrologic unit codes or HUCs) (Figure 12). In this type of map the entire HUC is shaded if the species has ever been reported from a unit. The shading color designates whether the species of interest is native or introduced. Because data are reported at various HUC levels, HUCs larger than 8-digit units may be mapped, but are designated by a lighter color of red.

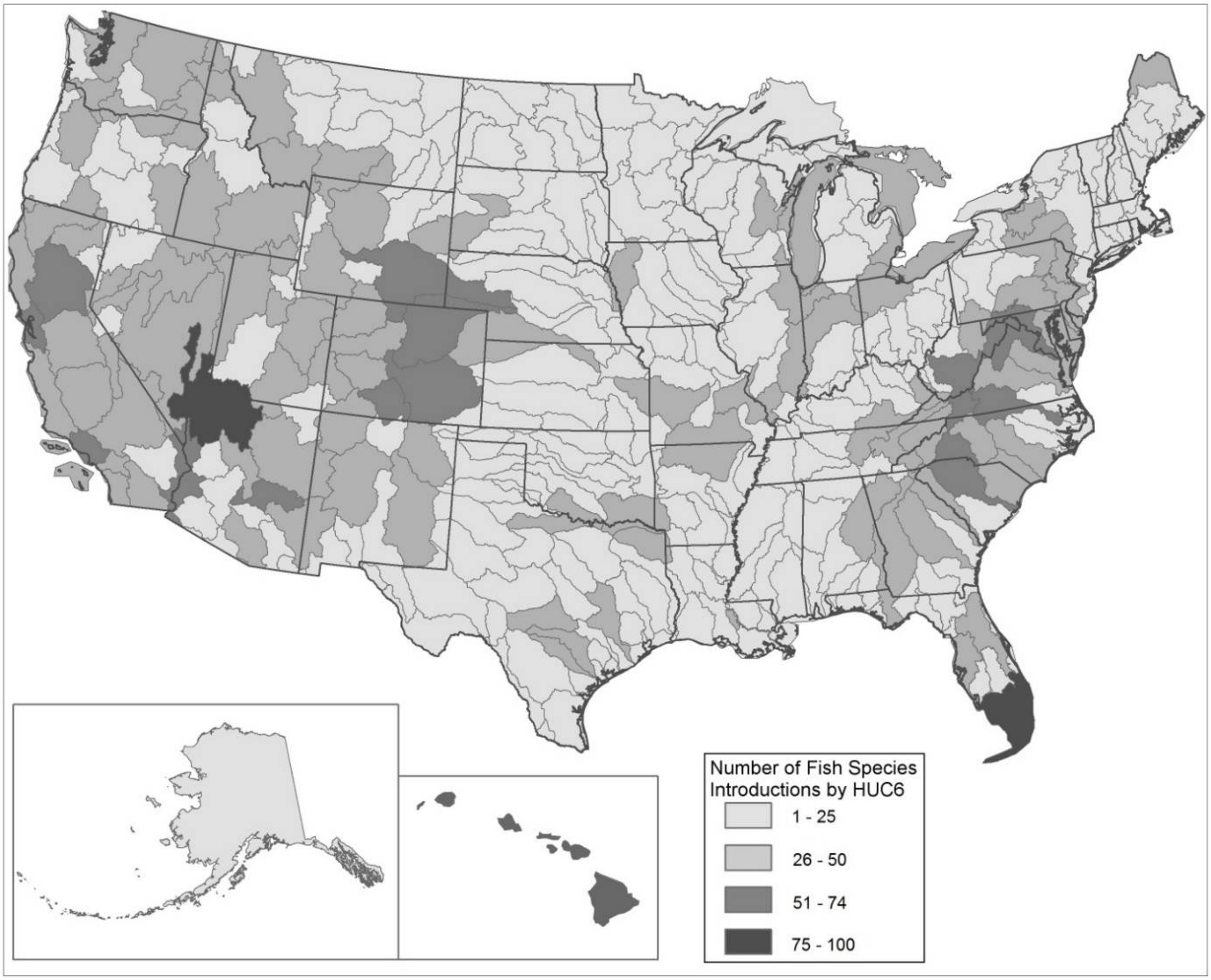


Figure 6. Number of fish species introduced in each HUC-6. This includes species that did not persist.

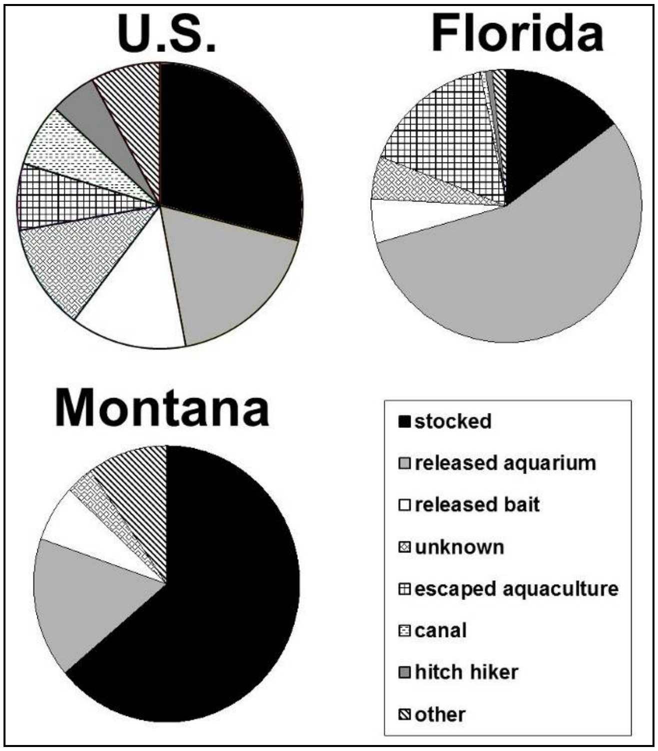


Figure 7. Proportion of fish species introduced by means of several different vectors.

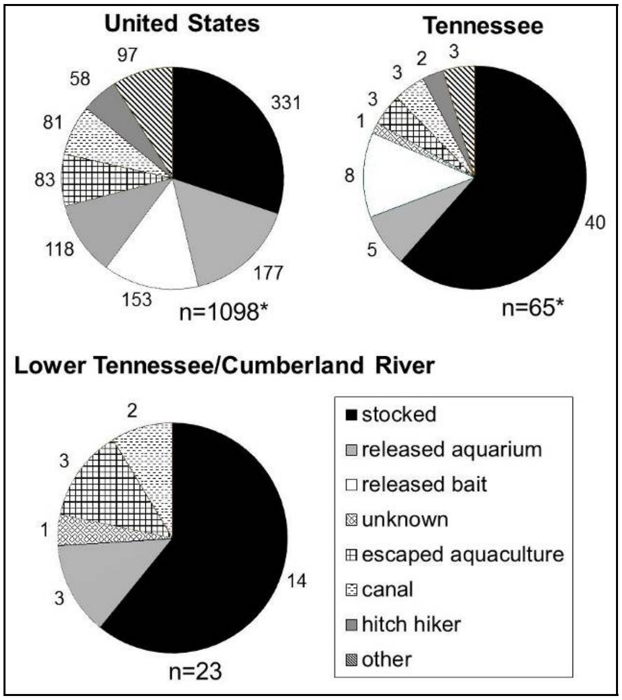


Figure 8. A comparison of fish introduction vectors in three different areas. Number of species (*and subspecies) introduced via several vectors. A single species may be introduced by more than one vector and, therefore, may be counted more than once.

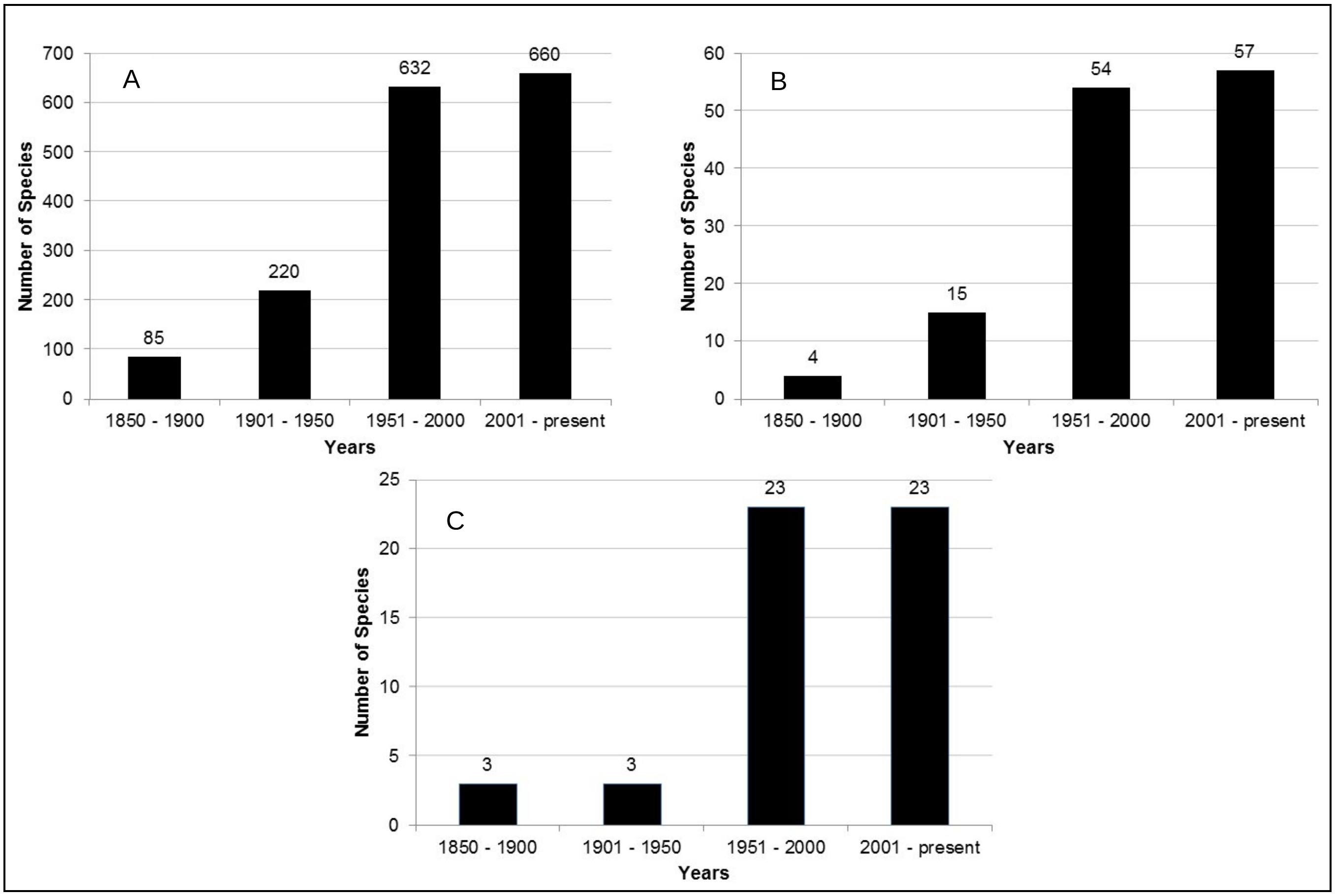


Figure 9. Number of freshwater fish species introductions in three different areas over time. A = United States; B=Tennessee; C=Lower Tennessee/Cumberland rivers

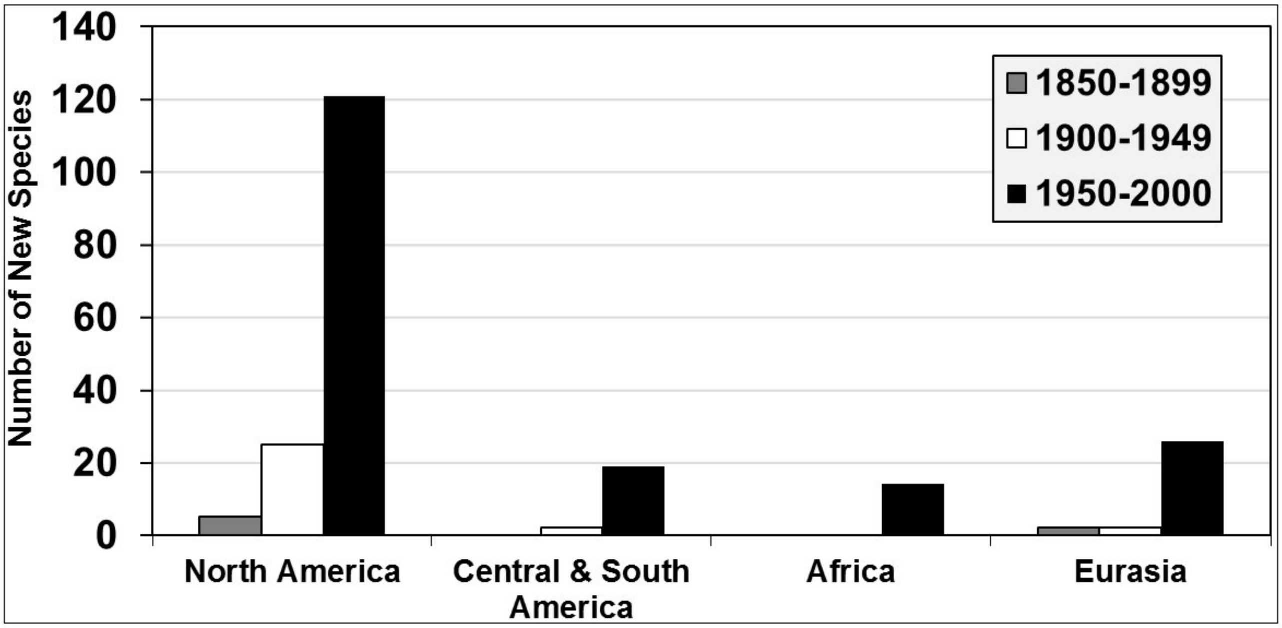


Figure 10. Source regions of introduced fishes in the Southeast over time.

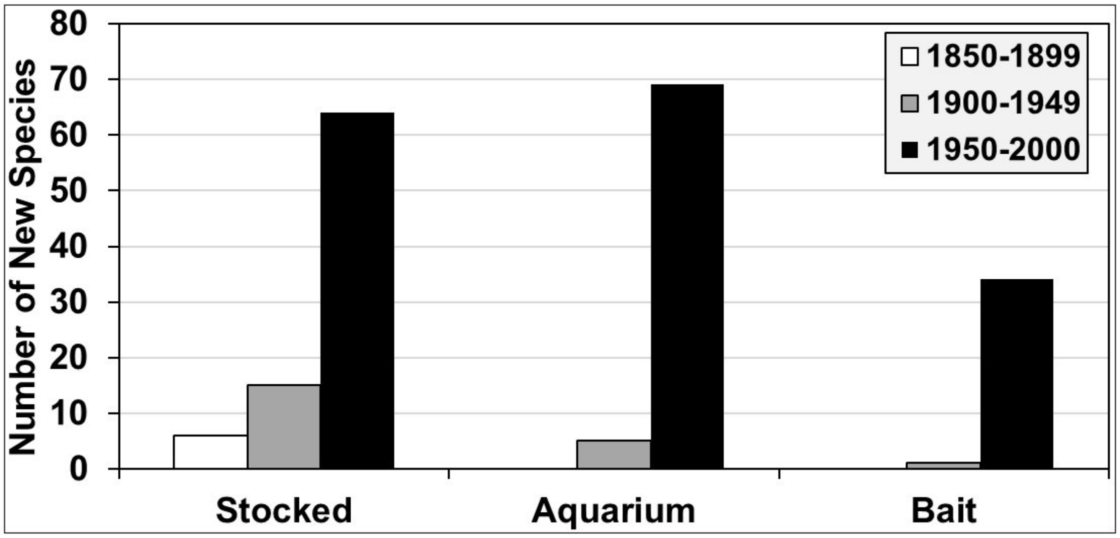


Figure 11. Pathways of fish introductions in the Southeast over time.

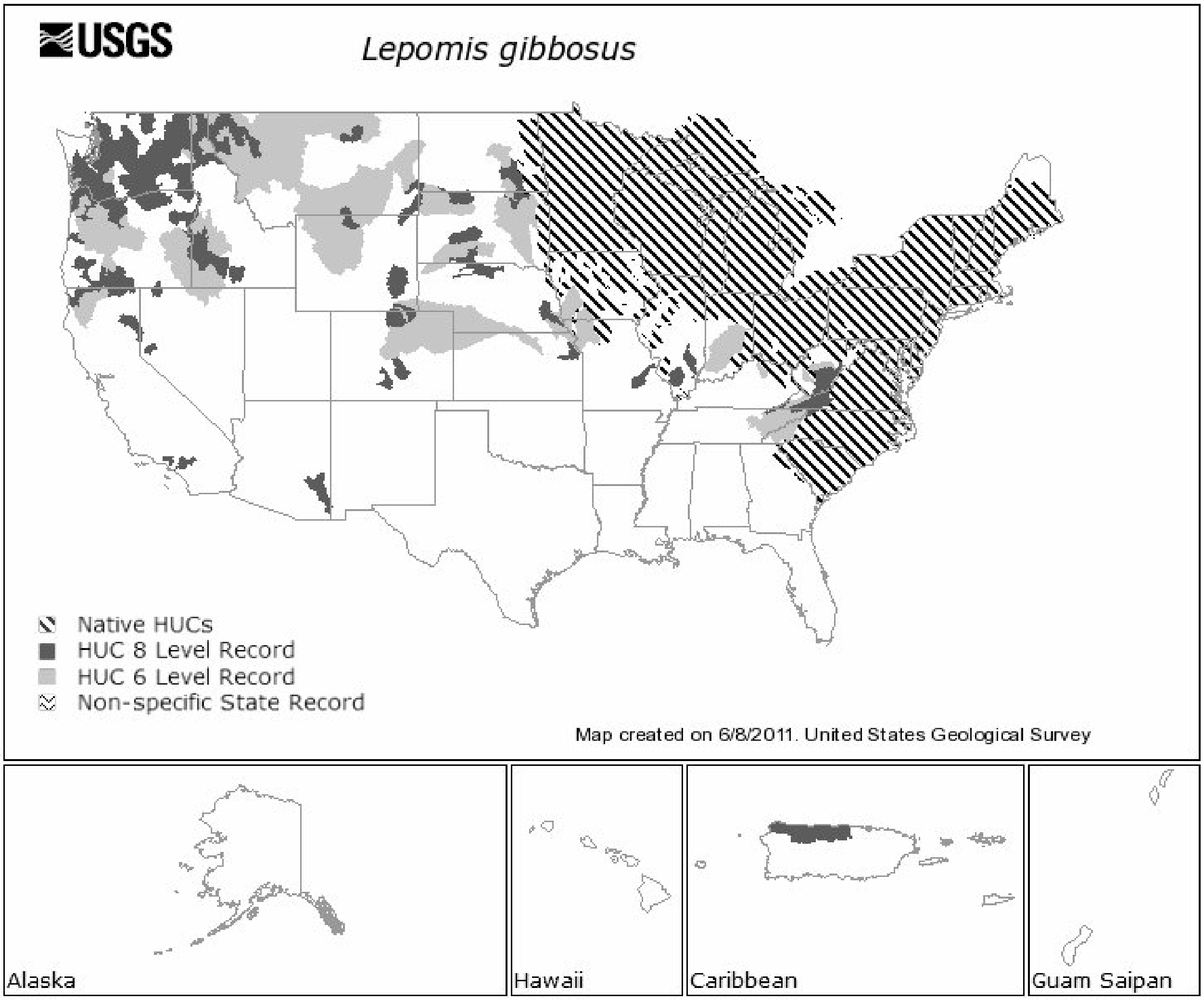


Figure 12. HUC based map of distribution of *Lepomis gibbosus*.

The second type of map, which has been developed only recently, is point distribution maps. These maps use Open Layers$^{\text{TM}}$ with Bing$^{\text{TM}}$ backgrounds and show point locations with latitudinal and longitudinal coordinates (Figure 13). The points are color-coded to show species status at that location (to the best of our knowledge) and are shape-coded to indicate the spatial accuracy of that point as accurate, approximate, or centroid of an area. The maps have zoom and pan capabilities. The background can include roads, satellite/aerial photograph, a hybrid of the two, or a solid tan background. Additional layers can be added such as HUC 6 or HUC 8 boundaries and state boundaries. Additional layers, such as temperature and precipitation data, will be added soon. Each point location can be identified by pointer selection. A balloon summarizing the collection will appear, with a link to the full collection record including the source of the record.

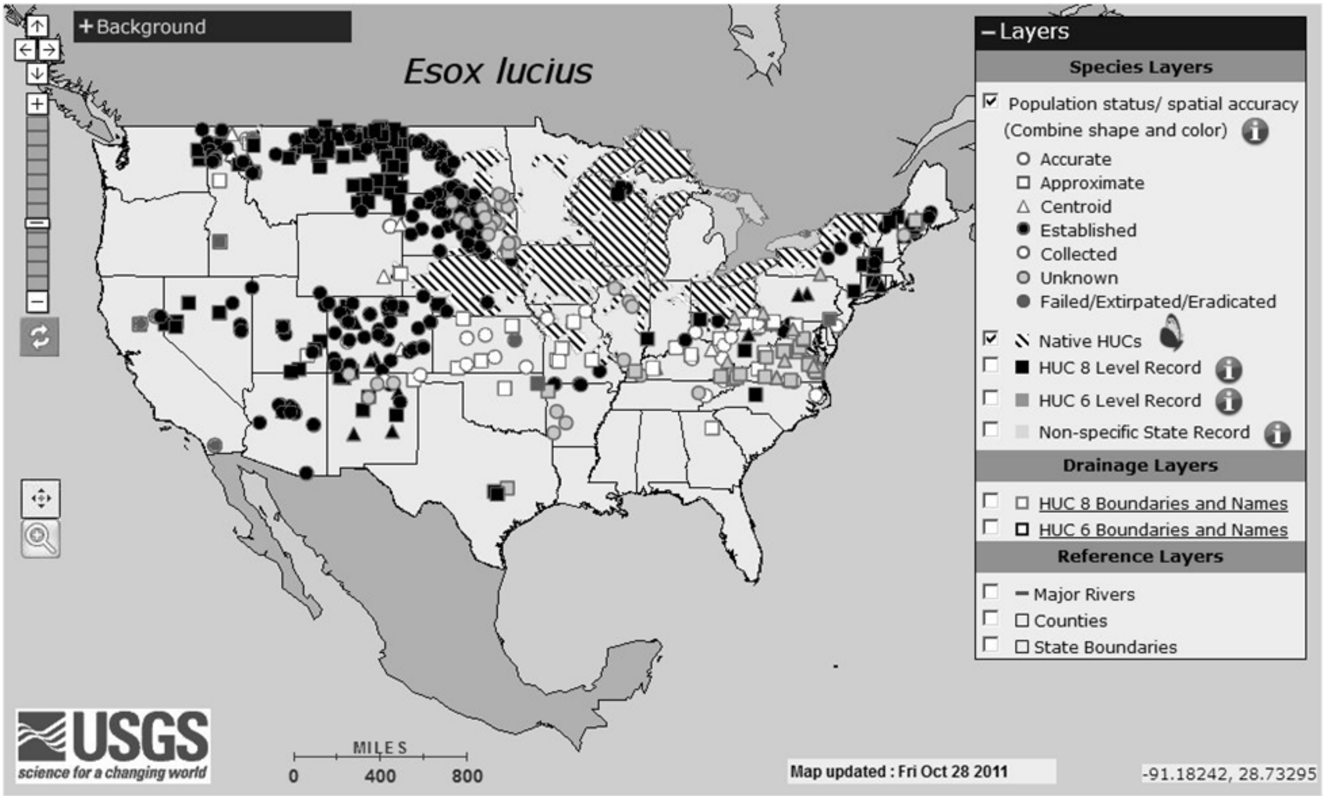


Figure 13. Point-based map of distribution of northern pike (*Esox lucius*).

NAS Alert System. An integral part of the NAS database is the NAS Alert System, launched in May 2004. The alert system acts as an early detection tool to alert natural resource managers of new invasions. The managers can then decide if actions such as quarantine, eradication attempts, or new monitoring efforts are warranted. The system is designed to notify registered users of new occurrences based on their desired categories of interest. The system is flexible, providing two different perspectives – one to a user interested in an area, the other to users interested in a species. Users can sign up for "State Watches", "Group Watches" and/or "Species Watches". Registered users receive an e-mail alert indicating the species and area to which it is new (country, state, county, drainage). The e-mail lists the location and includes links to the full specimen record in the database and to the fact sheet for that species. The location of each occurrence is shown on a Google™ map displayed by the specimen record for each alert.

When reports are received by the USGS, they are verified and entered into the NAS database. Each report is geo-referenced at various levels – country, state, county, HUC and latitude/longitude. When the record is submitted to the system it is compared to other records already in the system to determine if that species has previously been reported from the 1) country, 2) state, 3) county, and 4) HUC (Figure 14). If the species has been found in all four aforementioned geographic areas, no alert is generated. If it is new to any one of those regions, the system will generate an alert to tell the data entry person to which area it is new (country, state, county, HUC). If the record is more than two years old, an internal flag that acts as a reminder to update the species fact sheet is shown. If the report is from the current year or the year before, it goes into a holding area for review. The alerts in the holding area are reviewed to determine if they are “alert worthy”. Generally, alerts are issued for collections within the past six months. If the record is not determined to be alert-worthy, because of its age, the alert is not distributed. The alert is then deleted from the holding area. All county-level alerts are examined to determine if an alert truly represents a new area for a given species. For example, a record may indicate the species is “throughout the ‘XXX’ drainage” but had not been previously reported specifically from a particular county. In such cases a “false alert” is generated. False alerts often occur in large water bodies such as the Great Lakes or Mississippi River. The only time county-level alerts are determined to be alert-worthy is when they represent range expansion into a new area of that drainage. For example, county alerts are valid for Asian carp migrating farther upstream in the Mississippi River, or for a Great Lakes species moving eastward in Lake Ontario. Once an alert has met all the criteria, it is sent to those who signed up for that taxonomic group, species, or state.

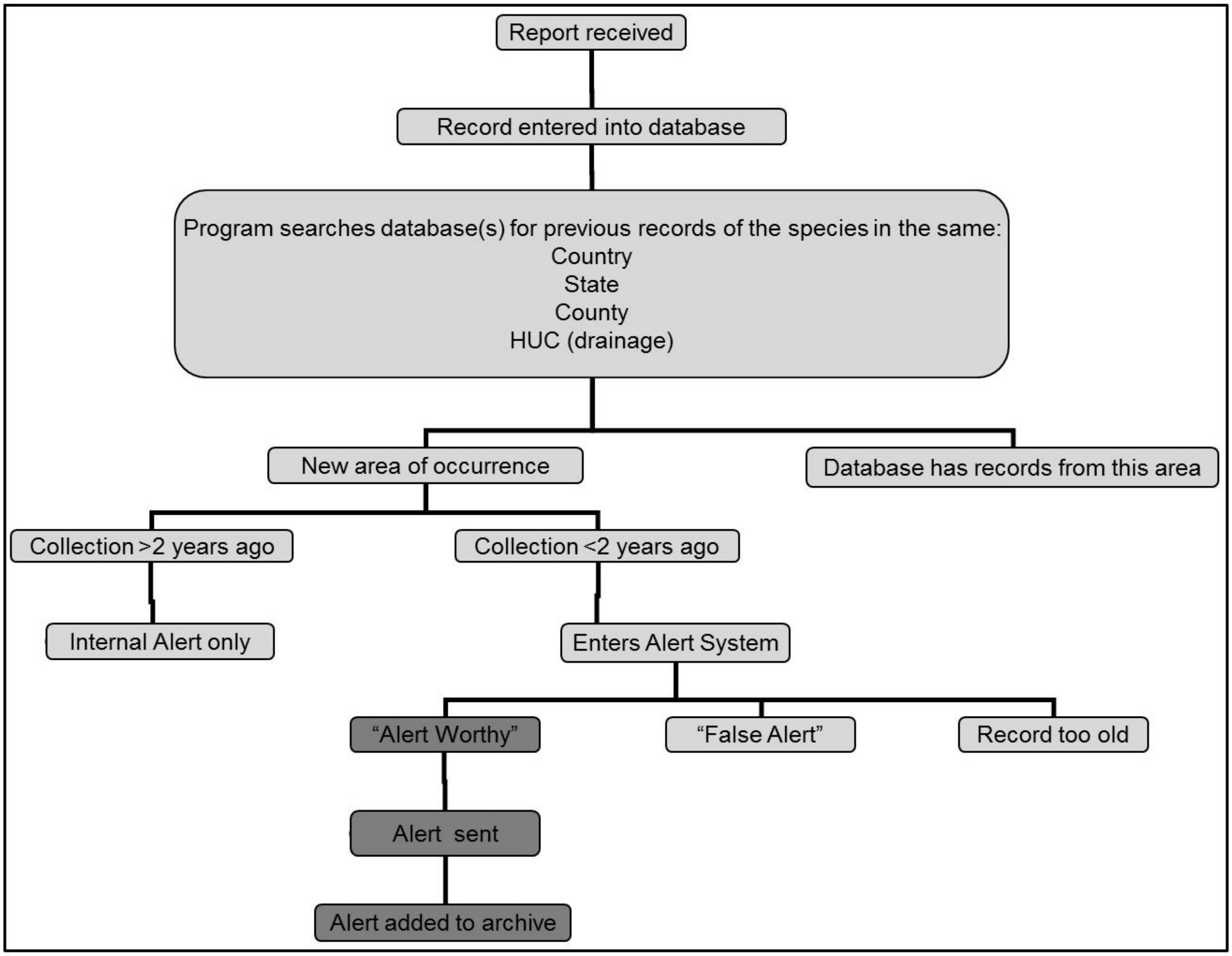


Figure 14. Flow chart of the NAS Alert System.

When a record does not automatically generate an alert because it is not the first time it has been recorded from an area, a "bonus alert" can be generated. When a record is designated as a bonus alert, a comment is included to explain the significance of the report. This option is used when a species has not been recorded for an area for many years, or when the 2nd and 3rd individuals are found for high profile invaders, or the first gravid female or juvenile are found – indicating reproduction. All alerts are archived on the public website (http://nas.er.usgs.gov/AlertSystem/default.aspx) and are searchable. The alerts are also available via an RSS feed, making them available for use on other agency websites. In addition, alerts are now available through the new NAS Twitter feed, which can be found at http://www.twitter.com/USGS_NAS or by following @USGS_NAS.

FISH SPECIES INTRODUCED TO THE LOWER TENNESSEE AND CUMBERLAND DRAINAGES

A total of 22 non-native fish species or hybrids have been recorded for the lower Tennessee and Cumberland drainages (Table 1). Most of these (10) are sport fish including Striped Bass (*Morone saxatilis*), Wiper (*Morone chrysops* x *M. saxatilis*), Tench (*Tinca tinca*), Northern Pike (*Esox lucius*), Redbreast Sunfish (*Lepomis auritus*), Yellow Perch (*Perca flavescens*), Sauger (*Sander canadensis*), saugeye (*S. canadense* x *S. vitreus*), Rainbow Trout (*Oncorhynchus mykiss*), and Brown Trout (*Salmo trutta*) (Burr and Warren 1986, Etnier and Starnes 1993, Fuller et al. 1999, Schofield et al. 2005). Ten of the 22 introduced species are from other countries (Figure 4, Table 1). Of those 10, eight species are established. Several are species of carps from Asia: Common Carp (*Cyprinus carpio*), Silver Carp (*Hypophthalmichthys molitrix*), Bighead Carp (*Hypophthalmichthys nobilis*), Black Carp (*Mylopharyngodon piceus*) (established in the larger basin as a whole), and Grass Carp (*Ctenopharyngodon idella*). Common Carp were introduced in the United States in the late 1800s as a food fish and are widely established. Silver, Bighead, Grass and Black carp are more recent introductions. They have been in the wild since the 1980s to 1990s. Each was imported for a different reason. Bighead Carp were imported to clean up aquaculture ponds and as a potential food fish. Silver Carp were introduced to clean phytoplankton filled waters, such as those at fish hatcheries and sewage treatment ponds. Grass Carp were imported as bio-control agents for aquatic weeds; and Black Carp, for control of aquatic snails in aquaculture ponds. All four species have escaped captivity and are now established in the region (Schofield et al. 2005, Nico and Jelks 2011).

The earliest species introduced to the lower Tennessee and Cumberland river basins were the Tench and American Shad. The American Shad (*Alosa sapidissima*) was stocked in the late 1800s as food for humans and forage for other fish, but failed to establish a population. Tench, a species from Europe, stocked in the late 1800s also failed to establish (Baughman 1947).

Threadfin Shad (*Dorosoma petenense*) were stocked as forage for sport fish. Atlantic Needlefish (*Strongylura marina*) and Blacktail Shiner (*Cyprinella venusta*) have gained access to the region via the Tennessee-Tombigbee Canal (Etnier and Starnes 1993). Aquarium species such as Goldfish (*Carassius auratus*), Red-bellied Pacu (*Piaractus brachypomus*), and Red-tail Catfish (*Phractocephalus hemioliopterus*) have been collected from the area, but are not known to be established. Most aquarium species are of low risk to the area because of their warm water requirements. Goldfish are an exception to this, however. They are established in many other areas of the country in cold waters (Fuller et al. 1999, Schofield et al. 2005). Three large Red-tail Catfish were collected in close proximity to one another on Lake Barkley in 2010. These fish can attain a length of 1.3 meters and a weight of 44 kilograms (Froese and Pauly 2011). Although they do require warm water, this species may have the ability to overwinter in thermal refugia such as power plant discharges.

Table 1. List of fish species introduced into the lower Tennessee and Cumberland drainages.

Scientific Name	**Common Name**	**Vector**	**Status**
Alosa sapidissima	American shad	Stocked for food and forage	Failed
*Carassius auratus**	Goldfish	Aquarium release	Established
*Ctenopharyngodon idella**	Grass carp	Stocked for bio-control	Established
Cyprinella venusta	Blacktail shiner	Canal connection	Established
*Cyprinus carpio**	Common carp	Stocked for food	Established
Dorosoma petenense	Threadfin shad	Stocked for forage	Established
Esox lucius	Northern pike	Stocked for sport	Established
*Hypophthalmichys molitrix**	Silver carp	Escaped aquaculture	Established
*H. nobilis**	Bighead carp	Escaped aquaculture	Established
Lepomis auritus	Redbreast sunfish	Stocked for sport	Established
L. miniatus	Redspotted sunfish	Unknown	Established
Morone chrysops x M. saxatilis	Wiper	Stocked for sport	Established
M. saxatilis	Striped bass	Stocked for sport	Established
*Mylopharyngodon piceus**	Black carp	Escaped aquaculture	Established
Oncorhynchus mykiss	Rainbow trout	Stocked for sport	Established
Perca flavescens	Yellow perch	Stocked for sport	Established
*Phractocephalus hemioliopterus**	Red-tail catfish	Aquarium release	Failed?
*Piaractus brachypomus**	Red-bellied pacu	Aquarium release	Failed
*Salmo trutta**	Brown trout	Stocked for sport	Established
Sander canadensis	Sauger	Stocked for sport	Established
Sander c. x S. vitreus	Saugeye	Stocked for sport	Established
Strongylura marina	Atlantic needlefish	Canal connection	Established
*Tinca tinca**	Tench	Stocked for sport	Failed

* foreign/exotic species

CONCLUSION

Twenty-three fish species have been introduced into the lower Tennessee and Cumberland drainages. Of these, only three species failed to establish. Stocking, aquaculture escapes, aquarium releases, and canal invasions via the Tennessee-Tombigbee Waterway are the vectors responsible for the introductions.

Management actions that could be undertaken to prevent new introductions include public education to inform aquarists not to release pet fishes and to inform anglers not to release unused, non-native bait and not to stock fish illegally. Although the Tennessee-Tombigbee Waterway has not been responsible for any major introductions to date, future introductions of harmful species could be prevented by installing barriers (such as electrical currents or bubble curtains) in the canal. Discontinuing stocking of existing non-native species likely would not have any effect because those species are already self-sustaining in the area. Aquaculture facilities may have better security now than when the Asian carps escaped. However, escapement from aquaculture facilities can be reduced by monitoring discharges, placing facilities outside flood zones, and developing effective disaster plans. Evaluation of cultured species is important to minimize the potential for establishment and spread.

Disclaimer

Acknowledgements

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FORESTS UNDER ATTACK BY NON-NATIVE INVASIVE INSECTS

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ABSTRACT. Nonnative invasive insects present one of the greatest threats to the integrity and viability of forest ecosystems. Increasing world trade and travel have amplified the risks of their inadvertent introduction into forests. While exclusion is the ultimate goal, some exotic insect pests enter North America undetected and become established. Examples of these pests include emerald ash borer (*Agrilus planipennis*), Asian longhorned beetle (*Anoplophora glabripennis*), Sirex woodwasp (*Sirex noctilio*), and hemlock woolly adelgid (*Adelges tsugae*). Each of these invasive pests threatens from one to several tree species as well as the ecosystems they help comprise. In some cases much is known about a particular pest and in others no information is available. Some of the challenges or obstacles that arise once an invasive insect is introduced include, knowledge of pest biology, developing detection and suppression methods, and understanding economic and ecological impacts. When these knowledge gaps occur an introduced pest can quickly cause significant damage. In many cases infestations can expand rapidly while ever increasing the scope of impacts. The resource management community must work in unison to prevent, detect, control or suppress these invasive pests in order to protect plant species and the ecological function of our forests.

CONTRIBUTED PAPERS
SESSION 1: BOTANY

Saturday, March 26, 2011

Moderated by:

L. Dwayne Estes

Center of Excellence for Field Biology

Austin Peay State University

FLORISTIC STUDIES IN THE CROSS CREEKS (STEWART COUNTY) AND DUCK RIVER UNIT (HUMPHREYS COUNTY) OF THE TENNESSEE NATIONAL WILDLIFE REFUGES

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ABSTRACT. The U.S. National Wildlife Refuge (NWR) System includes more than 550 units managed by the U.S. Fish and Wildlife Service for conservation and restoration of fish, wildlife, plants, and their habitats. Many are open at least part of the year for recreation and public use, including environmental education, hunting, fishing, hiking, bird watching, and various other activities. There are seven NWR's in Tennessee, where management for migratory waterfowl is the primary objective. Five of these, Chickasaw, Hatchie, Lower Hatchie, Lake Isom, and Reelfoot, are in the Mississippi River drainage. One, the Tennessee NWR, adjoins the Tennessee River (Kentucky Reservoir) while Cross Creeks NWR is bisected by the Cumberland River (Barkley Lake). All include a variety of habitat types but wetlands are dominant. This report provides an overview of floristic work in the two Middle Tennessee Refuges, i.e., Cross Creeks and the Duck River Unit of the Tennessee NWR.

CROSS CREEKS NATIONAL WILDLIFE REFUGE

This 3,588-ha (8,866-acre) NWR in Stewart County was established in 1962 as a replacement for the Kentucky Woodlands NWR in Lyon and Trigg Counties, Kentucky. That NWR was closed after it was mostly inundated by the newly-formed Barkley Lake (impounded Cumberland River). Cross Creeks NWR extends for about 16 river km (10 miles) on both sides of the Cumberland River (Barkley Lake) between the towns of Dover and Cumberland City. Most of the refuge is on river bottomland that was in cultivation prior to 1962; management is for resident, migratory, and over-wintering waterfowl. Wetland habitats dominate the landscape and include marshes, swamps, sloughs, wet meadows, dewatered zones, and bottomland forests. There are five permanent deep-water tributaries and 16 pools with water levels controlled by dams, levees, and water-gates. These pools are drawn down in summer, creating a dewatered zone where wildlife food crops are planted or native and introduced moist-soil species develop. Pool levels are then raised in autumn and winter, creating forage and cover for waterfowl. There is some agricultural production, mostly corn and soybeans, with portions left for wildlife, and a few successional fields and upland woods.

The flora has been studied 1975-present, including a floristic and vegetational analysis of a bottomland hardwood forest (Chester and Schibig 1993) and an extensive survey by Joyner and Chester (1994). More than 1025 vouchers housed at APSC document a vascular flora of 116 families, 367 genera, and about 640 species, including >20% that are non-native.

Four families include >1/3 of the flora, i.e., Asteraceae (74 taxa), Poaceae (68), Cyperaceae (42), and Fabaceae (34). The largest herbaceous genera are *Carex* (19 taxa), followed by *Polygonum* and *Cyperus* (12 each), *Panicum*-including *Dichanthelium* (9), *Eupatorium* and *Juncus* (7 each), and *Eragrostis* and *Ranunculus* (6 each). The largest tree/shrub genera are *Quercus* (10), *Carya* (5), and *Acer* and *Prunus* (4 each). Major genera of woody vines are *Smilax* (3) and *Vitis* (2).

Six state-listed taxa are known from the unit (Tennessee Natural Heritage Program 2008). Three are listed Threatened and two Special Concern (Table 1). Three formerly-listed taxa include *Paysonia lescurii* O'Kane & Al-Shehbaz, Lescur's Bladderpod (Brassicaceae), *Phacelia ranunculacea* (Nutt.) Const., Ocean-Blue Phacelia (Boraginaceae), and *Spiranthes ovalis* Lindl. var. *erostellata* Catling, October Ladies'-Tresses (Orchidaceae).

Table 1. State-listed taxa known from Cross Creeks NWR.

Heracleum maximum Bartr., Cow Parsnip (Apiaceae). Special Concern
Juglans cinerea L., White Walnut or Butternut (Juglandaceae). Threatened
Liparis loeselii (L.) Rich, Fen Orchid (Orchidaceae). Threatened
Pedicularis lanceolata Michx., Swamp Lousewort (Orobanchaceae). Special Concern
Sagittaria brevirostra Mackenzie & Bush, Midwestern Arrow-Head (Alismataceae). Threatened

DUCK RIVER UNIT, TENNESSEE NATIONAL WILDLIFE REFUGE

The Tennessee NWR encompasses about 20,780-ha (51,350-acres) in Benton, Decatur, Henry, and Humphreys Counties. Three discontinuous units, Big Sandy, Busseltown, and Duck River, extend for 105-km (65 miles) along the Tennessee River; the first two are on the west side of the River, the latter mostly on the east. The refuge was established in 1945 to coincide with the impounding of the River to form Kentucky Lake. Primary management is to provide food and protection for waterfowl as noted for Cross Creeks.

The Duck River Unit, mostly Humphreys County, includes 10,820-ha (26,740-acres), mostly in bottomlands of the impounded Duck and Tennessee Rivers that were agricultural prior to 1945. As in the case of Cross Creeks, wetlands are common and include the same types noted above. There are 12 man-made pools with regular draw-downs for development of native/introduced species and moist-soil plantings. Most of the drier bottomlands are cultivated in wheat, soybeans, milo, and corn. The unit also includes some upland forests with bluffs and outcrops, successional fields, home sites, and cemeteries.

The flora has been studied 1980-present, with an intensive survey in 2001-2003 (Gunn and Chester 2003); more than 1000 vouchers are on deposit at APSC. The flora is now known to consist of 125 families, 405 genera, and 725 species (>17% non-native). Large families accounting for more than 35% of the flora, are the Asteraceae (86 taxa), Poaceae (81), and Cyperaceae and Fabaceae (42 each). Large genera are *Carex* (21), *Panicum*-including *Dichanthelium* (12), *Cyperus* and *Solidago* (9 each), *Desmodium, Eupatorium, Polygonum* (8 each) *Eragrostis* (7), and *Aster, Bidens*, and *Ludwigia* (6 each). Large tree/shrub genera are *Quercus* (16), *Carya* (7), and *Acer* (4). *Smilax* and *Vitis* (4 each) are large woody vine genera.

Seven state-listed taxa have been documented (Table 2), included two listed Endangered, three Threatened, and two Special concern. One of these, *Echinochloa walteri* (Pursh) Heller, Coastal Cocks-Spur, represents a state record, and *Bolboschoenus fluviatilis* (Torr.) Sojak, River Bulrush, was documented for only the second time in the state.

Table 2. State-listed taxa known from the Duck River Unit, Tennessee NWR.

Bolboschoenus fluviatilis (Torr.) Sojak, River Bulrush (Cyperaceae). Special Concern
Heteranthera limosa (Sw.) Willd. (Pontederiaceae). Threatened
Liparis loeselii (L.) Rich, Fen Orchid (Orchidaceae). Threatened
Lysimachia fraseri Duby, Fraser's Yellow Loosestrife (Myrsinaceae). Endangered
Sagittaria brevirostra Mackenzie & Bush, Midwestern Arrow-Head (Alismataceae). Threatened
Spiranthes odorata (Nutt.) Lindl., Marsh Ladies'-Tresses Orchidaceae). Endangered
Echinochloa walteri (Pursh) Heller, Coastal Cocks-Spur (Poaceae). Special Concern

The seven Tennessee National Wildlife Refuges provide extensive and ideal landscapes for native flora, especially wetland species and some listed taxa. Floristic results from the two units presented here and from The Big Sandy Unit (in preparation by the first author), and the Reelfoot Lake-Lake Isom Units, summarized by Pyne (1989) and Heineke (1989), provide ample evidence for the floristic importance of these federal-controlled areas. Continued work in these units, and the other Tennessee units that have received less botanical attention, is warranted.

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FLORA AND VEGETATION OF SEVERAL GREAT VALLEY MESIC FOREST STANDS

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ABSTRACT. Fourteen samples of mesic forest vegetation from the Great Valley of East Tennessee were selected from a known larger sample for study of their flora and vegetation. The stands were examined about each three weeks during most months March through October or November, 2008 and 2009. Stands were located on lower slopes of narrow valleys or rarely open slopes of wide valleys. Substrate was usually sandstone but shale, limestone and dolomite underlay some stands. A flora of 526 taxa was found including nine rare or otherwise unusual plants. The flora were 4.8 percent introduced, 24.5 percent were woody, 5.9 percent pterodo-phytes, 7.4 percent grasses and 8.3 percent sedges. Sixty-two and one-half percent intraneous taxa occurred, northern and southern taxa had somewhat lower and higher percentages than in the Claiborne and Hancock County mesic flora sample. Forest vegetation types included beech-poplar (or poplar-beech) seven stands, beech-oak four stands, two sugar maple stands and a tulip poplar-oak dominated type. Relative importance of overstory mesic tree dominants may have varied in response to more moisture or lower temperatures of north or east aspects, average precipitation, elevation, distance to a protecting ridge to the south or west, and valley width.

INTRODUCTION

The flora and vegetation of the Southern Appalachians are relatively well known. The Blue Ridge and Cumberland Plateau have been areas of special study (Whittaker 1956, Golden 1974, 1981, White 1982, Wofford 1989, Radford et al. 1968, Schmalzer 1989, Hinkle et al. 1993). Studies of the local floras in the Valley include those of Bullington 1997, Wolfe 1956, Oxendine 1971, Hedge 1978, Mann et al. 1985 and Van Horn 1981, 1981, 1986. Vegetation studies include those of Martin 1971, 1989, Martin and De Selm 1976, De Selm 1993, 2006, and Stephenson et al. 1993 who describe several communities. This study includes results of the study of 14 mesic forest stands from the central and southern Valley and Ridge of Tennessee during 2008-2009 (Figure 1) and includes floristic data from years as early as 1990. Objectives were to compare results with the similar study of stands from the northern edge of the State (De Selm 2006) and to examine whether latitude, soil characteristics and site topographic characteristics may have influenced vegetation and floristic characteristics.

THE STUDY AREA

The study area was chiefly in the Valley and Ridge Physiographic Province of Fenneman (1938) which lies between the Cumberland Plateau and Mountains and the Blue Ridge in this study area. The area lies within the Tennessee River drainage and is characterized by ridges underlain by Paleozoic sandstones, limestones, dolomites and shales which have been extensively faulted and folded. Erosion has produced resistant rock ridges and softer bedrock (as shale or soft limestone) valleys trending northeast-southwest (Hardeman 1966, Rodgers 1953, Milici et al. 1972). Stand elevations range from 780 to 1060 feet although some Valley ridges rise as high as an elevation of slightly over 3000 feet. Stand slope angles ranged from 22 to 106 percent-median 55 (six stands also had sections of vertical cliffs). Sites consisted of lower slopes, sometimes with a toeslope and usually and adjoining valley terrace which was at least one meter above stream level.

Soils were mapped as Udults, Hapludults, Paleudults, and Rhodudults (U.S. Soil Conservation Service 1970). Soil textures were (county soil surveys) stony or gravelly, or rolling stony land, or fine sandy loams, loams or silt loams. Eleven series were mapped.

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† Dr. Hal R. De Selm died on July 12, 2011 in Knoxville, Tennessee.

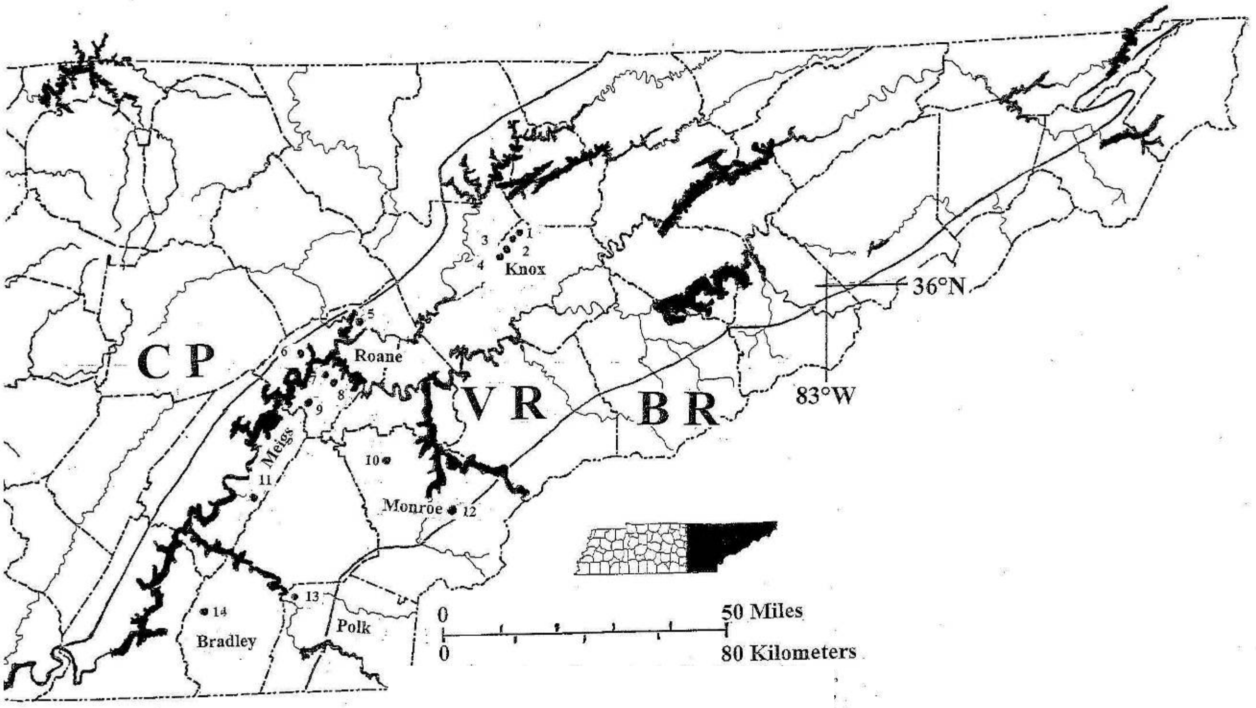

Figure 1. Map of East Tennessee showing CP-Cumberland Plateau, VR-Valley and Ridge, and BR-Blue Ridge physiographic regions, certain counties, and stand locations. Numbers are those of Table 1.

The climate is temperate with well-distributed precipitation varying from 48-54 inches over the area with some usually short summer droughts (Dickson 1960, De Selm and Schmidt 2001). Between 1890 and 1959, the average Tennessee River basin precipitation was 51 inches but varied from 37 to 64 inches for stations above Chattanooga (numbers include Blue Ridge mountain stations). Twenty-five years had precipitation below 51 inches (Tennessee Valley Authority Hydrolic Data Branch 1959). Variability was considerable. High precipitation events occurred, which resulted in floods. The flood events of 1867, 1948, and 1957 are well documented (Tennessee Valley Authority Division of Water Control Planning 1949, 1958). The lower end of the Chestuee Creek site may be flooded during high water periods on the Creek. Sometimes parts of various stand terraces may be temporarily flooded during winter storms. Occasional snowfall events may cause forest canopy losses. During the period November through March 1960, snowfall in the area of the study sites ranged from 30 to 55 inches, much more than the usual 10-12 inches for an entire winter season (Tennessee Valley Authority Division of Water Control Planning 1960).

While the precise size of the Valley vascular flora is not known, the Oak Ridge Reservation flora, of 840 taxa in parts of two counties (Mann et al. 1985), suggests that the Valley flora may be at least twice as large. Stevenson et al. (1993) place the area of the sample sites in the Appalachian Oak Forest Region of the eastern Deciduous Forest where mixed mesophytic forests have been described (Braun 1950, Martin 1971, 1989, Martin and De Selm 1976, De Selm 2006).

Land use practices include those of the Native Americans who cleared fields for crops and villages and burial sites and who hunted game animals and edible plants through the area (Swanton 1944, Lewis and Kneberg 1958). After about 1790, Euroamericans began settling the land and they logged the forest, cultivated row crops on lowlands, and put stock out to graze in the forests. Surface fires were set in spring to increase forage growth (Folmsbee et al. 1969, Killebrew et al. 1874).Forest use and misuse resulting in floristic and vegetational depletion continues as does the introduction of new diseases, insect and other animal pests and weeds (Nolt et al. 1997, Nash 2000).

METHODS

Stands were chosen for study in 2008 from a large number of mesic Valley stands found and examined 1998-2002 (Tables 1, 2); these were originally selected on the basis of no recent disturbances (especially logging and grazing), some canopy trees in the 24-30-inch class, and for floristic richness. During this first period of examination, tree diameters were obtained and some site factors were measured or estimated. Twenty stands were examined during most months, March-November 2008; in 2009 the set was reduced to 14 and examined with some repetition March-October about each three weeks. In each stand, a 100-200 meter transect was walked during each period of examination. Species present were recorded, unknowns collected and later determined using standard manuals and facilities and personnel at TENN. Literature sources included Gleason and Cronquist (1991), appropriate volumes of the *Flora of North America* (Flora of North America Editorial Committee 1993, et seq.), Chester et al. 1993, 1997, Case and Case 1991, Jones 2005, Snyder and Bruce 1986, and Wofford and Chester 2002. Nomenclature follows chiefly Chester et al. 2009, but with a few names from *Flora of North America*. Floras were compiled by site (Appendix I).

Table 1. Descriptors of sites used in the mesic forest study

1. Halls Gap (98-188) Knox, Big Ridge Park, 1998-2009, ravine, Rome Form., ss, NE, beech-tulip poplar.
2. Hill Rd. (90-583, 553) Knox, Big Ridge Park, 1990-2009, ravine, Rome Form., ss, NE, beech-oak.
3. Andersonville Pk. (98-186) Knox, Big Ridge Park, 1990-2009, ravine, Rome Form., ss, NE, tulip poplar-beech.
4. Rt. 441 (98-199) Knox, Fountain City, 1990-2009, ravine, Rome Form., ss, NE, beech-tulip poplar.
5. Elverton Rd. (08-01) Roane, Elverton, 2008-09, side slope, Pennington, sh, si, dol, ls, ss, NW, tulip poplar-beech.
6. Tanglewood Ln. (94-511) Roane, Harriman, 1994-2009, side slope, Newman, ls, NW, sugar maple-beech.
7. Laurel Bluff Church (99-130) Roane, Bacon Gap, 1990-2009, ravine, Rome Form., ss, NE, beech-oak.
8. Myers Rd. (99-124) Roane, Cane Creek, 2008-09, ravine, Newalla Form., ls, dol, E, sugar maple-oak.
9. Barnard Narrows (93-539) Roane, Bacon Gap, 1993-2009, ravine, Rockwood Form., ss, N, NE, beech-oaks.
10. Anderson Rd. (00-19) Monroe, Madisonville, 2000-09, ravine, Holston Form., ss, ls, NW, beech-tulip poplar.
11. Legg Hollow (90-600) Meigs, Decatur, 1990-2009, ravine, Rome Form., ss, NE, beech-tulip poplar.
12. Three Points Rd. (00-34) Monroe, Vonore, 2000-09, ravine, Bays Form., si, sh, NW, beech-tulip poplar.
13. Chestuee Ck. (00-99) Polk, Benton, 2000-09, ravine, Conasauga, sh, NE, tulip poplar-oak.
14. Laurel Bluff Rd. (90-517) Bradley, South Cleveland (1990-2009) open slopes, Conasauga, sh, NE, oak-beech.

[a] Sequence and abbreviations: number (north-south) stand number (date tree sampled) county, quadrangle, range in years seen, position, geologic formation, rock type, aspect, canopy dominants (based on relative density).

[b] Abbreviations: Rd. = road, Pk. = pike, Rt. – route, Bl. = bluff, Ck. = creek, Form. = formation, ss = sandstone, ls = limestone, sh = shale, si = silt stone, dol. = dolomite. Quadrangles (names following county) are USGA 7 ½ minute, Tennessee maps.

Table 2. Geographic description of stand locations.

1. Halls Gap, in Bull Run Ridge on unnamed branch of Bull Run Creek, latitude 36.17°
2. Hill Road in Bull Run Ridge on unnamed branch of Bull Run Creek, lat. 36.16°
3. Andersonville Pike in Bull Run Ridge on Smith Branch of Bull Run Creek, lat. 36.14°
4. Route 441 in Bull Run Ridge on unnamed branch of Bull Run Creek. Lat. 36.12°
5. Elverton Road in Cumberland Plateau outlier, on ridge east of Elverton Branch of the Little River embayment of Watts Bar Lake. Lat. 35.97°
6. Tanglewood Lane on unnamed ridge, D. Wright, owner, 0.7 miles east of the eastern crest of Walden Ridge. Lat. 35.96°
7. Laurel Bluff Church on Hurricane Ridge in Laurel Bluff Branch Gap. Lat. 35.80°
8. Myers Road, unnamed creek valley north from Huffine Springs on Stamp Creek Ridge. Lat. 35.79°
9. Barnard Narrows is a gap in Bacon Ridge cut by Barnard Narrows Branch. Lat. 35.76°
10. Anderson Road in Red Branch Valley of Bat Creek Knobs. Lat. 35.58°
11. Legg Hollow, Dry Fork Branch Gap in No Pone Ridge. Lat. 31.52°
12. Three Points Road on a branch of East Fork Hicks Creek embayment of Tellico Lake, in the declared boundary of Cherokee National Forest, 0.8 miles west of the Smoky Mountain fault. Lat. 35.30°
13. Chestuee Creek in an unnamed valley branch of Chestuee Creek south from the former Chestuee Mills. Lat. 35.23°
14. Laurel Bluff Road at the junction of an unnamed branch and Runner Branch on an unnamed ridge east of Rabbit Valley. Lat. 35.18°

Slope angle and aspect were measured in the field. County soil surveys were used to obtain site soil series textures. Topographic maps (USGS 7 ½ minete quadrangles, Tennessee) were used to estimate elevation, site latitude, valley bluff-to-bluff width, valley (gap) length, distance to sheltering ridge to the south or west, and ridge height above the stand.

Aspect was converted to numerical scale as in Frank and Lee (1966). Soil texture, as loam, silt loams and fine sandy loam have nearly the same water holding capacity (Longwell et al. 1965, whereas gravelly and stony soil holds less water (Hannah 1968) though actual series stone percentage was not stated (Swann et al. 1942). In the limestone derived series, shallow depth and high clay content may reduce available moisture levels (Carman 1970). Most other site factors above have been used with success by Tennessee researchers to predict species importance in the Valley by Martin (1971), Hedge (1979), Crownover (1983) on the Cumberland Plateau by Hinkle (1975, 1978), Smith (1977), Wade (1977), White and Miller (1988) and others. Vegetation stand data was converted to relative species density. Numbers of tree, shrub, vine, and herb taxa were converted to percent of the total flora by stand; herb types, as sedges, were relativized to total stand flora and to total stand herbs.

Tree fall gaps in stands were few but one large tree fell in each of three stands in 2008 taking down adjacent trees and limbs creating gaps 100-400 m^2, and in one stand, cutting the transect short. The growing season of 2009 had about 25 percent more precipitation than average, but the potential change on taxa seen per month in each stand in 2009 compared to 2008 was not seen.

RESULTS AND DISCUSSION

Flora

The flora of these mesic stands totaled 526 vascular taxa (Appendix). This exceeded the flora of 15 mesic stands from Claiborne and Hancock counties (of 380 taxa) by about 38 percent (see De Selm 2006). The set of sites from this study occurred in six counties—the most distant stands were 178 miles apart, whereas in the earlier study the latitudinal array was about eight miles. Many taxa are in common with the floras of the Oak Ridge Reservation (Mann et al. 1985) and the studies of Oxendine (1971), Hedge (1979), Bullington (1997), Wolfe (1956), and Van Horn (1981). The overstory floras were very similar to those detailed for the mesic types noted by Martin (1979), Martin and De Selm (1976), and De Selm (2006).

Rare plants in the flora included *Carex gravida, C. hirtifolia, Euonymus obovatus, Gratiola floribunda, Hydrastis canadensis*, and *Panax quinquefolia* (Bailey 2003). Also collected were the seldom seen *Bromus latiglumis* and *Poa wolfei*, known by Gattinger (1901). The cultivar *Euonymus japonicus,* rarely collected (few sheets at TENN), was found escaped at one site. It was noted as present in North Carolina and Georgia (Meyer et al. 1993).

The flora was very similar to that reported in the previous study of Valley mesic stands (De Selm 2006) with few Appalachian endemics and a typical presence class distribution. The flora included 4.8 precent introduced, 24.5 woody, 5.9 pteridophytes, 7.4 grasses and 8.3 percent sedge taxa. Intraneous taxa were 62.5 percent, local intraneous 4.7, northern 16.7, southern 15.7 and western 1.1 percent of the flora. These percentages were similar to others calculated for other samples of the Valley flora (Oxendine 1971, Hedge 1979, De Selm 1993). The northern percent was lower and the southern percent was higher than those of the mesic forest samples from the more northern Claiborne and Hancock counties (De Selm 2006). The differences averaged 7.4 percent.

Stand Comparison

Stand floras varied in richness from 157 to 203 taxa, tree taxa numbers were 28 to 43, shrubs-vines were 17 to 46, and herbs were 107-150 taxa. Grass taxa were 5-19, sedges 4 to 15, and pteridophytes 3 to 9 taxa. Percent tree, grass and sedge seemed to vary negatively with site floristic richness: trees lost about seven percent per 100 total taxa, grasses lost five percent and 3.5 percent in sedges. Shrubs and vines increased seven percent over 100 total species change but apparently pteridophtes were not affected.

A list of the 79 most abundant species of several western North Carolina cove forests has appeared (Jackson et al. 2009). Twenty-four western North Carolina mountain mesophytes were absent from the Valley sample. The other 55 occurred in the Valley sample floras with a presence of 7-100 percent with a median of 42 percent. The percent presence varied by site latitude with about 10 percent species loss per degree southward.

Vegetation Types

Mesic vegetation types seen by Martin (1971) were a group dominated by beech (seven variants with second dominants ranging from buckeye, sugar maple, to oaks or pine) and types dominated by sugar maple with the second dominant basswood or mixed hickories and a type dominated by northern red oak with tulip poplar as second dominant.

Mixed mesophytic vegetation seen in the earlier Valley study of this series (De Selm 2006) were three types dominated, in various combinations, by hemlock, beech and basswood. There were eight types dominated by mixtures of sugar maple, beech, tulip poplar, buckeye, and basswood. And there was a sugar maple-white oak and a sugar maple-yellow oak type.

The mesic vegetation types from this study included beech-tulip poplar (or poplar-beech) dominating seven stands. These were similar in general composition to the beech types of Martin (1971) but with less variation. Beech-tulip poplar dominated stands did not occur in the northern sample of mesic forests of De Selm (2006) but were sampled by Jordan (1976). The beech-oak (or oak-beech) types were similar to the beech-oak types of Martin (1971), but here, the order of importance of oaks was northern red, white and chestnut oak. The beech-oak stands also occurred in the northern mesic forest samples. The two sugar maple types were also similar in composition to those of Martin but the second dominants here were beech or white oak whereas in Martin's sugar maple stands, they were basswood or hickory. Five stands with sugar maple the leading dominant occurred in the northern counties mesic forest samples. Sugar maple types were seen by Jordan (1976). In five northern stands, the leading dominates were of hemlock, buckeye and basswood, whereas among the southern types these taxa were not the leading dominants of any stand.

Stand (forest) types were sorted by bedrock type. Nine of the 14 stands occurred on sandstone and beech was the leading dominant. The limestone and dolomite substrates supported two sugar maple types with beech and oaks. The siltstone and shale substrate supported three different types with first or second dominant beech, tulip poplar or oaks.

A comparison of forest types and floristic composition revealed that stands high in beech had more total taxa and a higher percent herbs, grasses and sedges. Those with high sugar maple had a high percent mesic trees, shrubs-vines and total herbs.

Each species may have been responding to the site characteristics individualistically (Gleason 1926, Whittaker 1956, Curtis 1959) with on-site competition for resources causing interactive species restriction (Barbour et al. 1999). Both positive and negative competitive allelochemical interactions occur which influence species success (Rice 1989) but plant allelochemicals also influence litter and soil microorganisms and chemical actions from both sources influence the degree of successful activities of mycorrhiza and disease organisms (Rice 1995) and this added interactions also influence survival and growth of community dominants. Allelochemical mechanism found in one case was a root-produced phytotoxin which triggered a "genome-wide" change in gene expression and ultimately death of the root system (Bais et al. 2003).

Relationships of Stand to Geographic Location and Site

Calculation of tree stand composition percent by species and species group enabled plotting of such data against geographic and site data. In some cases, point dispersion indicated or at least suggested a positive or a negative relationship. The north-to-south order of stands indicated southward, a higher percent beech, a lower percent of western North Carolina mesic taxa, a higher percent herbs and lower shrub-vine and grass taxa. Comparison with average precipitation (De Selm and Schmidt 2001) suggested a higher oak percent with lower precipitation and perhaps more sedges with higher precipitation.

Stand elevation data indicated that, with increased elevation, percent mesic trees and percent beech increased, but total percent trees was reduced. Stand aspect indicated higher percent beech, northern red oak, sedges as percent of herbs and less tulip poplar on the more mesic northeast-facing slopes.

Distance to the protecting south or west ridge indicated increased beech with less distance (more protection) while more sugar maple, northern red oak and more total oaks with greater distance occurred. The stands valley (usually gap) width measurements indicated that wider valleys had increased percent shrub-vines; northern red oak, sugar maple and total stand tree percentages increased whereas the beech percent decreased.

Using site and soil factors, the stands were arrayed from mesic to submesic. Factor data was converted to a similar low number scale with higher numbers referring to more soil moisture. Toward the mesic end of the array, there were: higher mesic tree percent, lower total tree and oak percents, higher grass and lower shrub-vines and ferns.

Factors influencing stand floristic composition included a history of land use, topography and sampling transect location and length. Stand histories were unknown and in the nearly five percent non-native floras may have been introduced during disturbance periods. The largest trees in the stands: 20 inches in one, 24 inches in four and 26-36 in nine stands suggest that logging practices have removed larger trees within the past 50-150 years. The largest trees among all stands were beech 37 percent, white oak 20, tulip poplar 13, northern red oak 10 and others less: sweetgum, hemlock, shagbark hickory and loblolly pine. Braun (1950) and Martin (1971, 1992) reported several mixed mesophytic stands with a high beech percent. Old logging practices were believed to have often left the then-useless beech in logged stands. Invaders into modern stands are known to be the maples (Abrams 1998, 2002, Frelich and Reich 2002, Franklin 2002) pines and tulip poplar (Burns and Honkala 1990). Here, these total 25.6 (range 7.1-52.1) percent of stems. Pines and red maple only occurred in six stands. Stand diameter distribution of maples, oaks, hickories, tulip poplar and pine are asymptotic—without the inverse "J" shapes of some old growth (White and White 1996). The numbers of overstory sample tree taxa in the beech stands was 36 as seen by Martin (1992).

SUMMARY AND CONCLUSIONS

Mixed mesophytic vegetation was reported from the Valley and Ridge by Braun (1950) and later in Tennessee by Wolfe (1956), Martin (1971), Crownover (1983), Stephenson et al. (1993), De Selm (2006), Jordan (1976) and other workers. The present study of 14 stands from the central and southern part of the Valley in Tennessee contrasted with the earlier study (De Selm 2006) of 15 stands located in two counties bordering Kentucky. In the present study, the flora was larger, probably by reason of the stands' latitudinal dispersion, floras of individual stands were large (compare with data cited by Parker 1989) and compared to the northern set, dominants were fewer. Beech was first or second dominant in 10 stands here versus four in the north set. Oaks and tulip poplar were also more important southward. Sugar maple dominated fewer stands and hemlock, buckeye and basswood, although present in most stands, dominated none. Floras of the stands ranged from 157-203 taxa, tree taxa ranged from 28 to 43 taxa including 10-21 seedlings and saplings included in the tree sample count (low, mostly below the range in Parker 1989). Shrubs-vines taxa per site averaged 31, with a range of 11 to 46. Herb taxa averaged 120 per site with a range of 100-151 which much exceeded data cited by Parker (1989).

Past logging, perhaps stock use, and road building doubtless spawned introduction of the non-native floristic element of nearly five percent. Logging probably has influenced bole diameter distribution here ranging to 36 inches and asymptotic. The, "inverse J distribution ... serve only as a guide" (Martin 1992).

The flora of this group of stands totaled 526 taxa which was 38 percent larger than that of the northern set, otherwise had similar characteristics although the northern and southern percentages were smaller and larger, respectively.

When stand characteristics were compared to stand latitude, there was an increased percent tulip poplar, shrubs-vines and western North Carolina mesophytes northward. Beech and herb percents increased southward. In the middle of the distribution centering in Roane County, there was increased percentages of trees, sugar maple, oaks, northern red oak, and grass. With more precipitation, the percent sedges and possibly oaks increased although northern red oak peaked in the mid precipitation range (140 cm). With increased stand elevation, the percent trees decreased but mesic tree and beech percents increased. Stands with a generally north aspect had more beech, northern red oak, grass and sedge percents, but less tulip poplar. Valleys, when of narrow width, were related to lower shrub-vine, sugar maple, northern red oak but more beech and a higher percent trees. Distance to the south or west ridge showed that with a more distant ridge (less protection) there was less beech, but more sugar maple, northern red oak and shrubs-vines. Using geographic and site factors, the stand array at the more moist end had higher percent mesic trees, fewer oaks, and fewer shrub-vines but more grass, fewer sedges and fewer ferns as percent of herbs.

Totaling stems in all stands of the potential invaders/increasers, red and sugar maple, tulip poplar and the pines, the stems numbered 45, 138, 165 and 15 with overall ratios of 5-11 inch versus 12+ inches being 2.4:1. The ratio of sugar maple and tulip poplar for those size classes was 11:1. However, red maple small:large was 4:1 suggesting a more recent invasion or increase (Abrams 1998). The sugar maple ratio was 2.4:1 suggesting an invasion/increase beginning in a somewhat earlier period. The tulip poplar and pine ratios of 1.2:1 and 1.1:1 suggest fairly long time community presence but with low species reproduction in the presence of shade tolerant and shade producing competitors (Burns and Honkala 1990). Diameter distributions of several invader/increaser taxa was asymptotic. The tree-sized taxa in the beech stands totaled 36 percent as seen in the old growth Lilley Cornett woods (Martin 1992).

Modification of views of vegetation development from multiple species, which represented virtual uniformity in life forms (Weaver and Clement 1929), to individualistic behavior (Gleason 1926 and others) has led to the recognition of multiple variants of a community as seen here. These dominants may still be in establishment phases following some long-term environmental (as climate) change or within decades (man-caused) disturbances. Each responded to the geographic macroclimatic factors and site microclimatic and edaphic factors individualistically (Wolfe et al. 1949). They may respond to the invasion and competition from weeds as *Lonicera* spp. and *Microstegium* probably individualistically (Bartlow and others 1996). Species interactions may be affected by their tolerance to shade and root competition for water and nutrients (Caldwell and Pearcy 1994). These interactions may be modified by allelopathic substances which may influence the source species growth, and those of neighbors, either positively or negatively as would such interactions between litter and soil organisms influence each other and the roots of organisms above them (Rice 1984, 1995). These communities are currently being modified by herbivores especially the growing deer herd (McShea et al. 1997), exotic animals (Bratton 1982, Kilgo 2009), insect pests (Montgomery et al. 2004) and are subject to atmospheric pollution (McLaughlin et al. 2007) and climate change (Greenland et al. 2003). Will the vegetation survive organism's attacks, and environmental changes? Calculations by Iverson et al. (1999) and projections by

Woodall et al. (2009) based on observations of differential "seedling success" indicates that "northward tree migration in the eastern United States is currently underway with rates approaching 100 km/century in many species." Further study may reveal whether local vegetation, including types such as those discussed here, might in the future remain floristically and structurally constant or change floristically or structurally or even modify their boundaries.

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APPENDIX
Flora of 14 Mesic Forest Stands[a][b]

Acalypha virginica L. – 5.
Acer negundo L. – 1, 2, 3, 4, 5, 7, 9, 11, 12, 13, 14.
Acer rubrum L. – 1, 2, 3, 4, 5, 7, 9, 10, 11, 12, 13, 14.
Acer saccharum Marsh. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Actea pachypoda Ell. – 2, 3, 6, 7, 8, 9, 11.
Adiantum pedatum L. – 1, 2, 3, 4, 6, 7, 9, 10, 11, 12, 13, 14.
Aesculus flava Aiton – 1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14.
Aesculus pavia L. – 10.
Ageratina altissima (L.) King and H. E. Robins. – 1, 2, 3, 4, 5, 6, 9, 11, 13. *Eupatorium*
Agrimonia parviflora Aiton – 3.
Agrimonia pubescens Wallr. – 1, 3, 7, 8, 11.
Agrimonia rostellata Wallr. – 2, 5, 6, 8, 9, 12, 13, 14.
Agrostis perennans (Walt.) Tuckerman – 2, 5.
**Albizia julibrissin* Durazz. – 1, 5, 6.
Allium canadense L. – 1, 4, 8, 10, 13, 14.
**Allium vinale* L. – 1, 2, 3, 4, 6, 7, 8, 12, 13, 14.
Alnus serrulata (Aiton) Willd. – 1, 2, 3, 5, 7, 8, 10, 14.
Amelanchier arborea (Michx. f. Fern.) – 2, 7, 10, 13, 14.
Amorpha fruticosa L. – 1, 12.
Ampelopsis arborea (L.) Koehne – 5.
Ampelopsis cordata Michx. – 3, 7, 13, 14.
Amphicarpaea bracteata (L.) Fern. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Anemone acutiloba (DC.) G. Lawson – 4, 6, 10, 11, 12. *Hepatica*
Anemone americana (DC.) H. Hara – 1, 2, 3, 4, 6, 7, 8, 9, 11, 12, 14. *Hepatica*
Anemone quinquefolia L. – 2, 3, 4, 8, 10, 12, 14.
Anemone virginica L. – 1, 2, 3, 8, 9, 11, 12, 13.
Angelica venenosa (Greenway) Fern. – 2, 13.
Antennaria plantaginifolia (L.) Hook. – 1, 2, 10.
Antennaria solitaria Rydb. – 2, 5.
Apios americana Medik. – 10.
Aplectrum hyemale (Muhl. ex Willd.) Nutt. – 7, 8, 10, 12.
Aralia spinosa L. – 10, 13.
Arisaema dracontium (L.) Schott – 13.
Arisaema quinatum (Nutt.) Schott – 11.
Arisaema triphyllum (L.) Schott – 1, 2, 3, 6, 7, 8, 9, 11.
Arnoglossum atriplicifolium (L.) H. E. Robins. – 2, 3, 4, 5, 7, 10, 11, 12, 14. *Cacalia*
Aruncus dioicus (Walt.) Fern. – 1, 2, 3, 6, 7, 9, 10, 11.
Arundinaria gigantea (Walt.) Muhl. – 1, 2, 13.
Asarum canadense L. – 2, 13.
Asclepias quadrifolia Jacq. – 9.
Asimina triloba (L.) Dunal – 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Asplenium platyneuron (L.) B.S.P. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13.
Asplenium resiliens Kunze – 7, 8, 13.
Asplenium rhizophyllum L. – 3, 4, 6, 8, 9, 12.
Astilbe biternata (Vent.) Britton – 2, 3, 6, 7, 12.
Athyrium felix-foemina (L.) Roth ssp, *asplenioides* (Michx.) Farw. – 2, 3, 5, 6, 7, 8, 10.
Aureolaria laevigata (Raf.) Raf. – 1, 2.
Aureolaria virginica (L.) Pennell – 2, 13.
Betula lenta L. – 5, 12.

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[a] Some older generic names have been added.
[b] Stand number corresponds to those in Table 1.

Bidens bipinnata L. – 3, 9.
Bidens frondosa L. – 1, 4, 7, 13, 14.
Bidens vulgata Greene – 5.
Bignonia capreolata L. – 1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14.
Boechera laevigata (Muhl. ex Willd.) Al-Shehbaz. – 3, 8, 9, 11, 13. *Arabis*
Bohemeria cylindrica (L.) Sw. – 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Botrypus virginianus (L.) Holub – 1, 2, 3, 4, 5, 8, 9, 10, 12, 13. *Botrichium*
Brachyelytrum erectum (Schreb.) P. Beauv. – 1, 2, 7, 8, 9, 10, 11.
Bromus latiglumis (Scribn. ex Sheer) Hitchc. – 2, 3.
Bromus pubescens Muhl. ex Willd. – 1, 2, 3, 11, 12.
**Buglossoides arvense* (L.) I. M. Johnston – 1, 8, 11, 13, 14. *Lithospermum*
Calycanthus floridus L. – 5, 10, 14.
Campanula americana L. – 1, 2, 3, 6, 9, 10.
Campanula divaricata Michx. – 1, 3, 5, 14.
Campsis radicans (L.) Seem. ex Bureau – 1, 2, 3, 4, 5, 9, 10, 13, 14.
Cardamine bulbosa (Schreb. ex Muhl.) B.S.P. – 13.
Carex albicans Willd. ex Speng. – 2, 3, 6, 9, 11.
Carex albursina E. Sheld. – 12.
Carex amphibola Steud. – 1, 4, 8.
Carex blanda Dewey – 2, 3, 5, 7, 8, 9, 10, 12, 13, 14.
Carex brunnescens (Pers.) Poir. – 1, 5.
Carex cephalophora Muhl. ex Willd. – 2, 13.
Carex cherokeensis Schwein. – 14.
Carex communis Bailey – 11.
Carex complanata Torr. & Hook. – 1, 5, 13.
Carex crebriflora Weig. – 14.
Carex crinita Lam. – 10.
Carex debilis Michx. – 4, 10.
Carex festucacea Schkur. ex Willd. – 13.
Carex floccosperma Dewey – 1, 5, 13.
Carex frankii Kunth – 7, 10, 11.
Carex gracilescens Steud. – 2, 4, 5, 6, 8, 9, 10, 12, 13, 14.
Carex gravida Bailey – 11.
Carex grayii J. Carey – 10, 13.
Carex grisea Wahlenb. – 1.
Carex hirtifolia Mackenzie – 6.
Carex kraliana Naczi & Bryson – 4, 6, 7, 8, 9, 12, 13.
Carex laxiflora Lam. 1, 2, 3, 4, 6, 8, 9, 11, 13, 14.
Carex leavenworthii Dewey – 2, 5.
Carex muhlenbergii Schkur. ex Willd. – 9.
Carex nigromarginata Schwein. – 1, 2, 5, 6, 8, 9, 11, 12, 14.
Carex normalis Mackenzie – 5, 9, 10.
Carex oxylepis Torr. & Hook. – 5, 8.
Carex pensylvanica Lam. – 3.
Carex prasina Wahlenb. – 8, 9, 11, 13.
Carex platyphylla J. Carey – 4, 10.
Carex projecta Mackenzie – 13.
Carex purpurifera Mackenzie – 6, 12, 13. 14.
Carex retroflexa Muhl. ex Willd. – 6, 14.
Carex rosea Schkur. ex Willd. – 1, 2, 5, 6, 7, 9, 10, 11, 12, 13, 14.
Carex sparganioides Muhl. ex Willd. – 5.
Carex squarrosa L. – 13.
Carex striatula Michx. – 2.
Carex styloflexa Buckley – 6, 8, 13.
Carex swanii (Fern.) Mackenzie – 4.
Carex venusta Dewey – 14.

Carex virescens Muhl. ex Willd. – 2, 4. 9.
Carex vulpinoidea Michx. – 11, 12.
Carpinus caroliniana Walt. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Carya cordiformis (Wang.) K. Koch – 2, 5, 6, 8, 11. 12, 13, 14.
Carya glabra (P. Mill.) Sweet – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Carya laciniosa (Michx. f.) G. Don – 2.
Carya ovalis (Wang.) Sarg. – 13.
Carya ovata (P. Mill.) K. Koch – 3, 4, 9, 12, 13.
Carya tomentosa (Lam. ex Poir.) Nutt. – 1, 2, 4, 5, 9, 11, 13, 14.
Caulophyllum thalictroides (L.) Michx. – 6, 11.
Ceanothus americanus L. – 9, 10, 11, 13.
**Celastrus orbiculatus* Thunb. – 3.
Celastrus scandens L. – 4.
Celtis laevigata Willd. – 9, 11, 13.
Celtis occidentalis L. – 1, 2, 3, 4, 6, 7, 8, 13, 14.
Cercis canadensis L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14.
Chaerophyllum tainturieri Hook. – 1, 2, 3, 5, 6, 8, 9, 11, 13.
Chamaecrista fasciculata (Michx.) Greene – 3, 9, 10. *Cassia*
Chamaecrista nictitans (L.) Moench – 1.
Chasmanthium latifolium (Michx.) Yates – 3, 5, 13.
Chasmanthium laxum (L.) Yates – 5, 10, 13, 14.
Chasmanthium sessiliflorum (Poir.) Yates – 10.
Chelone glabra L. – 1, 4, 9, 11.
Chimaphila maculata (L.) Pursh – 1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 14.
Chionanthus virginicus L. – 5, 7, 10, 13, 14.
Cimicifuga americana Michx. – 1, 2, 3, 4.
Cinna arundinacea L. – 2, 5, 6, 9, 10, 11, 13, 14.
Circaea lutetiana L. ssp. *canadensis* (L.) Asch & Magnus – 1, 2, 3, 4, 6, 8.
Claytonia virginica L. – 1, 2, 4, 5, 6, 9, 10.
**Clematis terniflora* DC. – 6.
Clematis virginiana L. – 1, 2, 3, 11.
Cocculus carolinus (L.) DC. – 1, 8, 9.
Collinsonia canadensis L. – 1, 2, 3, 5, 6, 7, 8, 9, 11.
Collinsonia verticillata Baldw. – 10, 12, 14.
**Commelina communis* L. – 2, 3, 6, 10, 14.
Conoclinium coelestinum (L.) DC – 10, 13.
Convolvulus L. sp. – 13.
Coreopsis auriculata L. – 12.
Coreopsis major Walt. – 1, 2, 10, 11, 13, 14.
Cornus amomum P. Mill. – 5, 10.
Cornus florida L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Cornus foemina P. Mill. – 5, 10.
Corylus americana Walt. – 2, 3, 5, 6, 8, 11.
Crataegus crusgalli L. – 5.
Crataegus intricata Lange – 13.
Crataegus marshallii Eggl. – 13.
Cryptotaenia canadensis (L.) DC. – 1, 2, 3, 4, 6, 7, 8, 9, 13, 14.
Cuscuta gronovii Willd. – 5, 7.
Cynanchum laeve (Michx.) Pers. – 13.
Cynoglossum virginianum L. – 8.
Cystopteris bulbifera (L.) Bernh. – 7.
Cystopteris protrusa (Weatherby) Blasdell – 6.
Danthonia spicata (L.) P. Beauv. ex Roem. & J. A. Shultes – 1, 2, 5, 8, 9, 10, 13, 14.
Delphinium tricorne Michx. – 3, 4.
Dentaria diphylla Michx. – 1, 2, 3, 4, 6.
Dentaria heterophylla Nutt. – 1, 2, 3, 4, 6, 9, 10, 13.

Dentaria laciniata Muhl. ex Willd. – 2, 3, 4, 6, 8, 9, 10, 12, 13.
Desmodium ciliare (Muhl. ex Willd.) DC. – 11, 13.
Desmodium glabellum (Michx.) DC. – 1, 5, 6, 9, 10, 11, 13, 14.
Desmodium glutinosum (Muhl. ex Willd.) Wood – 2, 3, 13.
Desmodium marilandicum (L.) DC. – 11.
Desmodium nudiflorum (L.) DC. – 1, 2, 3, 5, 6, 8, 9, 10, 11, 12, 13, 14.
Desmodium obtusum (Muhl. ex Willd.) DC. – 2, 13, 14.
Desmodium paniculatum (L.) DC. – 1, 2, 3, 5, 7, 9, 10, 11, 13.
Desmodium pauciflorum (Nutt.) DC. – 1, 2.
Desmodium rotundifolium DC. – 3, 13.
Desmodium viridiflorum (L.) DC. – 1, 11, 13.
**Dianthus armaria* L. – 1, 5.
Dichanthelium acuminatum (Sw.) Gould & C. A. Clark – 4, 5. *Panicun*
Dichanthelium boscii (Poir.) Gould & C. A. Clark – 1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 13.
Dichanthelium clandestinum (L.) Gould – 4, 7, 10.
Dichanthelium commutatum (J. A. Schultes) Gould – 1, 2, 3, 5, 6, 8, 9, 10, 11, 13, 14.
Dichanthelium dichotomum (L.) Gould – 5, 10.
Dichanthelium dichotomum (L.) Gould var. *microcarpon* (Muhl. ex Elliott) Freckmann & Lelong – 1, 5, 7, 8, 10, 13.
Dichanthelium laxiflorum (Lam.) Gould – 5.
Dichanthelium polyanthes (J. A. Schultes) Mohlenbrock – 10.
**Dioscorea polystachya* Turcz. – 1, 4, 8, 10, 11, 14.
Dioscorea villosa L. – 1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Diospyros virginiana L. – 1, 3, 6, 7, 8, 12, 14.
Diphasiastrum digitatum (Dill. ex A. Braun) Holub – 5. *Lycopodium*
Dryopteris intermedia (Muhl. ex Willd.) A. Gray – 7.
Dryopteris marginalis (L.) A. Gray – 7, 14.
**Elaeagnus umbellata* Thunb. var. *parviflora* (Wall. ex Royle) C. K. Schneid. – 1, 5, 6, 10.
Elephantopus carolinianus Raeusch – 5, 8, 9, 11, 14.
Elymus hystrix L. – 1, 2, 3, 4, 8.
Elymus riparius Wieg. – 14.
Elymus villosus Muhl. ex Willd. – 4, 10, 14.
Elymus virginicus L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 13, 14.
Epifagus virginiana (L.) W. Bart. – 1, 2, 4, 5, 6, 7, 9, 10, 11, 13.
Epigaea repens L. – 1, 5, 7.
Equisetum arvense L. – 1, 2.
Erigeron annuus (L.) Pers. – 1, 10.
Erigeron philadelphicus L. – 1, 2, 3, 4, 5, 6, 7, 9, 10, 11, 13, 14.
Erigeron pulchellus Michx. – 2, 11, 12.
Erigeron strigosus Michx. ex Willd. – 3, 9, 12.
Erigenia bulbosa (Michx.) Nutt. – 6.
Erythronium americanum Ker-Gawl. – 2, 3, 4, 5, 6, 9, 10, 12.
**Euonymus alatus* (Thunb.) Siebold – 5.
Euonymus americanus L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
**Euonymus hederaceus* Champ. & Benth. – 1, 2, 14.
**Euonymus japonicus* Thunb. – 1.
Euonymus obovatus Nutt. – 4.
Euphorbia carollata L. – 8, 11, 13.
Euphorbia mercurialina Michx. – 1, 2, 3, 8, 9, 11, 13.
Eurybia divaricata (L.) Nesom – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14. *Aster*
Eurybia macrophylla (L.) Cass. – 2.
Eutrochium fistulosum (Barratt) Lamont – 5, 6, 7, 10, 11, 13, 14. *Eupatorium*
Eutrochium purpureum (L.) Lamont – 1, 2, 3, 5, 7, 12, 13.
Fagus grandifolia Ehrend. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Festuca subverticillata (Pers.) Alex. – 1, 2, 3, 5, 6, 8, 9, 10, 11, 12, 13, 14.
Fragaria virginiana Duchesne – 1, 3, 6, 8, 9, 10, 11, 14.
Fraxinus americana L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.

Fraxinus pennsylvanica Marshall – 4.
Galactia volubilis (L.) Britton – 1, 3, 11, 13.
Galax urceolata (Poir.) Brummitt – 1, 4, 5, 7.
Galium aparine (L.) 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Galium circaezans Michx. – 1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13.
Galium obtusum Bigelow – 8, 10.
Galium pilosum Aiton – 2, 3, 6, 8, 9, 10, 11, 14.
Galium tinctorium L. – 10.
Galium triflorum Michx. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Gentianella quinquifolia (L.) Small – 2, 11, 12. *Gentiana*
Geranium maculatum L. – 2, 3, 4, 5, 6, 7, 9, 10, 11, 12, 13.
Geum canadense Jacq. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12, 13, 14.
Geum donianum (Trett.) Weakley & Gandhi – 1, 2, 3, 7, 8, 12, 13. *Waldsteinia*
Geum vernum (Raf.) Torr. & Gray – 1, 3, 11.
**Glecoma hederacea* L. – 13, 14.
Gleditsia triacanthos L. – 8.
Glyceria striata (Lam.) Hitchc. – 8, 10.
Goodyera pubescens (Willd.) R. Br. ex W. T. Aiton – 1, 5, 9, 10, 13.
Gratiola floridana Nutt. – 10.
Hamamelis virginiana L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 14.
Helianthus decapetalus L. – 1.
Helianthus microcephalus Torr. & Gray – 1, 2, 4, 5, 9, 10, 11, 12.
Helianthus tuberosus L. – 3.
Heuchera americana L. – 1, 2, 4, 5, 6, 7, 9, 11, 12, 13.
Heuchera villosa Michx. – 10.
Hexastylis arifolia (Michx.) Small var. *Ruthii* (Ashe) Blomquist – 1, 2, 3, 4, 5, 6, 7, 8, 12, 14.
Hieracium paniculatum L. – 5.
Hieracium venosum L. – 1, 2, 14.
Houstonia caerulea L. – 2, 5, 7, 9, 10.
Houstonia purpurea L. – 1, 2, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Hybanthus concolor (T. Forst.) Green – 6.
Hydrangea arborescens L. 1, 2, 3, 4, 5, 6, 8, 10, 13, 14.
Hydrangea cinerea Small – 5, 7, 8, 9, 10, 12.
Hydrangea radiata Walt. – 2, 4, 11.
Hydrastis canadensis L. – 6, 8.
Hydrophyllum canadense L. – 6.
Hymenocallis caroliniana (L.) Herbert – 4.
Hypericum densiflorum Pursh – 1.
Hypericum hypericoides (L.) Crantz – 5, 13, 14.
Hypericum punctatum Lam. – 1, 2, 4, 5, 6, 7, 8, 10, 13.
Hypericum stragalum P. Adams & Robson – 2, 3, 10.
Ilex decidua Walt. – 5.
Ilex opaca Aiton – 5, 9.
Impatiens capensis Meerb. – 1, 2, 3, 4, 5, 6, 7, 10, 11, 12, 13, 14.
Impatiens pallida Nutt. – 2, 3, 6.
Ipomoea lacunosa L. – 1, 3, 4, 8, 11, 14.
Ipomoea pandurata (L.) G. F. W. Mey. – 1, 4, 8, 10, 14.
Iris cristata Sol. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Jeffersonia diphylla (L.) Pers. – 3.
Juglans nigra L. – 1, 2, 3, 5, 6, 7, 8, 9, 11, 12, 13, 14.
Juncus biflorus Ell. – 5, 7, 8.
Juncus coriaceus Mackenzie – 1, 4, 5, 8, 10, 11, 13.
Juniperus virginiana L. – 1, 2, 3, 4, 8, 9, 10, 11, 12, 13, 14.
Kalmia latifolia L. – 1, 2, 3, 5, 7, 10, 11, 14.
Krigia biflora (Walt.) Blake – 5, 7, 8, 10, 13, 14.
Krigia dandelion (L.) Nutt. – 2.

Lactuca floridana (L.) Gaertn. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 13, 14.
**Lamium purpureum* L. – 3.
Laportea canadensis (L.) Weddell – 4, 6, 9, 13.
Leersia virginica Willd. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13.
**Lespedeza cuneata* (Dum.-Cours.) G. Don – 4, 5, 9, 10.
Lespedeza intermedia (S. Wats) Britton – 1, 11, 13.
Lespedeza procumbens Michx. – 3.
Lespedeza repens (L.) Bart. – 9.
Lespedeza stuevei Nutt. – 11.
Lespedeza violacea (L.) Pers. – 9, 11.
Lespedeza virginica (L.) Britton – 2.
Leucothoe fontanesiana (Steud.) Sleumer – 5.
**Ligustrum sinense* Lour. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13.
Lindera benzoin (L.) Blume – 1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 14.
Liquidambar styraciflua L. – 1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 13, 14.
Liriodendron tulipifera L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Lobelia cardinalis L. – 7, 9, 10.
Lobelia inflata L. – 2, 4, 8, 10, 11, 13.
Lobelia puberula Michx. – 6, 7, 9.
Lobelia siphilitica L. – 1, 3, 4, 6, 8, 9, 10, 11, 12.
**Lonicera japonica* Thunb. – 1, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
**Lonicera maackii* (Rupr.) Herder – 3, 5, 6, 14.
Luzula acuminata Raf. – 2, 4, 5, 6, 9, 11, 14.
Luzula bulbosa (Wood) B.B. Smyth & C. R. Smyth – 1, 5, 7, 8, 10.
Luzula echinata (Small) F. J. Herm. – 2, 3, 7, 12, 14.
Lycopus virginicus L. – 4.
Magnolia acuminata (L.) L. – 1, 2, 3, 4, 5, 6, 7, 12, 14.
Magnolia tripetala (L.) L. – 6, 7.
Maianthemum racemosum (L.) Link – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14. *Smilacina*
Matelea obliqua (Jacq.) Woods – 3, 8, 13.
Medeola virginica L. – 4, 7, 10, 11.
Melica mutica Walt. – 2, 3, 5, 8, 9, 10.
Menispermum canadense L. – 2, 3, 6, 9, 14.
**Microstegium vimineum* (Trin.) A. Camus – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Mimulus alatus Aiton – 7.
Mitchella repens L. – 5, 7, 10.
Mitella diphylla L. – 7, 9, 12.
Monotropa uniflora L. – 7.
Morus rubra L. – 2, 3, 4, 5, 7, 8, 9, 10, 11, 12, 13, 14.
Muhlenbergia schreberi J. F. Gmel. – 6, 8.
Muhlenbergia tenuifolia (Willd.) B.S.P. – 2, 3.
**Murdannia keisak* (Hassk.) Hand.-Mazz. – 14.
Nyssa sylvatica Marsh. – 1, 2, 3, 4. 5, 6, 7, 9, 10, 11, 12, 13, 14.
Obolaria virginica L. – 9, 12.
Onoclea sensibilis L. – 8, 11.
Osmorhiza claytonii (Michx.) C. B. Clarke – 1, 2, 11.
Osmorhiza longistylis (Torr.) DC. – 3, 7, 11.
Osmunda regalis L. var. *spectabilis* (Willd.) A. Gray – 10, 11.
Osmundastrum cinnamomea L. – 10. *Osmunda*
Ostrya virginiana (P. Mill.) K. Koch – 1, 2, 5, 6, 10, 12, 13, 14.
Oxalis dillenii Jacq. – 1, 5, 8, 10, 11, 12, 13, 14.
Oxalis grandis Small – 2, 3.
Oxalis stricta L. – 6, 7, 10.
Oxalis violacea L. – 2, 3, 4, 5, 8, 10, 12, 13, 14.
Oxydendron arboreum (L.) DC. – 1, 2, 4, 5, 6, 7, 9, 10, 12, 14.
Oxypolis rigidior (L.) Raf. – 11.

Packera aurea (L.) A. & D. Love – 2, 3, 9, 12. *Senecio*
Packera obovatus (Muhl. ex Willd.) Weber & A. Love – 1, 2, 3, 4, 8, 9, 12, 13.
Panax quinquefolius L. – 3, 6.
Panicum anceps Michx. – 5.
Parthenocissus quinquifolia (L.) Planch. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Passiflora lutea L. – 2, 5 9, 10, 13.
Pedicularis canadensis L. – 11.
Penstemon laevigatus (L.) Aiton – 2, 5, 13.
Persicaria virginiana (L.) Gaertn. – 1, 2, 3, 5, 6, 8, 9, 11, 12, 13, 14. *Polygonum*
Phaseolus polystachios (L.) B.S.P. – 11.
Phegopteris hexagonoptera (Michx.) Fee – 1, 2, 3, 6, 7, 10, 11, 12, 14. *Thelypteris*
Philadelphus pubescens Loisel. – 6, 7.
Phlox divaricata L. – 1, 2, 3, 4, 5, 6, 9, 11, 12, 13.
Phlox glaberrima L. – 1, 3, 4.
Phryma leptostachya L. – 1, 2, 3, 5, 6, 8, 9, 10, 11, 13, 14.
Physalis angulata L. – 6.
Phytolacca americana L. – 10, 12.
Pilea pumila (L.) A. Gray – 1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 13, 14.
Pinus echinata P.Mill. – 5, 9, 11.
Pinus strobus L. – 5, 9.
Pinus taeda L. – 13.
Pinus virginiana P. Mill. – 1, 2, 3, 4, 5, 7, 8, 9.
Platanus occidentalis L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 13, 14.
Pleopeltis polypodioides (L.) E. G. Andrews & Windham var. *Michauxiana* (Weath.) E. G. Andrews &Windham – 4, 6, 7, 8, 9, 14. *Polypodium*
Pluchea camphorata (L.) DC. – 9, 14.
Poa autumnalis Muhl. ex Ell. – 1, 2, 4, 5, 8, 9, 10, 11, 13, 14.
**Poa compressa* L. – 2.
Poa cuspidata Nutt. – 1, 2, 4, 6, 7, 8, 10, 11, 12, 13.
**Poa pretensis* L. – 1, 2, 4, 7, 14.
Poa saltuensis Fern. & Wieg. ssp. *languida* A. Haines – 8.
Poa sylvestris A. Gray – 1, 2, 13, 14.
Poa wolfei Scribn. – 5.
Podophyllum peltatum L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13.
Polemonium reptans L. – 4.
Polygonatum biflorum (Walt.) Ell. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 14.
Polystichum acrosticoides (Michx.) Schott – 1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Populus deltoides W. Bartram ex Marsh. – 5.
Potentilla canadensis L. – 1, 2, 3, 9.
Potentilla simplex Michx. – 1, 5, 8, 9, 10, 11, 12, 13.
Prenanthes altissima L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Prenanthes trifoliata (Cass.) Fern. – 6, 13.
Primula meadia (L.) A. R. Mast & Reveal – 8. *Dodecatheon*
Prosartes lanuginosa (Michx.) G. Don – 2, 4, 5, 6, 8, 11, 12. *Disporum*
Prunella vulgaris L. – 2, 10, 11, 12, 14.
Prunus angustifolius Marsh. – 5.
**Prunus hortulana* Bailey. – 4.
Prunus serotina Ehrh. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Pseudognaphalium obtusifolium (L.) Hillard & Burtt – 1, 2, 13. *Gnaphalium*
Pycnanthemum loomsii Nutt. – 2, 3, 5, 9, 11, 12.
Pyrularia pubera Michx. – 5.
Quercus alba L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Quercus coccinea Muenchh. – 2.
Quercus falcata Michx. – 1, 2, 4, 5, 6, 8, 9, 10, 11, 12, 13, 14.
Quercus muehlenbergii Engelm. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 14.
Quercus nigra L. – 13.

Quercus phellos L. – 13.
Quercus montana Willd. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12.
Quercus rubra L. – 1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Quercus shumardii Buckl. – 3.
Quercus velutina Lam. – 1, 4, 5, 6, 8, 9, 11, 13.
Ranunculus abortivus L. – 4, 6, 7, 11.
Ranuncularis fascicularis Muhl. ex Bigelow – 1.
Ranunculus hispidus Michx. – 1, 2, 3, 4, 6, 7, 8, 9, 10, 12, 13.
Ranunculus recurvatus Poir. – 2, 4, 6, 7, 8, 10, 11, 12.
Rhamnus carolinianus Walt. – 1, 2, 3, 4, 5, 13, 14.
Rhododendron maximum L. – 5.
Rhododendron periclymenoides (Michx.) Shinners – 9, 10.
Robinia pseudoacacia L. – 11, 13, 14.
Rosa carolina L. – 5.
**Rosa multiflora* Thunb. ex Murr. – 1, 2, 3, 5, 6, 8, 10, 11, 13, 14.
Rosa palustris Marsh. – 8, 9, 13, 14.
Rubus L. spp. – 1, 2, 3, 5, 6, 8, 9, 10, 11, 12, 13, 14.
Rudbeckia fulgida Aiton – 4, 6, 8, 10, 11.
Rudbeckia hirta L. – 11, 12.
Rudbeckia laciniata L. – 1, 3, 4, 7, 9.
Rudbeckia triloba L. – 6, 7, 9.
Ruellia carolinensis (J. F. Gmel.) Steud. – 1, 2, 3, 4, 5, 8, 9, 12, 13.
Ruellia strepens L. – 4.
Salvia lyrata L. – 1, 3, 7, 9, 10, 11, 14.
Salvia urticifolia L. – 3.
Sambucus canadensis L. – 1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Samolus valerandi L. ssp. *parviflorus* (Raf.) Hulten – 9.
Sanguinaria canadensis L. – 1, 2, 3, 4, 6, 12.
Sanicula canadensis L. – 1, 2, 3, 4, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Sanicula marilandica L. – 9.
Sanicula odorata (Raf.) K. M. Pryer & L. R. Phillippe – 1, 2, 4, 13.
Sanicula smallii E. P. Bickn. – 6, 8, 12.
Sanicula trifoliata E. P. Bickn. – 6, 9, 10, 13, 14.
Sassafras albidum (Nutt.) Nees – 1, 2, 3, 5, 6, 9, 10, 11.
Satureja vulgaris (L.) Fritsch – 8.
Sceptridium biternatum (Sav.) Lyon – 13.
Sceptridium dissectum (Spreng) Lyon – 1, 3, 4, 8, 9, 10, 11, 13. *Botrichium*
Scleria oligantha Michx. – 13.
Scrophularia marilandica L. – 11.
Scutellaria elliptica Muhl. ex Spreng. – 5, 6, 7.
Scutellaria elliptica Muhl. ex Spreng. var. *hirsuta* (Short & Peter) Fern. – 1, 10, 13.
Scutellaria integrifolia L. – 2.
Scutellaria ovata Hill – 9.
Sedum ternatum Michx. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 11, 12.
Silene stellata (L.) W. T. Aiton – 5, 11, 13, 14.
Silene virginica L. – 1, 2, 11.
Silphium asteriscus L. – 4.
Silphium asteriscus L. var. *trifoliatum* (L.) Clev. – 3, 9.
Sisyrinchium angustifolium P. Mill. – 10, 11, 12, 13, 14.
Smallanthus uvedalia (L.) Mackenzie ex Small – 8, 13. *Polymania*
Smilax bona-nox L. – 2, 3, 4, 5, 7, 8, 9, 10, 12, 13.
Smilax glauca Walt. – 1, 2, 3, 4, 5, 7, 8, 9, 10, 11, 13, 14.
Smilax herbacea L. – 1, 2.
Smilax hugeri (Small) J. B. S. Norton ex Pennell – 8.
Smilax pulverulenta Michx. – 1.
Smilax rotundifolia L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.

Smilax tamnoides L. – 1, 8.
Solidago arguta Aiton – 5.
Solidago caesia L. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Solidago curtisii Torr. & A. Gray – 4, 11, 12.
Solidago erecta Banks ex Pursh – 1, 2, 5, 10, 14.
Solidago flexicaulis L. – 1, 2, 3, 7, 8, 10, 11, 14.
Solidago gigantea Aiton – 7, 10.
Solidago juncea Aiton – 1, 3, 5, 10, 11.
Solidago nemoralis Aiton – 3.
Solidago rugosa P. Mill. ssp. *aspera* (Aiton) Cronquist – 5.
Solidago speciosa Nutt. – 2, 5, 14.
Solidago sphacelata Raf. – 1, 3, 8.
Solidago ulmifolia Muhl. ex Willd. – 1.
Sorghastrum elliottii (C. Mohr) Nash – 11.
Sphenopholis intermedia (Rydb.) Rydb. – 1.
Sphenopholis nitida (Biehler) Scribn. – 7, 8, 11, 12, 13, 14.
Sphenopbolis obtusata (Michx.) Scribn. – 9, 11, 13, 14.
Spigelia marilandica L. – 1, 2, 3, 6, 9.
Spiranthes tuberosa Raf. – 1.
Stachys tenuifolia Willd. var. *latidens* (Small ex Britton) Nelson – 1.
Staphylea trifolia L. – 9.
**Stellaria media* (L.) Vill. – 1, 6, 8.
Stellaria pubera Michx. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 14.
Symphoricarpos orbiculatus Moench – 6.
Symphyotrichum cordifolium (L.) Nesom – 13, 14. *Aster*
Symphyotrichum laeve (L.) A. & D. Love var. *concinnum* (Willd.) Nesom – 9.
Symphyotrichum lateriflorum (L.) A. & D. Love – 1, 3, 4, 5, 6, 7, 9, 10, 11, 12, 13, 14.
Symphyotrichum lowrieanum (Porter) Nesom – 3.
Symphyotrichum ontarionis (Wieg.) Nesom – 1, 6, 10.
Symphyotrichum oolentangiense (Riddell) Nesom – 6.
Symphyotrichum shortii (Lindl.) Nesom – 6.
Symphyotrichum undulatum (L.) Nesom – 6, 11, 12.
Symphyotrichum urophyllum (Lindl.) Nesom – 1, 2, 3, 5, 6, 7, 9, 10, 11, 12, 13, 14.
Teucrium canadense L. – 3.
Thalictrum dioicum L. – 10.
Thalictrum pubescens Pursh – 5.
Thalictrum revolutum DC. – 4, 5.
Thalictrum thalictroides (L.) Eames & Bolvin. – 1, 2, 3, 4, 5, 6, 8, 9, 10, 12, 13, 14. *Anemonella*
Thaspium barbinode (Michx.) Nutt. – 1, 2, 3, 8, 9, 11, 13.
Thaspium trifoliatum (L.) A. Gray – 12.
Thelypteris novaboracensis (L.) Nieuwl. – 6, 7.
Tiarella cordifolia L. – 1, 2, 3, 4, 5, 6, 7, 9, 10, 11, 12, 13, 14.
Tilia americana L. var. *heterophylla* (Vent.) Loud. – 1, 2, 3, 6, 9, 11, 14.
Tipularia discolor (Pursh) Nutt. – 4, 5, 7, 8, 9, 10, 11, 12, 13, 14.
Toxicodendron radicans (L.) Kuntze – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Trachelospermum difforme (Walt.) A. Gray – 11.
Tradescantia subaspera Ker-Gawl. – 3, 6, 14.
Trillium catesbaei Ell. – 6.
Trillium luteum (Muhl.) Harbison – 1, 2, 3, 4, 5, 6, 8, 9, 11, 12, 13, 14.
Triosteum angustifolium L. – 2.
Tsuga canadensis (L.) Carr. – 6, 7, 14.
Ulmus alata Michx. – 1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Ulmus americana L. – 3, 5, 6, 8, 9, 10, 11, 12, 13, 14.
Ulmus rubra Muhl. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Uvularia grandiflora Sm. – 2, 12.
Uvularia perfoliata L. – 2, 3, 4, 5, 6, 8, 9, 12.

Uvularia sessilifolia L. – 1, 4, 6, 10.
Vaccinium arboreum Marsh. – 1, 5, 10.
Vaccinium corymbosum L. – 5, 6, 7, 10.
Vaccinium pallidum Aiton – 1, 2, 5, 10.
Vaccinium stamineum L. – 5, 7, 10.
Valerianella radiata (L.) Dufr. – 1, 7, 11, 12.
Verbena urticaefolia L. – 4, 5, 6, 12, 13.
Verbesina alternifolia (L.) Bitton and Kearney – 2, 3, 4, 6, 9, 12, 14.
Verbesina occidentalis (L.) Walt. – 3, 4, 5, 7, 8, 13.
Verbesina virginica L. – 1, 2, 3, 6, 11, 14.
Vernonia gigantea (Walt.) Branner & Coville – 13.
**Veronica hederifolia* L. – 6.
**Veronica persica* Poir. – 3.
Viburnum acerifolium L. – 5, 7, 11, 13, 14.
Viburnum prunifolium L. – 4.
Viburnum rufidulum Raf. – 1, 2, 3, 4, 5, 6, 9, 10, 11, 12, 13.
Vicia caroliniana Walt. 1, 9, 11.
**Vinca minor* L. – 1, 3, 7, 10.
Viola blanda Willd. – 2, 7.
Viola canadensis L. – 2.
Viola cucullata Aiton – 2, 4, 12, 13.
Viola palmata L. – 2, 11, 12, 13.
Viola pedata L. – 12.
Viola pubescens Aiton – 5, 13.
Viola sagittata Aiton – 2.
Viola sororia Willd. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 13, 14.
Viola striata Aiton – 2, 3, 4.
Vitis rotundifolia Michx. – 1, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Vitis L. spp. – 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 11, 12, 13, 14.
Woodsia obtusa (Spreng.) Torr. – 6, 9, 11.
Xanthorhiza simplicissima Marsh. – 5, 9.

IMPACTS OF FARMING SYSTEMS ON SOIL CHARACTERISTICS

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ABSTRACT. Monitoring the changes of soil characteristics over time is important to better manage our farm, as well as to protect our natural resources. Modification of farming systems from conventional tillage to no till and organic based farming practices may change soil quality indicators. Soil characteristics such as soil organic matter (SOM), water holding capacity (WHC), bulk density (BD), total porosity, and aggregate stability (AS) are considered vital soil quality indicators. This study was designed to determine the effects of different farming practices in western Kentucky on selected soil physical characteristics. Soil samples were collected from the Murray State University farm in Calloway County, Kentucky. In August 2010, surface soil (0-7.5 cm and 7.5-15 cm) were taken collected from five fields: (1) sod as a control field, (2) 3 years of organic farm (OF3), 5 years of organic farm (OF5), (4) 15 years of no-tillage systems (NT), and 15 years of conventional tillage (CT) systems. The results show that organic farming and no tillage practices resulted in improving SOM, BD, porosity and AS. The highest values of these properties were found in 5-yr organic farming and the lowest were in 15-yr conventional tillage systems at the depth of 0 to 15 cm. Soil compaction, as indicated by bulk density, reduced up to 15% and the ability of soil to hold water increased about 22% after 5 years of organic farming when comparing to 15-yr of conventional tillage farming practices. Better soil quality under organic farming and no till practices indicates that continued organic matter input can gradually improve soil properties and regenerate degraded lands.

RED AND WHITE MULBERRIES (*MORUS RUBRA* AND *M. ALBA*: MORACEAE) AND THEIR INTERSPECIFIC HYBRIDS: ARE WE LOSING OUR NATIVE SPECIES?

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ABSTRACT. Red Mulberry (*Morus rubra*) is the only mulberry native to eastern North America. White Mulberry (*Morus alba*) is naturalized from Asia and is widespread throughout the United States as a result of escapes from cultivation, both as a street planting and from an unsuccessful attempt in the 1830s to establish a silk industry in this country (leaves used to feed larvae of the silkworm, *Bombyx mori*). *Morus rubra* and *M. alba* often have leaves of similar size and shape, and fruits are usually the same color. For over a century, plant keys and other literature have distinguished the two species based on leaf pubescence. According to botanical publications, leaves of *M. alba* are glossy above with very restricted or absent pubescence below, whereas *M. rubra* is generally scabrous above and pubescent below. The two species readily hybridize, and the fairly common trait of intermediate pubescence has been the presumed result. Results of this study demonstrate that leaf pubescence is not a diagnostic trait for separating these two species. Molecular data indicate that many individuals with pubescent leaves are in fact, *M. alba*, and not *M. rubra*. Further, hybrids are often cryptic when judged on pubescence. The frequency of hybrids formed between the native *M rubra* and the invasive *M. alba* is swamping out the native species in some areas. Hybrids and pure *M. alba* have been located in relatively undisturbed habitats, misidentified by current morphological criteria, but confirmed by molecular data. This study has identified a difference in leaf vein patterning that appears to be much more credible than pubescence for distinguishing the two species, plus their hybrids. Reliable identifications based on morphology are imperative for field botanists and conservation personnel, particularly where *M. rubra* is an imperiled species.

THE EXPANSION OF *MACROTHELYPTERIS TORRESIANA* (THELYPTERIDACEAE), AN INVASIVE FERN IN THE SOUTHEASTERN UNITED STATES

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ABSTRACT. *Macrothelypteris torresiana* (False Maiden Fern) is an invasive exotic Asian species first collected in the United States from Seminole County, Florida, in 1904. Since then it has been documented from much of the lower southeastern United States and is now found throughout most of the region between South Carolina, Florida, and eastern Texas. Recently, *M. torresiana* has been expanding its range northward into the interior of the United States with reports from southern Tennessee, southwestern Virginia, and southern Illinois. Here, we report the first occurrence of *M. torresiana* from Kentucky and discuss the spread of this species in the U.S. since 1970. A discussion of the ecological conditions and associated flora of the Todd County, Kentucky, population is also presented.

THE STATUS OF *ELAEAGNUS MULTIFLORA* (ELAEAGNACEAE), A POTENTIALLY INVASIVE ASIATIC SHRUB IN TENNESSEE

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ABSTRACT. *Elaeagnus multiflora,* a native shrub of China and Japan, is cultivated in the eastern United States for its edible fruits and ability to tolerate poor conditions. Currently, this species is introduced at a number of scattered localities in eastern U.S., particularly in Midwestern and Northeastern states. In the Southeast, it has been reported from Alabama, Mississippi, North Carolina, Virginia, and West Virginia. Examination of herbarium specimens revealed that *E. multiflora* was first collected in Tennessee in 1983 from woods at Fort Donelson National Battlefield in Stewart County. According to recent floristic accounts, this species is considered a watch list taxon in Tennessee and is not currently considered part of the naturalized flora. The objectives of this study were to 1) assess the current distribution of *E. multiflora* in Tennessee, 2) determine if the species is actively naturalizing at known populations, and 3) characterize the ecology of each population. Preliminary results indicate that *E. multiflora* occurs in Lawrence, Maury, and Stewart counties in the Western Highland Rim region of middle Tennessee. The population in Stewart County consists of numerous subpopulations. The other county occurrences are located 127 km to the southeast, each consisting of a single population located 0.5 km apart. This species inhabits upland, cherty or gravelly, acid soils of mixed deciduous forests. It is capable of establishing in edge habitats and in mature forests; this coupled with the likelihood of dispersal by birds suggests that *E. multiflora* has the potential to become a serious pest.

THE VASCULAR FLORA OF THE CLARKS RIVER NATIONAL WILDLIFE REFUGE, MARSHALL, MCCRACKEN AND GRAVES COUNTIES, KENTUCKY

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ABSTRACT. Kentucky is one of only seven states in the U.S. to have lost more than 80% of its wetlands. Most of the 1,266,000 acres of wetlands lost occurred in western Kentucky, a region formerly covered by extensive swamps, bottomland hardwood forests, marshes, and wet prairie. The Clarks River National Wildlife Refuge is an 8,265-acre refuge in Graves, Marshall, and McCracken counties Kentucky established to protect remaining wetland habitat. The purpose of this study is to conduct an inventory of the vascular flora of the refuge and qualitatively characterize plant communities to help increase the knowledge of its botanical resources. Collections have been pressed in triplicate and prepared for deposition in the Austin Peay State University herbarium (APSC) as well as the herbaria at the Missouri Botanical Garden (MO) and the University of Tennessee (TENN). Over 600 specimens have been vouchered during approximately 50 collection trips representing 92 families, 216 genera, and 478 species and intraspecific taxa. Among the species collected were 379 (82.4%) native species and 81 (17.6%) introduced species including 1 state record and 159 county records. The collection, to date, is comprised of 108 woody species, 243 herbaceous species, 102 graminoids, and 7 pteridophytes. These data represent collection efforts from May 2009 to November 2010. Additional work will be conducted through 2011 with focuses on consummation of common taxa, assessment of rare plant populations, and community type characterization and mapping.

A COMPARATIVE STUDY ON THE GENETIC DIVERSITY OF AN INVASIVE PLANT, *LONICERA JAPONICA* IN ITS NATIVE AND INTRODUCED RANGES

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ABSTRACT. It is critical to understand the role of species genetic diversity in the process of plant invasion. In this study, inter-simple sequence repeat (ISSR) polymorphism markers were used to determine and compare genetic diversity of Japanese honeysuckle (*Lonicera japonica*) in both native and introduced ranges. By studying plant samples collected from 7 populations with a total of 37 individuals at the molecular level, our results indicate significant genetic variability within and among Japanese honeysuckle populations from both ranges. Genetic diversity of the native population is higher than all populations from the introduced ranges. A lower level of genetic diversity found in populations sampled from the introduced range suggests that lost genetic diversity may largely be due to founder effects and bottleneck events that occurred during the invasion process. These losses however, might not preclude the rapid adaptive evolution of life history traits that enhance invasion. Our study also indicates that even with reduced genetic variation in the introduced range, a substantial level of genetic diversity exists in the invading populations. Therefore, it is very likely that a certain level of genetic variability is required to permit an adaptive response of the introduced species to the new selective pressure after introduction. We consider that genetic diversity could be a prerequisite of invasion success for some invaders such as Japanese honeysuckle.

TALL FESCUE MANAGEMENT IMPACTS ON SELECTED SOIL PROPERTIES

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ABSTRACT. Tall fescue forage systems have been commonly utilized in the United States which could change the magnitude of soil properties. Differences in soil physical properties, such as water holding capacity, macroporosity, bulk density, water content at field capacity, porosity and bulk density indicate changes in soil quality. The objective of this study was to characterize the effects of tall fescue management systems on selected soil physical properties. Soil samples were collected from three tall fescue fields (Tall Fescue Endophyte +, Tall Fescue Endophyte – and Tall Fescue-MaxQ) in Calloway Co. Kentucky. Overall, soil surface properties of the three tall fescue systems were relatively similar, indicating that the magnitude of biomass inputs from grasses were too low to change the soil property levels substantially or their stand age was relatively considered young (less than four years). However, the baseline data generated from this study can serve as starting points for assessing and monitoring the impacts of various tall fescue types on soil characteristic.

CONTRIBUTED PAPERS

SESSION II: AQUATIC BIOLOGY

Saturday, March 26, 2011

Moderated by:

Steven W. Hamilton

Center of Excellence for Field Biology

Austin Peay State University

COLONIZATION, TRENDS AND STATUS OF TWO NON-NATIVE SPECIES IN KENTUCKY LAKE, *PLECTOMERUS DOMBEYANUS* AND *DAPHNIA LUMHOLTZI*

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Reservoirs are inherently dynamic systems with ecological communities that change throughout their life cycles as habitats and ecosystems evolve. When reservoirs are inundated, ecosystems are converted from riverine to lake-like, and massive changes take place in the previously existing aquatic habitats. Thus, in a sense, all the species in reservoirs are "invaders" into these new “lakes”. Kentucky Lake is one of the largest reservoirs in the United States, and its ecological communities have been changing since it was constructed in 1944. Kentucky Lake has a history of species invading from other regions with some being successful and others not. Zebra mussels (*Driessena polymorpha*), for example, have invaded but with limited success and have not become established to the extent they have elsewhere in the country (Reed 2002). Other species have colonized and become integrated into the ecological community of Kentucky Lake, e.g., the Asian clam *Corbicula fluminea*. To better understand the colonization, spread, and long-term dynamics of invasive species in Kentucky Lake, we compare two very different species that have successfully colonized Kentucky Lake and many other Midwestern reservoirs: the native southeastern freshwater mussel, the bankclimber *Plectomerus dombeyanus* (Figure 1 A), and the exotic Asian zooplankter, *Daphnia lumholtzi* (Fig. 1B).

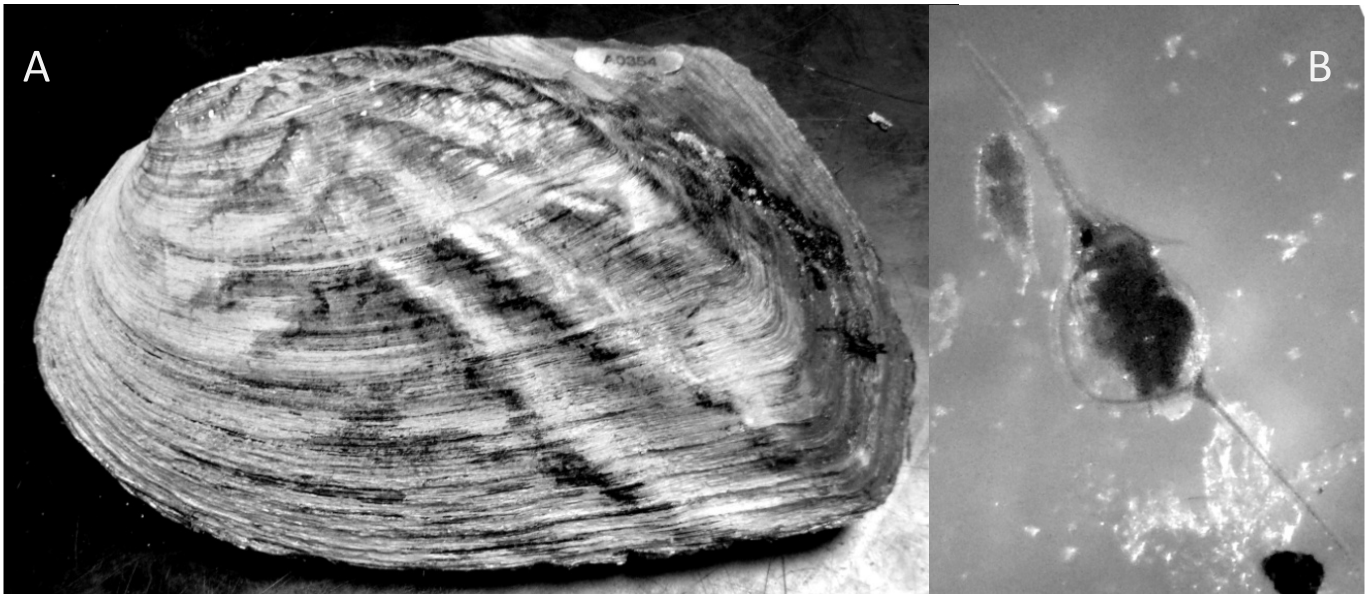

Figure 1. (A) *Plectomerus dombeyanus*, (B) *Daphnia lumholtzi*; photos by Laura Francisco.

Plectomerus dombeyanus is a southern species that appears to have worked its way up the Mississippi River invading the Tennessee River and Kentucky Lake in the early 1980s. It was first reported in Kentucky Lake in 1981 (Pharris et al. 1984). Much of its colonization in Kentucky Lake took place in the 1990’s, first along the main body of the lake and eventually into the embayments by 2002. It is now the dominant mussel at many, if not most, areas in the lake (Levine, pers. obs.). Because it is long lived, its interactions with other 38 mussel species (Hubbs 2009), if at all, and other effects on the reservoir environment are difficult to ascertain. An indirect assessment of population dynamics was determined by an examination of the rings that are formed approximately annually (Downing et al. 1992) and by length-frequency relationships (Payne and Miller 2000). Overall, *P. dombeyanus* exhibits a length-frequency structure that reflects low recruitment over time (fewer and fewer individuals in smaller size classes) and an ageing cohort (Figure 3 A). Population ageing is evident from a shifting mode of the length-frequency distribution. At this point, adult *P. dombeyanus* remain common, but what is controlling the population structure is unknown. Will the species eventually crash? We do not have enough information at this time to know if other common freshwater mussel species in the lake are following a similar pattern.

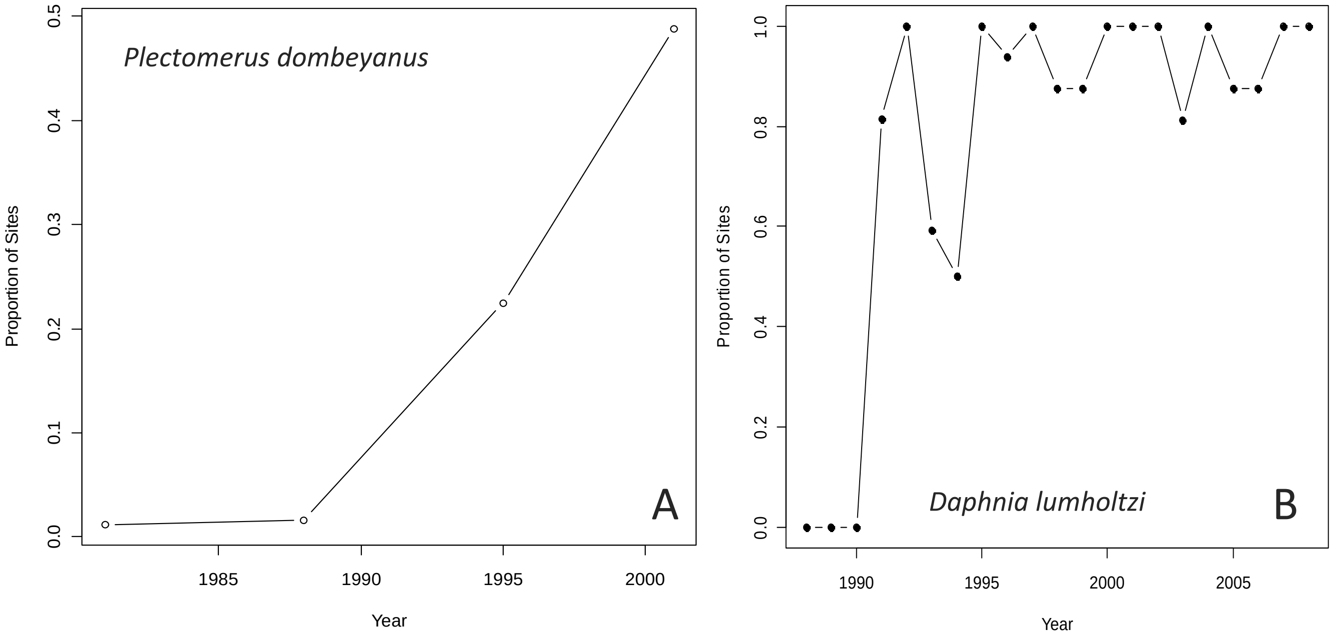

Figure 2. (A) *P. dombeyanus* proportion of sites occupied in surveys conducted in 1981, 1988, 1995 and 2001 (B) *D. lumholtzi* proportion of sites occupied from on-going bimonthly monitoring in the Kentucky Lake Monitoring Program.

Daphnia lumholtzi was first found in the Kentucky Lake in 1991 and by 1995 had established itself as one of the most common cladocerans (Fig. 2B) (Yurista et al. 2001). This is an exotic species of zooplankton that most likely originated from a southeastern Asian population, but no one really knows how it got here. It appeared in several Midwestern reservoirs at essentially the same time (Havel et al. 2005). Its interactions with other zooplankton species and effects on the Kentucky Lake environment have been much easier to determine because the population has been closely monitored since its arrival (White et al. 2007). Further its short life cycle allowed more generations to be analyzed over time. *D. lumholtzi* exhibits distinct annual peak densities late in summer (Fig. 3B) in contrast with the earlier summer peaks exhibited by most native cladocerans. Yurista et al. (2004) observed that once it became established, its densities were considerably higher than now, and densities have stabilized over the past 10 years. They noted that there appeared to be an initial competition between *D. lumholtzi* and the native cladoceran *Diaphanosoma birgei* whereby *D. birgei* annual peaks were and continue to be pushed earlier in the summer. The changes in timing of the annual peaks have been interpreted as evidence for complementary dynamics with the potential for reducing negative interactions between these two species (Havel and Graham 2006). *D. lumholtzi* can now be considered a “regular” and stable component of the zooplankton community. We are beginning to see a shift in the timing of peak abundance of all of the major zooplankton species, which is a focus of ongoing research on Kentucky Lake. The *D. lumholtzi* peak abundance timing is shifting in concert with other Kentucky Lake zooplankton species. The reasons for the changes are not yet apparent. Data from the ongoing long-term Kentucky Lake Monitoring Program (White et al. 2007) may provide some of the answers.

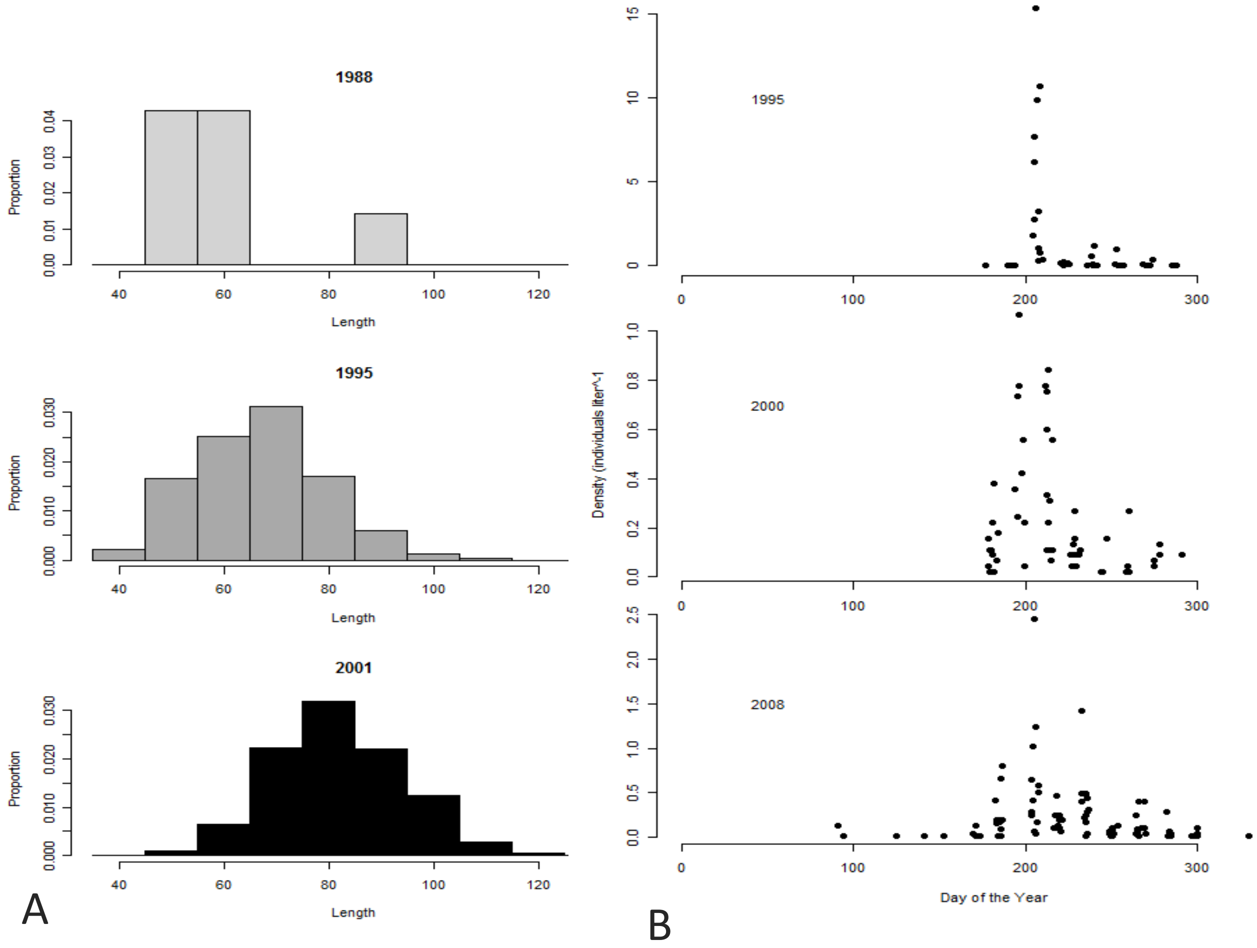

Figure 3. (A) *P. dombeyanus* long-term dynamics, shown by length frequency histograms over the course of the initial colonization from surveys conducted in 1988, 1995 and 2001 (B) *D. lumholtzi* annual pattern of peak densities from three years, a small amount of random variation (jittering) was added to the day of the year to help visualize points from the same date and density.

To survive in a reservoir setting, invasive species must co-exist with a plethora of other potentially adaptable species, including those from within the native, regional species pool and those from the outside. In these two cases, one species (*D. lumholtzi*) has been shown to successfully interact with native taxa, while the other (*P. dombeyanus*) has no known interactions. Thus they present two possible mechanisms for being a successful invader into a reservoir environment. Other Kentucky Lake invasive species where less is known include *Eurytemora affinis*, a copepod of marine origin (Williamson and White 2007) and a number of aquatic macrophytes. Given the potential for invasions, it remains surprising that there are comparatively few successful invaders, but those that are successful are often very successful. Although it has been argued to the contrary (Havel et al. 2005), invading a reservoir may be more difficult than invading a natural system because of the ever-changing physical dynamics of most reservoirs. Invasions of species from outside the regional species pool can be facilitated by the complex nature of reservoir ecosystems acting as conduits (Havel et al. 2005). The Tennessee and Cumberland river systems may function as such conduits promoting the spread of adaptable species including *Plectomerus*. This is certainly true for Asian carp (silver and bighead) that are recent very successful invaders. Understanding how different invasive species become integrated into and interact with the ecosystem helps us to understand community dynamics in human-created environments.

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AN ASSESSMENT OF THE INFLUENCE OF *PODOSTEMUM CERATOPHYLLUM* ON NUTRIENT AND SESTONIC ALGAL LEVELS IN THE GREEN RIVER

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In-stream primary productivity can be strongly influenced by nutrient availability and limitation (Dodds *et al*. 2002). Nutrient availability is influenced by bedrock geology, biotic activity, and groundwater recharge (Fisher Wold and Hershey 1999, Penick *et al.* 2012). The most common limiting macronutrients in freshwater ecosystems are nitrogen (N) and phosphorus (P) (Dodds *et al*. 2002).

Sestonic algae are common primary producers in riverine systems, with biomass levels varying seasonally (Atkinson *et al*. 2008) and in response to nutrient availability (Figueroa-Nieves *et al*. 2006) and filamentous algal growth (Fileto *et al*. 2010). The macrophyte *Podostemum ceratophyllum* Michx. can be common in riverine systems in the southeastern U.S. and provides important benthic habitat for macroinvertebrates (Hutchens *et al.* 2004), yet contributes little to energy budgets (Hill and Webster 1984).

In September 2010 we experimentally tested the influence of *P. ceratophyllum* on nutrient and sestonic algae levels from two sites in the Green River, Kentucky. We predicted that *P. ceratophyllum* would outcompete sestonic algae for nutrients. Sites were distinct according to: moderate *P. ceratophyllum* standing crop plus low sestonic algal levels (upstream: 37.1527, -86.4668) vs. no *P. ceratophyllum* plus high sestonic algal levels (downstream: 37.2434, -86.0139). Three treatments were established in 12 L mesocosms: sterilized cobbles (= control), cobbles with natural periphyton growth, and cobbles with *P. ceratophyllum*. Treatments were replicated (n = 4) and experiments were run for 6 d. This experiment was conducted twice, each time using one of two sources of water: downstream water with higher measured nutrient concentrations and upstream water with lower measured nutrient levels. Ammonia, nitrate, and soluble reactive phosphorous (SRP) levels were quantified using spectrophotometry at the beginning of each experiment and in 2-d intervals. Sestonic algal levels were quantified as chlorophyll-α concentrations using spectrofluorometry at 0 and 6 days.

There were no significant differences in nutrient levels between treatments. Between-treatment nutrient levels were similar for SRP and nitrate while ammonia levels increased only slightly in both water source experiments. There was also a lack of between-treatment differences in sestonic chlorophyll-α levels.

The presence of *P. ceratophyllum* did not lead to amplified nutrient loss, suggesting that this species does not intensely compete for nutrients with other riverine primary producers. Notably, we conducted our experiments in fall when *P. ceratophyllum* productivity would be expected to decline (Hill and Webster 1984), potentially leading to reduced nutrient uptake.

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THE DISTRIBUTION, ABUNDANCE, AND HABITAT COLONIZATION OF *HYDRILLA VERTICILLATA* (HYDROCHARITACEAE), IN A HIGH-GRADIENT RIVERINE SYSTEM

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ABSTRACT. The Emory River Watershed (ERW) is a high-gradient river system located within the Cumberland Plateau, including cool, clear streams that are part of the National Wild and Scenic River System. Currently, the ERW is infested by the invasive submersed macrophyte, *Hydrilla verticillata*, representing a unique situation for the species in North America. In 2010, a study was initiated for the U.S. National Park Service to determine the origin and extent of the hydrilla infestation. The distribution, relative abundance, and occupied habitat types of the species within the ERW was also established. Fieldwork was conducted from July-September 2010. *Hydrilla* was discovered in a private lake feeding into Daddy's Creek in Cumberland County, which was subsequently confirmed to be the source of the infestation. A systematic survey of 29.5 miles of the ERW was conducted from Devil's Breakfast Table on Daddy's Creek to Camp Austin on the Emory River revealing that *Hydrilla* covers 26% of the ERW and prefers slow-moving pool and run habitats. In addition, *Hydrilla* has expanded a distance of more than 50 miles downstream to Harriman in Roane County, and it is suspected that is has colonized beyond the ERW boundaries. Future research should be conducted to determine potential impacts that hydrilla may have on the inhabiting flora and fauna within the ERW.

EFFECTS OF ATRAZINE EXPOSURE ON IMMUNE FUNCTION OF A DRAGONFLY LARVA, *PLATHEMIS LYDIA*

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ABSTRACT. Agricultural runoff containing herbicide is known to have adverse effects on freshwater organisms, affecting large groups across trophic levels. Aquatic insects are particularly susceptible, and herbicide runoff has the potential to affect immunity in this group. We examine the effect of atrazine, a commonly used herbicide in the United States, on dragonfly larvae (*Plathemis lydia*) during a long-term exposure at ecologically relevant concentrations. Larvae were exposed to concentrations of 0, 1, 5 and 10 ppb atrazine for three or six weeks. Hemolymph phenyloxidase (PO) activity, cuticular PO, gut PO, number of hemocytes, encapsulation ability and PO of encapsulated individuals were measured at the end of each trial period as indicators of immune strength. Concentration had a significant effect on hemocyte count (df=3, P=0.04) after controlling for the effect of larva size. There was a significant interaction (df=3 P=0.02) between time and concentration for hemolymph PO measurements and concentration had a significant effect at six weeks only (df=3 P=0.03). Concentration had a significant effect on cuticular PO (df=3, P=0.05) and a marginally significant effect on gut PO (df=3, P=0.06) at six weeks only. Therefore, atrazine affects both hemocyte numbers and phenyloxidase activity in *P. lydia*; the exact impact of the changes is unclear. However, the changed immune function demonstrated in this study is likely to increase susceptibility to pathogens or decrease available energy for growth and metamorphosis.

STREAM OR DITCH? NEWLY DEVELOPED STATE SOP FOR DETERMINING HYDROLOGIC STATUS

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ABSTRACT. On June 1, 2009, the General Assembly passed Public Chapter 464, which codified into law a new statutory definition of “wet weather conveyance”, directed TDEC to develop new rules and guidance concerning hydrologic determinations (HD), and develop a program to certify outside individuals as "qualified hydrologic professionals” (QHP), whose stream calls would enjoy a presumption of correctness. In response, TDEC proposed, received public comment on, revised, and promulgated new rules and a guidance document that were approved by the State’s Water Quality Control Board in October 2010. Together these provided a detailed SOP on how to consistently and accurately determine the jurisdictional status of a watercourse, utilizing both Primary Field Indicators (considered individually definitive), and Secondary Indicators that require scoring in aggregate to make a determination. The new rules also provided an framework for the QHP program. The SOP was to a large degree based upon concepts and methodologies originally developed and revised by the North Carolina Division of Water Quality since 1997, and the procedures are based on the relationships between the underlying disciplines of biology, geology, geomorphology, meteorology, and hydrology that are involved in creating, maintaining, and identifying hydrologic features. A list of benthic macroinvertebrate taxa that will be considered as Primary Indicators was also developed. To qualify, a taxon must be an “obligate lotic aquatic organism whose life cycle includes an aquatic phase of at least two months”, per the new statutory definition within P.C. 464. Also in response to P.C. 464, TDEC has developed a 3-day HD training course, which will both provide consistent instruction for TDEC staff (or anyone else who desires it), as well as be part of the overall QHP certification process. Two pilot courses were held at Montgomery Bell State Park in November 2010 and March 2011, primarily for TDEC staff. Future courses open to the various other interested parties have been planned, and are being coordinated through the UT Water Resources Research Center, who in conjunction with TDEC, will also facilitate the QHP certification process.

BISPHENOL-A CONCENTRATIONS IN NATURAL WATERS AND BOTTLED DRINKING WATERS: A POSSIBLE SOURCE OF HUMAN EXPOSURE

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ABSTRACT. Bisphenol A is a man-made chemical used in industry and consumer products all over the world such as plastic beverage bottles, compact disks, canned food linings and many others. Human exposure to this chemical comes from its leaching from plastics into food and liquids. Exposure to BPA is reported to cause major complications in the endocrine systems of many animal species and human populations. Knowledge on contamination levels of BPA is essential in order to prevent future contamination and protect wildlife and humans from negative health effects. Very limited information is available on the BPA levels in various brands of bottled drinking water, fountain water, river water and lake water. In this study, BPA concentrations were measured in several brands of bottled waters and in water samples collected from Bee Creek, Red Duck Creek, Clarks River and Kentucky Lake. Enzyme-linked immunosorbent assay (ELISA) method was used to determine BPA levels in these samples. Correlations between BPA and E. Coli are anticipated based on statistical analysis of data from Red Duck Creek. Results revealed that detectable levels of BPA were found in all samples analyzed. The results were compared with published reports and evaluated the potential exposure of BPA to human populations in this region.

EVALUATING MACROINVERTEBRATE DIVERSITY IN POND COMMUNITIES: A COMPARISON OF TWO SAMPLING TECHNIQUES

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ABSTRACT. Milan Army Ammunition Plant (MLAAP), located in Gibson and Carroll counties, Tennessee, is a munitions production and storage facility comprising 90.48 km^2 of upland hardwood forest, interspersed with agricultural fields. In 1987, MLAAP was placed on the Environmental Protection Agency's National Priority List for groundwater contamination. Due to this environmental status, MLAAP has established an Integrated Natural Resources Management Plan, coupling organismal research and land use practices to develop long-term sustainability of natural resources. Macroinvertebrates, often used in water quality monitoring, are among the taxa being studied. In June, 2009, a research project was begun to compare the efficiency of two sampling techniques in inventorying macroinvertebrate diversity within pond communities at MLAAP. Funnel-trap and dip net sampling methods were employed in 10 ponds. Four funnel-traps were set in each pond for two consecutive 48-hour periods during June 18-22 and December 8-12, 2009 and April 18-22, 2010. Dip net samples were collected on June 29 and November 11, 2009 and April 2, 2010 with two collectors sampling simultaneously for 30 minutes in each pond. A total of 10,082 individuals comprising 146 unique taxa were identified. Statistical analysis comparing sampling methods showed significant differences in taxa richness within cattle ponds. The differences between sampling methods for Shannon-Weaver values were significant among all ponds. A significant difference between sampling methods for Shannon-Weaver Index values was also found within non-cattle ponds as well as for the summer sampling season. Jaccard's Similarity Coefficient values were generally low (mean = 0.2896, range = 0.125 – 0.576), signifying both methods collected very different sets of taxa. Differences between Jaccard's Similarity Coefficient values were significant among cattle and non-cattle ponds and for the winter sampling season. Evaluation of sampling methods regarding addition of new taxa indicated the dip net method more effectively added new taxa in the orders Coleoptera, Odonata and Hemiptera, while the funnel-traps were more successful adding to the order Diptera. These results can be attributed to the mobility of the dip nets, versus the funnel-traps, which are dependent on invertebrate movement. Taxa accumulation curves indicate a combination of sampling methods would be the best strategy for assessing the biodiversity of pond habitats. Due to sampling method constraints and habitat limitations to sampling effort, the choice of sampling technique should be based on habitat structure. Timed-effort sampling cannot be standardized if habitat complexity is not taken into consideration and passive sampling alone will not produce accurate community diversity data.

THE NOISY STREAM:
BIOACOUSTICS IN AQUATIC INVERTEBRATES

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ABSTRACT. Terrestrial insects such as grasshoppers, crickets, and cicadas commonly use audible calls to communicate within their own species over considerable distances. We know the adults of many aquatic invertebrates acoustically communicate above the water, and furthermore, that water is an amazing medium for transmitting sound. Yet, as of now, there has been no documentation of aquatic insects vocalizing underwater. What does it sound like to live in a stream and how could invertebrates utilize this medium for communication? Together we will explore the bioacoustic potential of these water-dwelling invertebrates and how these sounds, as well as the sounds created in a stream system, could aid in future biomonitoring of our waters. Developing a set of reference recordings of invertebrates and physical conditions in the stream for potential use in future hydroacoustics studies is the first step in this biomonitoring process.

EFFECTS OF THE INVASIVE SUBMERSED MACROPHYTE, *HYDRILLA VERTICILLATA*, ON THE COMMUNITY COMPOSITION OF AQUATIC MACROINVERTEBRATES INHABITING A HIGH-GRADIENT RIVERINE SYSTEM

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ABSTRACT. The Obed Wild and Scenic River (OWSR) is a national park located within the Emory River Watershed (ERW) of Tennessee. Between 2004 and 2009, *Hydrilla verticillata* was discovered in several localities of the ERW, which gave rise to a detailed study of its distribution and location within the Obed Wild and Scenic River (OWSR) for the U.S. National Park Service. Because hydrilla has established itself in a high-gradient ecosystem, a study to determine potential impacts of this invasive macrophyte on the aquatic macroinvertebrate communities was necessary. Sampling was conducted within hydrilla-infested and non-infested pools, in addition to corresponding downstream riffle habitats. Study sites included infested areas of Daddy's Creek and Obed River, and comparative non-infested portions of Clear Creek within the OWSR. Preliminary analysis of data from samples collected in late September 2010 has suggested hydrilla-dwelling taxa most prevalent in hydrilla-infested pools, specifically representatives from tribe Tanytarsini (Diptera) and genera *Oxyethira* and *Triaenodes* (Trichoptera). Riffle communities from Potter's Ford (non-hydrilla site) and Antioch Bridge (hydrilla site) have demonstrated no significant differences in taxa richness and Simpson's Index of Diversity ($p > 0.05$). Further identification and evaluation is currently being conducted to establish differences between remaining sites and habitats, in addition to possible influences hydrilla may have on this high-quality river system.

THE INFLUENCE OF THERMOKARST DISTURBANCE OF SEDIMENT DELIVERY AND MACROINVERTEBRATE COMMUNITY DYNAMICS IN ARCTIC HEADWATER STREAMS

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ABSTRACT. Recent research has highlighted increased permafrost degradation and thermokarst formation as a result of a warming arctic climate. Thermokarst failures influence organic matter and sediment delivery to adjacent streams, which may affect benthic macroinvertebrate communities. Our studies focused on two headwater streams on the North Slope of Alaska. Data collected in summer 2010 near Toolik Lake, AK revealed a near doubling of total sediment deposited in downstream areas impacted by thermokarst features relative to upstream reference reaches which were unaffected ($P<0.01$). Organic matter delivery significantly increased in impacted reaches ($P<0.01$), primarily the very fine fraction (<250um). However, very fines contributed only a small portion (~10-15%) of total sediment to impacted reaches. Benthic macroinvertebrates showed a variable response in abundance and biomass between study streams. Preliminary data shows no significant shift in species richness, biomass or abundance of benthic invertebrates between reference and impacted reaches. Functional feeding group composition was variable: scraper (Ephemeroptera: Baetidae) abundance was significantly lower in one impacted site ($P<0.05$), though results were not consistent between sites. Benthic organic matter showed no significant shift between reference and impacted reaches. Results from previous years' research indicate thermokarst impacts on benthic biota vary spatially and temporally, and intensity may shift as landscape recovery occurs. Patterns suggest a relatively high resiliency and quick recovery time of macroinvertebrates in response to these disturbances.

CONTRIBUTED PAPERS

SESSION III: ZOOLOGY

Saturday, March 26, 2011

Moderated by:

Andrew N. Barrass

Center of Excellence for Field Biology

Austin Peay State University

A COMPARISON BETWEEN AVIAN COMMUNITIES IN FESCUE FIELDS AND TALL GRASS PLANTINGS WITHIN THE GREEN RIVER CONSERVATION RESERVE ENHANCEMENT PROGRAM, KY

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ABSTRACT. A goal of the USDA Conservation Reserve Enhancement Program (CREP) tall grass plantings is to improve wildlife habitat. We conducted breeding bird surveys in seven CREP fields and seven fescue hayfields during 2006, and winter bird surveys and vegetation surveys in ten CREP fields and nine fescue hayfields in 2007/08. We used Mann Whitney U Tests to compare frequencies between hayfields and CREP tall grass plantings. Northern cardinal, American goldfinch, and the scrub guild were sighted more frequently in CREP fields than in fescue hayfields during the 2006 breeding survey. Swamp sparrow, song sparrow, and the open woodland guild were seen more frequently in CREP fields than in fescue hayfields during the winter 2008 survey. Total species richness was also higher in CREP fields in winter 2008. Song sparrows and species richness appear to be positively influenced by the density of tall stems.

INTRODUCTION

Grassland bird populations are in a decline (Johnson and Schwartz 1993, Delisle and Savidge 1997, Klute et al. 1997) that has been attributed in part to the conversion of natural grassland habitats into agricultural lands (Klute et al. 1997, Delisle and Savidge 1997, Patterson and Best 2003).

Avian habitat use is influenced by the vegetation structure of the habitat. Specifically, grassland bird abundance within a habitat is affected by vegetation structure (Klute et al. 1997, Patterson and Best 2003). CREP plantings provide vegetation structure that is very different from pastures. A number of studies conducted in the United States have demonstrated that CRP plantings enhance avian habitat compared to agricultural fields (Johnson and Schwartz 1993, Delisle and Savidge 1997, Klute et al. 1997, Patterson and Best 2003, Blank and Gill 2006).

In 2001 the USDA Natural Resources Conservation Service implemented the Conservation Reserve Enhancement Program (CREP) in the Upper Green River watershed of Kentucky. CREP is a program related to CRP (Kentucky.gov, 2006). A goal of Conservation Reserve Enhancement Program (CREP) tall grass plantings is to improve wildlife habitat.

METHODS

To evaluate the influence of tall grass plantings on birds, we conducted avian surveys to study the influence of CREP in the Upper Green River watershed of Kentucky on grassland and Neotropical songbird species richness. We conducted breeding bird surveys in seven CREP fields and seven fescue hayfields during 2006. Winter bird surveys and vegetation surveys were conducted in ten CREP fields and nine fescue hayfields in 2007/08.

We tallied bird species as present if they were observed perched in the field or feeding directly over the field. Birds at the field boundaries were not included in the analysis. Both the 2006 & 2007/08 surveys were conducted between sunrise and within one to three hours after noon. The fields were divided into transects approximately fifty meters wide. The number of transects varied depending on the area of the field. Bird surveys were conducted during the second walk down each transect. Each transect survey lasted three minutes.

We took vegetation measurements in each field to evaluate vegetation structure during the winter avian survey period. A transect was established down the middle length and width of each site. Four randomly located one meter square sampling quadrats were studied on each transect. We recorded the height of the tallest stem and the number of stems greater than 150 cm in height for each quadrat, and calculated the median of these measures for each site as STEM HEIGHT and STEM NUM respectively.

We sorted avian species into guilds based on Cornell Lab of Ornithology life history habitat classifications. Mann-Whitney U Tests were performed using Statistica 7 (*StatSoft® Software* 2004) to compare frequencies of occurrence of avian guilds (open woodland, town, grassland, scrub, and forest guilds), total species richness, and the frequencies of occurrence of five common species (Northern Cardinal, American Goldfinch, Common Yellowthroat, Swamp Sparrow, and Song Sparrow) between the two field types.

We used zero-inflated Poisson regression PROC NLMIXED (*Statistical Analysis with SAS/STAT® Software* 2010) to develop exploratory models relating species richness and the frequency of abundant birds to highest vegetation per m^2 and median stems > 150 cm/m^2. The zero-inflated Poisson regression was designed to deal with an overabundance of zeros located in the data.

RESULTS

Preliminary analyses showed that Northern Cardinal (*Cardinalis cardinalis*) ($p = 0.01$, z adjusted = 2.58), American Goldfinch (*Spinus tristis*) ($p < 0.01$, z adjusted = 2.69), and the scrub guild ($p = 0.02$, z adjusted = 2.30) were sighted more frequently in CREP fields than in fescue fields during the 2006 breeding bird surveys. Swamp Sparrow (*Melospiza georgiana*) ($p < 0.01$, z adjusted = -2.73), Song Sparrow (*Melospiza melodia*) ($p < 0.01$, z adjusted = -3.43), and the open woodland guild ($p < 0.01$, z adjusted = -2.91) were sighted more frequently in CREP fields than in fescue fields during the winter 2008 bird surveys. Total species richness ($p < 0.01$, z adjusted = -2.79) was also higher in CREP fields in winter 2008. No individual species or avian guilds were observed more frequently in fescue fields.

The exploratory zero-inflated Poisson regression model found that: 1) Swamp Sparrow presence was significantly and positive related to STEM HEIGHT ($t_{19} = +0.02349$, $p = 0.0395$), 2) Species richness was significantly and positively related to STEM HEIGHT ($t_{19} = +0.01924$, $p = 0.0043$), and 3) Species richness was significantly and positively related to STEM NUM ($t_{19} = +0.01039$, $p = 0.0081$).

Overall, the winter CREP vegetation was taller than the winter fescue vegetation. Winter CREP had a higher percentage of lodging and a lower percentage of bare ground.

DISCUSSION

During the 2006 breeding bird surveys Northern Cardinal, American Goldfinch, and the scrub guild were sighted more frequently in CREP fields than in fescue hayfields. Swamp Sparrow, Song Sparrow, and the open woodland guild were sighted more frequently in CREP fields than in fescue hayfields during the winter 2008 bird surveys. Total species richness was also higher in CREP fields in winter 2008. No individual species or avian guilds were observed more frequently in fescue fields. Our findings are consistent with those of Patterson and Best (1996), Best *et al.* (1997), and Blank and Gill (2006), who found that CREP and CRP fields during the breeding season have significantly higher abundance of birds, significantly higher species richness, and are significantly more important for individual bird species, particularly grassland and early-successional generalist species. For the winter season, previous studies (Best *et al.* 1998, Blank and Gill 2006) have found that CREP and CRP fields are not significantly more important than row crop fields for bird communities in either measurements of density or richness. In contrast, this study found CREP fields have significantly higher species richness than did fescue hayfields.

Overall, the winter CREP vegetation was taller than the winter fescue vegetation. Winter CREP had a higher percentage of lodging and a lower percentage of bare ground compared to the hayfields. When considering land management options, the differences between the species supported by CREP versus fescue hayfields should be considered. CREP fields tend to have greater vegetation structural heterogeneity, which indicates improved habitat for successional avian species and guilds compared to fescue hayfields. The fescue hayfields in this study supported open grassland birds in agreement with other studies (Klute *et al*. 1997) which have found that fescue hayfields typically provide better habitat for bird species dependent upon short grass habitats (Eastern Meadowlark and Grasshopper Sparrow). To better investigate the influence of CREP tall grass plantings on avian communities, a larger sample size is needed for both avian and vegetation surveys. The control of additional variables such as field size, CREP seed mixes, field edge difference, potential overlapping bird use of each field treatment, and field management will also help provide more accurate data, deeper context, and more powerful analyses.

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COMPETITION BETWEEN LARVAL SALAMANDERS IN NATURAL POPULATIONS: ASSESSING RELATIONSHIPS OF SIZE, DENSITY AND HABITAT USE WITH OBSERVATIONAL DATA

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ABSTRACT. Experimental manipulations are often used to determine the importance of factors that affect the outcome of competition. Observational approaches are utilized less often, but can provide equally valuable information on unmanipulated competing populations. Conservation measures could thus be greatly improved by an understanding of how unmanipulated communities function. Experimental tests of larval salamanders are frequent, but assessment of unmanipulated natural populations has been less common. Spotted (*Ambystoma maculatum*) and mole (*A. talpoideum*) salamanders have exhibited competitive behavior in experimental tests yet competition in natural ponds has not been examined. Populations of larval spotted and mole salamanders were sampled for larval size and density relationships in Land Between the Lakes National Recreation Area, Kentucky over a two-year period. Analysis revealed neither species' density nor size was reduced in comparing single-species ponds versus ponds where coexistence occurred. When syntopic, *A. maculatum* was significantly higher in density, whereas *A. talpoideum* tended to be larger in size, a trait often associated with competitive dominance. Larval size for *A. maculatum* varied with overall density in ponds containing conspecifics only versus heterospecifics, indicating *A. talpoideum* presence influenced their size when coexisting. AIC selection procedures supported models showing a positive correlation for larval size, indicating pond quality was similar for both species. Other supported models suggested intraspecific competition was more important for *A. maculatum*, whereas interspecific competition was more important for *A. talpoideum*, opposite of experimental research on these species. Using observational data to assess competition formed alternate conclusions to previous experimental results, showing the need for more observational approaches to be performed in conjunction with experimental tests. Habitat management decisions should incorporate observational data as it may better represent how natural communities interact compared to experimental approaches, and offer more insight into critical community components that could be priorities for conservation.

A COLD-BLOODED KILLER, PRESENCE OF RANAVIRUS IN AMPHIBIAN POPULATIONS IN WEST TENNESSEE, USA

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ABSTRACT. Ranaviruses have been linked to pervasive die-offs in amphibian populations throughout the Americas, Europe, Asia, and Australia. Studies have found ranaviruses throughout the Southeast United States; however, no research has identified this pathogen in West Tennessee. It has been shown that anthropogenic induced stress is a pivotal cause of amphibian susceptibility to disease and this could exacerbate ranavirus infected amphibian mortalities. Poor water quality from cattle-access (an anthropogenic stressor) has been suggested to be linked with ranavirus infection. This study describes the presence of Frog Virus 3, an aggressively pathogenic strain of ranavirus, at cattle-access and non-access ponds from select sites in West Tennessee. Frog tissues tested were shown to contain ranavirus DNA and to be prolific in the ponds examined. These data support the notion that Frog Virus 3 could be more widespread than previously hypothesized and point to the need for developing management and conservation techniques to reduce ranavirus-derived amphibian declines.

THE AMPHIBIANS AND REPTILES OF CHEATHAM WILDLIFE MANAGEMENT AREA, CHEATHAM COUNTY, TENNESSEE

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ABSTRACT. A year-long survey of the herpetofauna of Cheatham Wildlife Management Area (CWMA) was conducted from September 2009 through August 2010. CWMA is a 8,422-hectare tract of upland habitat located near Ashland City in Cheatham County, Tennessee. It is within the Western Highland Rim physiographic province and is mostly forested with wildlife openings and man-made ponds interspersed throughout. Reptiles and amphibians on CWMA were systematically sampled utilizing: 1) pitfall and box traps in conjunction with drift fences, 2) cover-board arrays, 3) cover-tin arrays, 4) stream transects, 5) hoop nets, 6) incidental capture, and 7) minnow traps. Sampling devices were checked once each month on a schedule that varied slightly depending on approval by CWMA's manager. Based on historical records, 28 reptilian and 29 amphibian species have been documented for Cheatham County. This survey documented 20 reptilian and 18 amphibian species as represented by 2,355 individual captures. All captured individuals were sexed and aged (if possible), weighed, measured, and released (except for voucher specimens) at point of capture. Voucher specimens were euthanized by approved means, fixed in a 10% formalin solution, and will be accessioned into the David H. Snyder Museum of Zoology at Austin Peay State University, Clarksville, Tennessee.

REASSESSMENT OF THE FISHES OF THE LITTLE RIVER SYSTEM, WESTERN KENTUCKY

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ABSTRACT. The Little River represents one of the most impaired watersheds in western Kentucky due to continued nonpoint source pollution associated with agriculture and urbanization. To evaluate historical changes in habitat, water, and fish community quality in the Little River, we sampled fishes and conducted habitat assessments during periods of high and low base flow in 2010 and compared findings to those of previous studies conducted from 2000 to 2003. Ten families, 23 genera and 33 species were represented in the June/July sampling, with the September/October sampling resulting in 14 families, 27 genera and 39 species. Intolerant species comprised 18.2% of species in the June/July sampling and 15.4% in the September/October sampling (compared to 5% and 21%, respectively in the previous study). Tolerant species comprised 45% of species in the June/July sampling and 38% in the September/October sampling (compared to 21% and 29%, respectively in the previous study). The highest re-occurrence of species were Bluntnose Minnow (14 of 16 sites), Scarlet Shiner (14 of 16 sites), and Striped Shiner (14 of 16 sites), which are all tolerant to non-point source pollution. Comparisons of current Kentucky Index of Biotic Integrity scores to scores from previous studies indicate the water quality and diversity of fish species in the Little River have decreased over the past ten years, with the majority of sites receiving a 'Fair' or 'Poor' rating. The rare species, *Etheostoma microlepidum*, was found at the same localities as in previous studies and at one site not previously captured. Although the range of this species in the watershed appears stable, habitat and water quality scores for each site where it occurs have declined and its status in the Little River should be closely monitored. The Little River continues to be an imperiled watershed that is in need of management and remediation. Future work will include examination of historical land-use changes in the system to highlight areas most in need of management.

FISH COMMUNITY RESPONSE TO PHRAGMITES REMOVAL AT CLEAR CREEK WILDLIFE MANAGEMENT AREA

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ABSTRACT. ***Phragmites australis*** **is an invasive aquatic plant found throughout most of the United States. This plant has been shown to alter wetland hydrology and is related to a decline in larval fish assemblages and egg development. Effective control and eradication of** ***Phragmites*** **is an option for biologists who wish to return affected wetlands to their pre-invasion state. Clear Creek Wildlife Management Area (WMA) is an 858 acre property located in Hopkins County, KY which is heavily impacted by the invasion of** ***Phragmites*****. The Kentucky Department of Fish and Wildlife Resources treated half of the WMA with herbicide in fall 2009 to restore the wetland by removing the** ***Phragmites*** **and allowing native flora to return. The goal of this project was to study the effects of this removal on the fish community. Following treatment, three locations (*****Phragmites*** **treated,** ***Phragmites*** **untreated, and non-*****Phragmites*****) were selected and sampled by seining and electroshocking. Each sample was scored and compared using the Kentucky Index of Biotic Integrity (KIBI), the Shannon's Biodiversity Index (SBI), and the Jaccard's Similarity analysis. The average KIBI and SBI values illustrated no significant difference among the three sampling locations. However, the treatment area had the highest single KIBI and SBI value during the study and highest ratio of species per individuals sampled. Additionally, the Jaccard's Similarity analysis indicated a divergence in the fish community composition at the treated location compared to the untreated location. These results highlight the importance of monitoring habitat restoration efforts to guide future management attempts.**

EFFECTS OF HERBICIDAL REMOVAL OF COMMON REED (*PHRAGMITES AUSTRALIS*) ON THE DIET OF LAKE CHUBSUCKER (*ERIMYZON SUCETTA*) IN CLEAR CREEK

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ABSTRACT. Research at Clear Creek Wildlife Management Area is being conducted for the investigation of gut contents in a small catostomid fish, the lake chubsucker (***Erimyzon sucetta*****), for the purpose of determining whether or not herbicidal treatment and removal of invasive common reed (*****Phragmites australis*****) is altering its diet. Using six sites at each of three different treatments controlling for presence of** ***Phragmites*** **and herbicidal application, fish were collected over the 2009 and 2010 field seasons. The gut contents of the fish are being identified, separated, and quantified for use in statistical analyses. Parameters used for analyses include frequency of occurrence, percent composition by dry weight, and percent composition by number. Though floral communities are expected to change from the herbicidal treatment of a** ***Phragmites*** **infested area, I hypothesize there will be no significant alterations in lake chubsucker diet due to the nature of selective foraging and the limited duration of this study, despite gradual alteration of prey communities.**

SYSTEMATICS OF THE SLENDER MADTOM, *NOTURUS EXILIS*, WITH DESCRIPTION OF A NEW SPECIES FROM ARKANSAS

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ABSTRACT. The Slender Madtom, *Noturus exilis,* is disjunctly distributed in the eastern and interior highlands with additional isolated populations in glaciated regions in Wisconsin, Illinois and Iowa. A previous phylogeographic study of *N. exilis* found high levels of genetic structure among populations and deep phylogenetic splits among several geographically defined clades. Variation in fin pigmentation also has been noted. These features suggest unrecognized taxonomic diversity in *N. exilis*. We explored this possibility using comparisons of mitochondrial and nuclear DNA markers and morphological data. High levels of population structure throughout the range of *N. exilis* were observed, and identical haplotypes were only found within river systems and not shared at large distances or among drainages. Three deep phylogenetic divisions (4.8-6.2% sequence divergence) were found: (A) the Red River (White River), (B) the Arkansas and Neosho rivers, and (C) the remainder of the interior and eastern highlands portions of their range. Despite high levels of genetic variation, little variation in morphology was observed across the range of the species. Only those from the Little Red River in Arkansas were morphologically diagnosable. This form is currently being described as a distinct species of madtom as it differs from other populations genetically and morphologically.

STATUS OF THE ACOUSTIC MONITORING OF BATS AT THE U.S. FOREST SERVICE, LAND BETWEEN THE LAKES

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ABSTRACT. The acoustic monitoring of bats in Land Between the Lakes began last summer and continued to expand this year. The sampling includes a total of three driven transects and five stationary monitoring sites. Acoustic monitoring was performed during June and July of 2009 and 2010. Each transect is driven for approximately 30 miles at 15 miles/hr with an ANABAT high frequency recording unit attached to the roof of each vehicle. Acoustic monitoring of bats was performed at stationary sites after transects were recorded using two recording units, Avisoft Ultrasound gate and ANABAT. The transects and stationary sites were GPS tracked and plotted on a GIS map for further investigation. Field acoustic samples were analyzed in the lab using proprietary software. The total number of calls collected for all three transects were 1527 and for all five stationary sites equaled 277. The number of bat calls recorded for 2009 and 2010 varied among each stationary and transect site. We plan to continue the acoustic monitoring of bats this season.

AN INVESTIGATION OF SEX RATIOS OF *PERIMYOTIS SUBFLAVUS* AND OBSERVATIONS OF BREEDING AND MATERNITY BEHAVIORS AT DUNBAR CAVE STATE NATURAL AREA, IN MONTGOMERY COUNTY, TENNESSEE

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ABSTRACT. Dunbar Cave State Natural Area is located in Clarksville, TN. During the spring emergence and summer swarming periods of 2009 and 2010, eleven dates were selected to sample and band bats within and adjacent to the cave. The cave survey hand-removal method was used to trap roosting individuals that were no longer in torpor during the emergence cave surveys. The mist-net trapping technique was implemented in place of the previously used harp-trapping method for capturing swarming individuals due to the detection of White-Nose Syndrome in March of 2010. A total of 218 bats have been captured within and near Dunbar Cave over the 17-month study period. Thirteen of these bats were recaptures, one of which that was trapped twice. Therefore, total of 205 individuals were trapped. All recaptured individuals were *P. subflavus. P. subflavus* was the dominant species sampled in the cave chambers and at the entrance throughout the study. The sex ratio of individuals was investigated in order to determine if *P. subflavus* use Dunbar Cave as a bachelor colony, maternity colony, and/or a swarming site.