PROCEEDINGS OF THE 22ND NORTH AMERICAN PRAIRIE CONFERENCE

North American Prairie Conference

22nd

Tallgrass Prairie
CENTER
University of Northern Iowa

Restoring a National Treasure

August 1-5, 2010

Dave Williams, Brent Butler, Daryl Smith, editors

PAST NORTH AMERICAN PRAIRIE CONFERENCES

1ST 14-15 Sept. 1968 Illinois	Schramm P, editor. Proceedings of a Symposium on Prairie and Prairie Restoration; 1968 Sep 14-15. Knox College Biological Field Station Special Publication No. 3. Galesburg (IL): Knox College; 1970. 66 pages. For availability, contact: Biological Field Station, Knox College, Galesburg, IL 61401.
2ND 18-20 Sept. 1970 Wisconsin	Zimmerman JH, editor. Proceedings of the Second Midwest Prairie Conference; 1970 Sep 18- 20; Madison, WI. Published by the editor; 1972. 242 pages. For availability, contact: James H. Zimmerman, 2114 Van Hise Avenue, Madison, WI 53705.
3RD 22-23 Sept. 1972 Kansas	Hulberi LC (c/o D Hartnett), editor. Third Midwest Prairie Conference Proceedings. 1972 Sep 22-23; Manhattan, KS. Manhattan: Kansas State University; 1973. 91 pages. For availability, contact: Konza Prairie Office, Division of Biology, Kansas State University, Manhattan, KS 66506.
4TH 19-22 Aug. 1974 North Dakota	Wali MK, editor. Prairie: A Multiple View. Grand Forks: University of North Dakota Press; 1975. 433 pages. Pemble RH, Stuckey RL, Elfner LE. Native Grasslands Ecosystems East of the Rocky Mountains in North America: A Preliminary Bibliography. Grand Forks: University of North Dakota Press; 1975. 466 pages. (Neither publication is available).
5TH 22-24 Aug. 1976 Iowa	Glenn-Lewin DC, Landers RQ Jr, editors. Proceedings of the Fifth Midwest Prairie Conference; 1976 Aug 22-24; Ames, IA. Ames: Iowa State University; 1978. 230 pages. For availability, contact: Department of Botany, Iowa State University, Ames, IA 50011.
6TH 12-17 Aug. 1978 Ohio	Stuckey RL, Reese KJ, editors. The prairie peninsula –In the "shadow" of Transeau. Proceedings of the Sixth North American Prairie Conference; 1978 Aug 12-17; Columbus, OH. Columbus: College of Biological Sciences, Ohio State University; 1978. 278 pages. For availability, contact: Ohio Biological Survey, 1315 Kinnear Rd., Columbus, OH 43212-1192.
7TH 4-6 Aug. 1980 Missouri	Kucera CL, editor. Proceedings of the Seventh North American Prairie Conference; 1980 Aug 4-6; Springfield, MO. Springfield: Southwest Missouri State University; 1983. 321 pages. For availability, contact: Department of Biology, Southwest Missouri State University, Springfield, MO 65804.
8TH 1-4 Aug. 1982 Michigan	Brewer R, editor. Proceedings of the Eighth North American Prairie Conference; 1982 Aug 1-4; Kalamazoo, MI. Kalamazoo: Western Michigan University, Department of Biology; 1982. 176 pages. For availability, contact: Department of Biology, Western Michigan University, Kalamazoo, MI 49008.
9TH 29 July-1 Aug. 1984 Minnesota	Clambey GK, Pemble RH, editors. The prairie: Past, present, and future. Proceedings of the Ninth North American Prairie Conference; 1984 Jul 29-Aug 1; Moorhead, MN. Fargo (ND): Tri - College University Center for Environmental Studies; 1986. 264 pages. For availability, contact: Tri-College University, 306 Ceres Hall, North Dakota State University, Fargo, ND 58105.
10TH 22-26 June 1986 Texas	Davis A, Stanford G, editors. The prairie: Roots of our culture, foundation of our economy. Proceedings of the Tenth North American Prairie Conference; 1986 Jun 22-26; Denton, TX. Dallas (TX): Native Prairies Association of Texas; 1988. 344 pages. For availability, contact: Native Prairies Association of Texas, 7171 Mountain Creek Parkway, Dallas, TX 75249.
11TH 7-11 Aug. 1988 Nebraska	Bragg TB, Stubbendieck JL, editors. Prairie pioneers: Ecology, history and culture. Proceedings of the Eleventh North American Prairie Conference; 1988 Aug 7-11; Lincoln, NE. Lincoln: University of Nebraska Printing; 1989. 292 pages. For availability, contact: Department of Biology, University of Nebraska at Omaha, Omaha, NE 68182-0040.

{continued on inside back cover}

PROCEEDINGS OF THE 22ND NORTH AMERICAN PRAIRIE CONFERENCE

Held August 1 - 5, 2010
University of Northern Iowa
Cedar Falls, Iowa

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Gail Zlatnik - Copy editor
Tracey Toenjes - Electronic copy editor

Brent Butler - Design, Layout, Logo, and Cover design

"What a thousand acres of Silphium looked like when they tickled the bellies of the buffalo is a question never again to be answered, and perhaps not even asked."

Aldo Leopold, 1949, A Sand County Almanac

Artwork by: B.A. Seger

PREFACE

A distinctive landscape of long grass, wind and sky once dominated the horizon from the forest margins of Indiana and Wisconsin across Illinois, Iowa, southwest Minnesota and northwest Missouri into the eastern regions of the Dakotas, Nebraska, and Kansas. North to south, it extended from Winnipeg to the coastal prairies of Texas and Louisiana.

The deep black soil created by the prairie proved to be productive. The demise of the tallgrass prairie was rapid and extensive. Usually the prairie wilderness was gone within the first five to ten years after settlers arrived. In the first century after Euro-American settlement, more than 90 percent was converted to agricultural uses. Today, tallgrass prairie is the most decimated ecosystem in continental North America. Less than 2% of the good quality original tallgrass prairie remains, most on land not suited for tilling such as the shallow rocky soils of the Flint Hills of eastern Kansas. Many states like Iowa have lost more than 99.9% of their pre-settlement prairie. In these states, prairie exists as isolated remnants, tiny islands awash in an agricultural sea, scattered among railroad rights-of-way, roadside ditches, old settler cemeteries, rocky outcrops, and other out-of-the-way places.

Peter Schramm initiated this biennial conference in response to "a renewed interest in prairie by people from many different areas of environmental concern." This concern was generated by "the almost complete disappearance of tallgrass prairie in the Midwest, and continuing threats to the few remaining remnants." A perusal of the proceedings of the nineteen conferences since that time will verify that this interest has continued unabated for more than four decades.

The conference theme was "Recapturing a Vanishing Heritage" when we gathered twenty years ago for the 12th North American Prairie Conference at the University of Northern Iowa. The papers in the 12th Proceedings portray that theme as they describe efforts to preserve, understand, manage, restore and interpret remnants of the tallgrass prairie. They are part of the attempt to answer Leopold's question.

The intersection and interaction of land, climate, biota, and Native American culture that created the tallgrass prairie will never occur again. Therefore, in most of the tallgrass prairie region, restoration of degraded remnants and reconstruction of new prairie patches to approximate pre-settlement prairie is the only means to maintain elements of this historic ecosystem. In the latter part of the 20th century, prairie restoration provided the basis and impetus for the relatively new science of restoration ecology. With ever increasing human population and extensive alterations of ecosystems, the 21st century will likely become the Century of Restoration.

"Restoring a National Treasure," was selected as the theme of the 22nd conference to highlight efforts that strive to ask the question that Leopold raised. The four-day conference program included 124 papers and five symposia dealing with a wide range of prairie topics. Like previous conferences, it included a field trip day as well as pre- and post conference field trips. Keynote addresses by Reed Noss, Mark Ackelson, John Price, Richard Leopold, and Daryl Smith provided a broader perspective and additional insights into the uniqueness of the tallgrass prairie and efforts to preserve and restore it.

This volume, the Proceedings of the 22nd North American Prairie Conference, contains manuscripts of 37 papers. Each manuscript was reviewed by one or more persons outside of the editorial staff. A list of reviewers follows this section. Their careful reviews, constructive criticism and suggestions, and timely responses were integral to the successful completion of this publication. We also thank the authors of the papers, without their work and willingness to share their findings with others there would have been no publication.

The North American Prairie Conference provides opportunities to share ideas, engage in stimulating discussions and develop or renew friendships. We lost a good friend and prairie advocate with the death of Paul Christiansen, Professor Emeritus Cornell College, in fall 2007. Paul was one of the early prairie reconstruction experts and mentored many finding their way into the world of prairie. He presented a paper at the first conference in 1968 and was an active participant at subsequent conferences. He continued to contribute to prairie reconstruction and restoration, research, preservation, and management until shortly before his death.

As you read the papers included herein, please ponder on ways we can continue to ask the question, "What a thousand acres of Silphium looked like when they tickled the bellies of the buffalo..." Even though we can't recreate the original tallgrass prairie, attempts at recovery through restoration and reconstruction allow us to learn more about this vanishing ecosystem and help us formulate increasingly better questions. These questions will enable us to learn more about the ecological, economic, educational and cultural benefits of this ecosystem that is part of our biological and cultural heritage. Future generations will benefit from the dividends that accrue from our investment in "Restoring a National Treasure."

Daryl Smith

Daryl Smith

Tallgrass Prairie
CENTER
Restoring a National Treasure

University of Northern Iowa

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SEED AND SOIL ECOLOGY

EDUCATION AND CULTURAL STUDIES

CLOSING KEYNOTE

22ND NORTH AMERICAN PRAIRIE CONFERENCE
SCHEDULE | 2010 | UNIVERSITY OF NORTHERN IOWA

SUNDAY AUGUST 1

TIME	EVENT	PLACE
3-7 P.M.	Registration/Exhibitors Set-up	Schindler Ed Center
6-8 P.M.	Wine & Snacks Social Gathering	Schindler Ed Center | Patio
7 P.M.	Meeting of Session Moderators	Schindler Ed Center | Room 247

MONDAY AUGUST 2

TIME	EVENT	PLACE
8:30 A.M.	Welcome	Schindler Ed Center | Rooms 244, 245, 246
8:45 A.M.	Reed Noss: Southern Grasslands: Ecological History and Future	Schindler Ed Center | Rooms 244, 245, 246
9:45 A.M.	Mark Ackelson: Prairie Preservation and Recovery in Iowa	Schindler Ed Center | Rooms 244, 245, 246
10:30-10:50 A.M.	Break	Schindler Ed Center | Lobby
10:50-11:50 A.M.	Concurrent Session 1	Schindler Ed Center
NOON	Lunch	Meal plan or on your own
1-3 P.M.	Concurrent Session 2	Schindler Ed Center
3-3:20 P.M.	Break	Schindler Ed Center | Lobby
3:30-5 P.M.	Tallgrass Prairie Center Open House/Book signing	Tallgrass Prairie Center
5:30-7:30 P.M.	Marshall Center School Museum Open	North of Schindler Parking
5-6:30 P.M.	2012 Conference Planning Meeting	To be arranged upon request
6:30-7:30 P.M.	Dinner	Meal plan or on your own
8 P.M.	Photographer and author, Bill Witt: Enchanted by Prairie	Schindler Ed Center | Room 246
9 P.M.	Organizational Meeting: Iowa Native Plant Society and Iowa Prairie Network	Schindler Ed Center | Room 122

TUESDAY AUGUST 3

TIME	EVENT	PLACE
8 A.M.-5 P.M.	Field Trips	Meet McLeod Parking Lot
5-6:30 P.M.	Equipment Display	McLeod Parking Lot
5:30 P.M.	Barbecue Social Time (West of the McLeod Parking Lot)	Athletic Field Parking Lot
6 P.M.	Barbecue	Athletic Field Parking Lot
5:45 - 6:30 P.M.	Mesquakie Dancers	Athletic Field Parking Lot
AFTER BBQ	Roadside Discussion	Tent area, Athletic Field Parking
AFTER BBQ	USFWS Prairie and Savanna Forum	Tent area, Athletic Field Parking

WEDNESDAY AUGUST 4

TIME	EVENT	PLACE
8-10:20 A.M.	Concurrent Session 3	Schindler Ed Center
10-10:20 A.M.	Break	Schindler Ed Center | Lobby
10:20-NOON	Concurrent Session 4	Schindler Ed Center
NOON	Lunch	Meal plan or on your own
1-2:40 P.M.	Concurrent Session 5	Schindler Ed Center
2:40-3 P.M.	Break	Schindler Ed Center | Lobby
3-4:20 P.M.	Concurrent Session 6	Schindler Ed Center
4:40-5:20 P.M.	Poster Sessions - Odd numbered poster presenters	Schindler Ed Center | Rooms 320, 321, 322, 325
4:40-5:20 P.M.	Biofuels forum	Schindler Ed Center | Room 252
4:40-5:20 P.M.	Discussions of Implications from small town like writings for our time	Schindler Ed Center | Room 247
5-6 P.M.	Marshall Center School Museum Open	North of Schindler Parking
6-8 P.M.	Social and Banquet	Maucker Union
8:30 P.M.	John Price: A Personal and Literary Journey into Prairie	Maucker Union

THURSDAY AUGUST 4

TIME	EVENT	PLACE
8-10 A.M.	Concurrent Session 6	Schindler Ed Center
10-10:40 A.M.	Poster Sessions - even numbered poster presenters	Schindler Ed Center | Rooms 320, 321, 322, 325
10:40-11 A.M.	Break	Schindler Ed Center | Lobby
11 A.M.-Noon	Closing Session	Schindler Ed Center | Rooms 244, 245, 246
11-11:15 A.M.	Richard Leopold: Importance of Iowa Prairies	Schindler Ed Center | Rooms 244, 245, 246
11:15 A.M.-Noon	Daryl Smith: Restoring a National Treasure	Schindler Ed Center | Rooms 244, 245, 246

22ND NORTH AMERICAN PRAIRIE CONFERENCE CONCURRENT SESSION MODERATORS

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Bohdan Dziadyk
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Jim Stubbendieck
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Marian Langan

Nikki Guillot
Deborah Lewis
Bruno Borsari
Matt Fisher

22ND NORTH AMERICAN PRAIRIE CONFERENCE FIELD TRIPS

DOOLITTLE PRAIRIE, KURTZ RECONSTRUCTION & MARIETTA SAND PRAIRIE

Doolittle State Preserve is a 26-acre site dotted with several prairie potholes, temporary and seasonal shallow wetlands, surrounded by tallgrass prairie. Kurtz Reconstruction is a diverse planting on Carl Kurtz prairie seed farm. Marietta Sand Prairie, a 17-acre state preserve with, "a large and unusual selection of species," (Robert Mohlenbrock) adjacent to a 212-acre Marshall County Conservation Board acquisition with remnant sand prairie and fen.

LOCATION - Doolittle Prairie- NW Story County, Kurtz Reconstruction- near St Anthony, Marrietta Sand Prairie- 8 miles NW of Marshalltown

LEADERS - Carl Kurtz, Marty Malloy, Lloyd Crim

KAUTEN FEN, ROWLEY FEN & PATTON PRAIRIE

Becky's Fen is a rich, 13-acre privately owned site that refused to succumb to tiling. Rowley Fen is a 43-acre property owned by Buchanan County Conservation containing floral gems: grass of Parnassus, numerous sedges, several gentian species and rare sage willow. Numerous butterfly species have also been identified at this site, including sedge skipper, a species of special concern in Iowa. Also owned by Buchanan County, Patton Prairie is a 45-acre wet meadow prairie remnant with 140 native plant species.

LOCATION - Kauten Fen- Near Fayette, Rowley Fen - Near Rowley, Patton Prairie- Near Aurora

LEADERS - John Pearson, Rebecca Kauten, Sondra Cabell

IOWA DNR PRAIRIE RESOURCE CENTER & BRUSHY CREEK STATE RECREATION AREA

The 6,000 acres surrounding the lake provides a variety of opportunities. Stops will include Root Cellar Prairie, a 15-acre remnant overlooking the lake with a small bulldozer salvaged section, savanna, reconstructed prairies and the Prairie Resource Center which includes 80-species, Iowa-origin seed production plots (cardinal flower, blazingstar, birds & insects) and seed processing facility.

LOCATION - Brushy Creek State Park - near Lehigh

LEADERS - Bill Johnson & Eliot LaFollette

FOSSIL AND PRAIRIE PARK PRESERVE & WILKINSON PRAIRIE

Devonian fossils, visitor center, and 60-acre prairie remnant on adjacent knob owned by Floyd County Conservation. The Fossil and Prairie Park is one of the few public parks where you can collect and keep fossils for free- brachiopods, pelecypods, gastropods, horn coral, crinoids, bryozoans and cephalopods weather out of the soft shale, clay and limestone and can easily be picked up by hand. 2nd stop- Wilkinson Prairie is a scenic, 10-acre Cerro Gordo County Conservation property situated on a partially exposed limestone shelf protected between a railroad r-o-w and the Shell Rock River.

LOCATION - Fossil & Prairie Center - near Rockford, Wilkinson Prairie - near Rock Falls

LEADERS - Doug Schroeder, Barb Schroeder and Todd Von Ehwegen

HAYDEN PRAIRIE & DAUBENDIEK PRAIRIE DAUBENDIEK PRAIRIE

Hayden Prairie & Daubendiek Prairie Daubendiek Prairie, a 9-acre privately owned remnant half wet prairie slough/half old field returning to prairie and savanna. At 240 acres Hayden Prairie is Iowa's largest piece of prairie outside the Loess Hills also prized as a rare example of high-quality black soil prairie.

LOCATION - Daubendiek Prairie - 10 miles north of New Hampton, Hayden Prairie - North Central Howard County

LEADERS - Laura Jackson and Mark Leoschke

22ND NORTH AMERICAN PRAIRIE CONFERENCE FIELD TRIPS

HAYDEN PRAIRIE, BLUFFTON HILL PRAIRIE & LUDWIG PRAIRIE

This expedition includes a stop at a high-quality, 240-acre, black soil prairie and for contrast continues east to a hill prairie island in the forest above the Upper Iowa River and if time allows, Ludwig Prairie, an interesting but small limestone outcrop prairie.

LOCATION - Hayden Prairie - North Central Howard County, Bluffton Hill Prairie- Near Bluffton in Winneshiek County, Ludwig Prairie- Near Spillville

LEADER - Larry Reis

HERITAGE VALLEY HILL PRAIRIE AND RAY PRAIRIE

Heritage Valley is an Iowa Natural Heritage Foundation property above the Upper Iowa River featuring multiple hill prairies, woodland and oak savanna. Ray Prairie is a 9-acre mesic to wet-mesic great example of a northeastern Iowa prairie. Home to at least 140 plant species. Substantial management has helped to reduce woody invaders and return fire to the landscape.

LOCATION - Ray Prairie - north of Waverly in Bremer County, Heritage Valley - Upper Iowa River in Allamakee County

LEADERS - Joe McGovern & Tom Rosburg

NEAL SMITH WILDLIFE REFUGE

US Fish and Wildlife's 5,000-acre prairie reconstruction and Prairie Learning Center with rolling prairie vistas and occasional bison and elk sightings. Tour includes US Fish's Land Management Research and Demonstration program for developing new techniques for managing habitats and other ongoing field research.

LOCATION - Neal Smith National Wildlife Refuge, near Prairie City, east of Des Moines

LEADER - Pauline Drobney, Karen Viste-Sparkman, Elizabeth Bach and Korey Kinnick

CEDAR HILLS SAND PRAIRIE & BLACKMUN PRAIRIE CEDAR HILLS

Cedar Hills is a 90-acre sand prairie with sedge meadow, small fen and 360 native plant species owned by The Nature Conservancy. Blackmun Prairie's 175 acres include some exceptional prairie remnants, a diversity of landscape and a mile of meandering North Beaver Creek owned by the Department of Natural Resources.

LOCATION - Cedar Hills Sand Prairie, 10 miles NW of Cedar Falls Blackmun Prairie, near Ackley

LEADERS - Scott Moats and Justin Clark

3 LOCAL REMNANTS

Small remnants visited annually by Daryl Smith's Tallgrass Prairie Seminar; Bennington Township Cemetery (mesic), Dunkerton Railroad Prairie (wet mesic to dry mesic) and Raymond Outcrop (xeric limestone outcrop).

LOCATION - North and east of Waterloo within Black Hawk County

LEADER - Daryl Smith

RESEARCH PROJECTS AT CEDAR RIVER NATURAL RESOURCE AREA

Tallgrass prairie species composition and above ground biomass production, Effects of burn timing on grasshopper and ground beetle assemblages, Effects of mycorrhizal innoculant and micronutrients on prairie establishment.

LOCATION - Northwest of LaPorte City

LEADERS - Dave Williams, Vern Fish, Jim Weimer, Mark Meyers, Molly Schlumbohm, Jim Mason, Cynthia Cambardella, Anna Abney, Ben Hoksch and Chris Barber

CEDAR HILLS SAND PRAIRIE

Cedar Hills is a 90-acre sand prairie with a wide variety of communities: dry sand ridge, revegetated blowouts and dunes, mesic prairie, swale with Palms muck, sedge meadow, small fen and 360 native plant species owned by The Nature Conservancy.

LOCATION - 10 miles NW of Cedar Falls

LEADER - Daryl Smith

CAMPUS AND TALLGRASS PRAIRIE CENTER PROJECTS

Prairie plantings, research plots, the prairie root tubes, Dry Run Creek Restoration, wetland park, campus trail system, pervious concrete, green roof and parking lot infiltration cells.

LOCATION - UNI campus and vicinity

LEADERS - Dave Williams, Rebecca Kauten, Amy Meehan, Billie Hemmer, Kellie Evers, Sarah Benedict, Julie Wynia and Sue Grosboll

CONFERENCE CONTRIBUTORS

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Allendan Seed Company
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Educational Resources in Environmental Science
Iowa Crop Improvement Association
Iowa Natural Heritage Foundation
Iowa Wildlife Federation
JFNews
Living Roadway Trust Fund
(Iowa Department of Transportation)
Midewin National Tallgrass Prairie
Midwest Prairie Paintings
Minnesota Native Wildflower/Grass Producers Association
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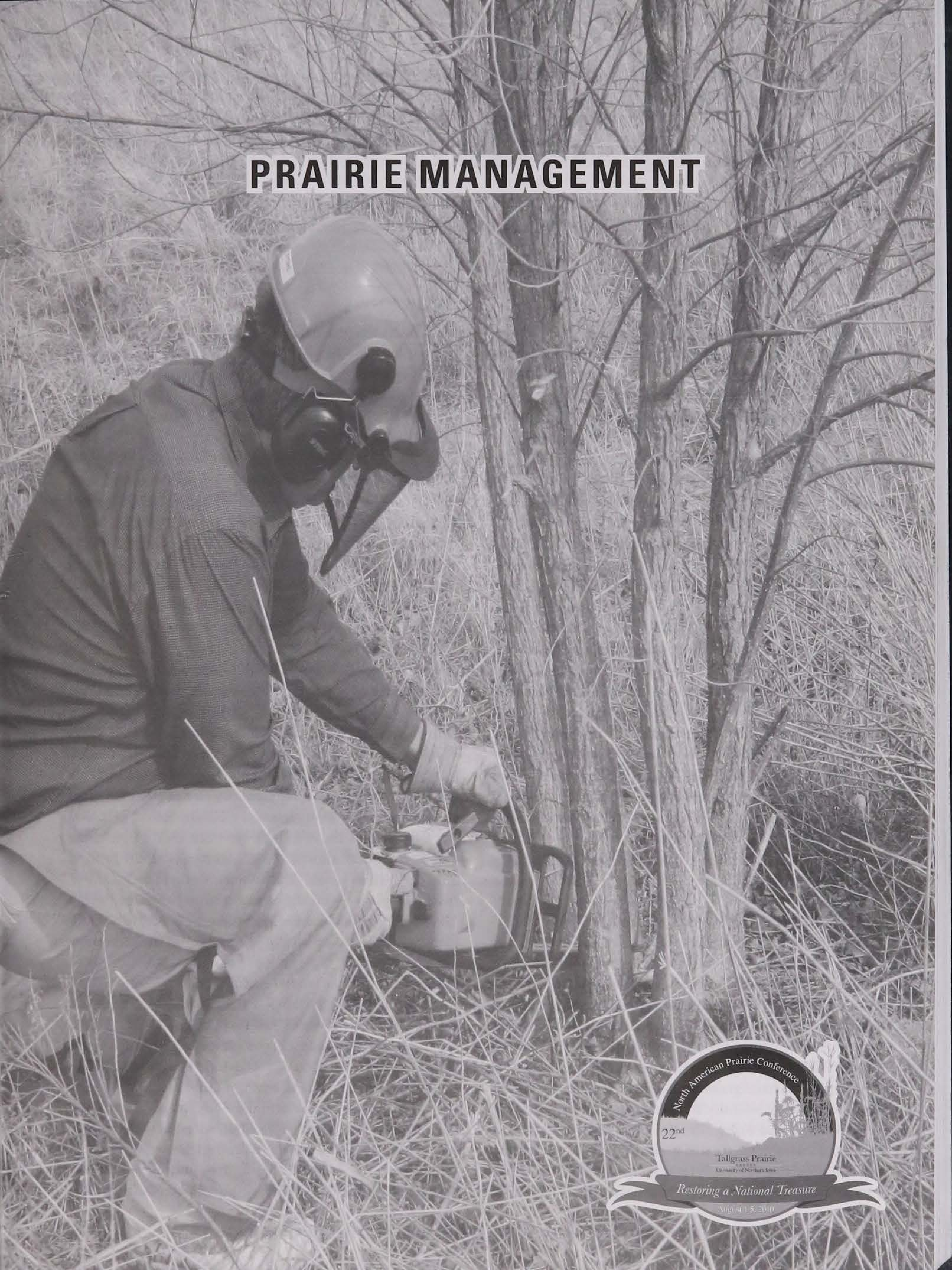

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August 1-5, 2010

CONSERVATION PLANNING FOR PRAIRIE WATERFOWL: WHAT ARE WE DOING FOR GRASSLAND BIRDS?

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Abstract: Conservation planning and management activities in the northern plains have strongly emphasized waterfowl. We address issues involving nongame grassland birds. We argue that most species of prairie waterfowl are in fact grassland birds and, accordingly, protection and management of nesting habitat for waterfowl involves grasslands. Waterfowl differ from other grassland birds in that they have a strong constituency that provides political and financial support for management that benefits them. Many management practices applied for waterfowl also benefit many other species of grassland birds; these practices include protection of wetlands and native grassland, conversion of croplands to grasslands, restoration of wetlands, and periodic management interventions using fire, grazing, and haying. A few waterfowl management practices, such as nest baskets, do not benefit nongame species, and few practices are beneficial to all nontarget species. The consequences of predator management for nongame birds are much more complex than for waterfowl, because nongame birds are influenced by a broader variety of predators. Although most waterfowl management practices are beneficial for a number of nontarget species, nongame grassland birds as a group have more-diverse habitat requirements than do waterfowl and do not respond to management practices as uniformly as do waterfowl.

Key Words / Search Terms: Grassland birds, Great Plains, management, prairie birds, waterfowl

INTRODUCTION

The plight of North America's grassland birds has been widely acknowledged. Most species have declined in number over most of their ranges (Sauer and Link 2010). The primary cause of most declines has been the conversion of grassland to cultivation (e.g., Best et al. 1995), a process that continues still (Askins et al. 2007). Remaining grasslands are mostly used for grazing domestic livestock, a practice that is beneficial to certain grassland bird species, depending on such factors as the intensity, frequency, and timing of grazing (e.g., Kantrud and Kologiski 1982, Bock et al. 1992). Most grassland that is not dedicated to economic uses is managed as wildlife habitat, especially for nesting waterfowl and other game birds. Our purpose here is to assess the influence of waterfowl management on other grassland birds. Specifically, we seek to ascertain management practices that also benefit grassland birds and to identify needs of grassland birds that are not met by management for waterfowl.

A FOCUS ON WATERFOWL

In North America, considerable attention has been devoted to waterfowl and their habitats since at least the 1930s. Because most North American waterfowl are migratory, the federal governments have played major roles. Although the first national wildlife refuges (U.S.) and bird sanctuaries (Canada) were created to protect migratory wildfowl such as pelicans and other colonial bird species (Salyer and Gillett 1964, Taylor and Jorgenson 1985), most refuges had an emphasis on waterfowl (Leopold et al. 1968).

In addition to appropriated funds, hunters provided direct financial support for the acquisition of waterfowl habitat. Beginning in 1934 with the Migratory Bird Hunting Stamp Act, every adult duck hunter in the U.S. has been required to obtain a federal duck stamp, in addition to whatever state licenses are required. Duck stamps cost one dollar at first but have risen in price to fifteen dollars currently. The three-quarters of a billion dollars garnered from the sale of those stamps has been used to purchase and develop waterfowl habitat encompassing 2.1 million ha (5.3 million acres) (http://www.fws.gov/duckstamps/Info/Stamps/stampinfo.htm, accessed 15 November 2010). Likewise, Canadian hunters purchase a Canadian Wildlife Habitat Conservation Stamp in addition to the migratory game bird permit. The stamps provide a source of funds for habitat conservation in Canada.

Many of the duck stamp dollars in the U.S. went for the acquisition of Waterfowl Production Areas. These are typically wetlands or complexes of wetlands with adjoining upland habitats. The U.S. Fish and Wildlife Service (USFWS) owns nearly 3,000 of these sites, which average 90 ha in size (Fischman 2003, http://library.fws.gov/Pubs9/NWRS_waterfowl01.pdf, accessed 15 November 2010). Most are in the northern prairies, protecting critical wetland habitat for breeding and migrating waterfowl. Additionally, the USFWS has purchased conservation easements on even larger areas; these agreements typically preclude the landowner from draining, filling, leveling, or burning wetlands, although they can be cultivated when they are dry (http://www.fws.gov/sacramentovalleyrefuges/pl_easements.html, accessed 26 December 2010).

In the U.S., the Federal Aid in Wildlife Restoration (Pittman-Robertson) Act became effective in 1938 and provided funding to states for various wildlife conservation activities (http://www.fws.gov/laws/lawsdigest/fawild.html, accessed 26 December 2010; http://frwebgate1.access.gpo.gov/cgi-bin/TEXTgate.cgi?WAISdocID=hcL7yu/6/1/0&WAISaction=retrieve, accessed 26 December 2010). These funds are derived from excise taxes on arms, ammunition, and archery equipment. States are required to cost-share at least 25% of project costs. Because federal aid funds are derived primarily from hunters, it was natural that they be used for game species. The nonfederal matching funds are supplied largely by hunters, too, because most state wildlife agencies rely on financial support from license sales.

Flyway Councils were established in 1952 to coordinate the involvement of states and provinces with federal agencies in waterfowl management (Jahn and Kabat 1984). These were set up in alignment with bird migration flyways: Pacific, Central, Mississippi, and Atlantic. A National Waterfowl Council, consisting of representatives from each flyway council, also was formed. These councils deal with coordination and management of state and federal work on waterfowl and recommend annual hunting regulations and management practices and policies (Jahn and Kabat 1984).

Much effort went into surveys of waterfowl, as well. Extensive surveys of breeding waterfowl populations began in the 1940s (Crissey 1984) and became operational in 1955 (Smith 1995). These are the most extensive surveys of any wildlife group in the world. These surveys have institutional funding, supported by the USFWS, Canadian Wildlife Service, and many states.

Waterfowl conservation has also received institutionalized support through the North American Waterfowl Management Plan. Agreed to by the United States and Canada in 1986 and Mexico in 1994, the plan emphasizes the role of partnerships in protecting habitat for waterfowl. The North American Wetlands Conservation Act, passed in 1989, provided a solid base of funding that requires matching by partners. Further, it stipulates that at least half of the annual funding, currently more than $47 million (http://www.fws.gov/birdhabitat/Grants/NAWCA/index.shtm), be directed to Canada and Mexico.

NEGLECTED NONGAME

In contrast to waterfowl and other game bird species, nongame species have been largely neglected by wildlife management. Certainly, many national wildlife refuges (U.S.) and national wildlife areas (Canada) were acquired because of their value to nongame birds, notably colonial waterbirds and species at risk (Salyer and Gillett 1964, Taylor and Jorgenson 1985). The National Audubon Society has protected certain key areas as sanctuaries, The Nature Conservancy and Nature Conservancy of Canada have acquired many unique properties, and local and regional agencies have done likewise. But there has been no extensive and coordinated habitat protection or management program for nongame birds.

One reason for the lack of an international conservation program for nongame birds was the lack of funding. There was no major federal support in the U.S. or Canada. Some states had nongame checkoff programs to collect voluntary contributions from taxpayers, while others had devoted tax funds, or received income from sales of specialized license plates. No such programs existed in Canada.

Further, there was no coordination of nongame efforts. Nothing comparable to flyway councils existed. And in contrast to the well-organized and well-funded surveys of waterfowl, the most extensive surveys of grassland and other nongame birds in North America, the Breeding Bird Survey and the Christmas Bird Count, rely on volunteers to collect information (Robbins et al. 1986, http://birds.audubon.org/christmas-bird-count, accessed 26 December 2010).

Fortunately, the nongame bird scene has changed dramatically in recent years. The Partners in Flight program, launched in 1990, generated a coordinated focus on nongame birds, originally neotropical migrants, but now landbirds in general (http://www.partnersinflight.org/description.cfm, accessed 26 December 2010). Modeled loosely after the North American Waterfowl Management Plan, it brought together numerous federal, state, and provincial government agencies, and nongovernmental organizations concerned with migratory birds. Advocates for other groups of species also developed plans, including the North American Waterbird Conservation Plan (http://www.pwrc.usgs.gov/nacwcp/nawcp.html, accessed 30 December 2010) and the United States (http://www.fws.gov/shorebirdplan/, accessed 30 December 2010) and Canadian (http://www.ec.gc.ca/Publications/default.asp?lang=En&xml=4A90A2A1-1260-41CC-B4F2-4E736D6F6E0E, accessed 30 December 2010) shorebird conservation plans. The North American Bird Conservation Initiative (http://www.nabci.org/about_frame.htm, accessed 30 December 2010) was established to promote integration among these plans as well as other bird conservation activities ("all birds, all habitats").

Funding for nongame programs, at least in the U.S., has been increasing, too. A federal program, the State Wildlife Grants, provided $61 million in 2009 and $76.5 million in 2010 to the states (http://wsfrprograms.fws.gov/subpages/grantprograms/swg/SWG_Funding.htm, accessed 27 December 2010). Funds are intended to be used to benefit "species of greatest conservation need" but not species that receive funding under traditional federal aid programs such as Pittman-Robertson. Each state was required to complete a Comprehensive Wildlife Conservation Plan/Strategy by 2005.

One positive outcome so far of the planning process has been the development of lists of species meriting special attention. In the northern prairies area, those lists include several species closely associated with grasslands (e.g., North Dakota: http://gf.nd.gov/conservation/levels-list.html, accessed 27 December 2010; South Dakota: http://gfp.sd.gov/wildlife/docs/species_concern.pdf, accessed 27 December 2010; Montana: http://fwp.mt.gov/wildthings/conservationInAction/fullplan.html, accessed 27 December 2010), such as northern harrier (*Circus cyaneus*), Swainson's hawk (*Buteo swainsoni*), ferruginous hawk (*Buteo regalis*), upland sandpiper (*Bartramia longicauda*), long-billed curlew (*Numenius americanus*), marbled godwit (*Limosa fedoa*), Wilson's phalarope (*Phalaropus tricolor*), burrowing owl (*Speotyto cunicularia*), Sprague's pipit (*Anthus spragueii*), Baird's sparrow (*Ammodramus bairdii*), grasshopper sparrow (*Ammodramus savannarum*), Le Conte's sparrow (*Ammodramus leconteii*), Nelson's sparrow (*Ammodramus nelsoni*), and chestnut-colored longspur (*Calcarius ornatus*).

MANAGEMENT PRACTICES FOR WATERFOWL: WHAT DO THEY DO FOR NONGAME BIRDS?

As attention has now turned to nongame, and integration of the disparate bird groups is encouraged, it seems appropriate to examine the potential commonalities of waterfowl and nongame species. Here we briefly review the management practices that are commonly employed to favor waterfowl, especially during the breeding season. We then indicate what the consequences of those practices are to grassland birds.

WETLAND PROTECTION AND ENHANCEMENT

Perhaps the most widespread waterfowl management practice is the protection of wetlands. Clearly, the nearly 3,000 Waterfowl Production Areas, each of which may contain several wetlands of a variety of classes and sizes, provide habitat for a broad host of nongame birds as well as waterfowl (e.g., Naugle et al. 2001, Johnson et al. in prep.). So also do the thousands of hectares of wetland protected from destruction by easements. Roughly 1.8 million ha (4.4 million acres) of wetland and associated upland in Canada have been protected under the Prairie Habitat Joint Venture through easements, management agreements, and purchase (http://phjv.ca/pdf/PHJV_Achievements_Web.pdf, accessed 15 November 2010). In addition, many wetlands that had been drained are now being restored, with the provision of waterfowl habitat as a prime motivation. Restored wetlands offer habitat for many nongame birds as well as waterfowl (Ratti et al. 2001).

UPLAND PROTECTION AND ENHANCEMENT

Although protection and restoration of wetlands directly benefit waterfowl and many nongame bird species, grassland birds benefit indirectly. Besides providing wetlands for waterfowl, managers seek to ensure that suitable upland nesting cover is available as well. The uplands surrounding wetlands in Waterfowl Production Areas and other wetlands secured under the North American Waterfowl Management Plan typically are maintained as grasslands, and thus provide breeding habitat for many species of grassland birds (Duebbert 1981, Renken and Dinsmore 1987, Hartley 1994) as well as ducks. Many of these protected sites were formerly croplands and were planted to dense nesting cover (Duebbert et al. 1981), a mixture of introduced species of grasses and legumes. More recently, planting of native species and local ecotypes has been encouraged, although it is not clear that breeding birds use native plantings in preference to introduced plantings; conventional wisdom is that birds respond to vegetation structure, not species composition (Wiens 1969). One farmland retirement program in the U.S., the Conservation Reserve Program (CRP), has converted huge areas of cropland to grassland, mostly of combinations of introduced species of grasses and forbs. Under the CRP, landowners receive annual payments for a 10- to 15-year contract period. These grasslands function much like planted cover for duck nesting and have proven beneficial to both nesting ducks (reviewed by Reynolds 2000) and to many species of grassland birds (reviewed by Johnson 2000). In Canada, the Permanent Cover Program, which converted cropland to perennial herbaceous cover for grazing and haying, also benefits many species of grassland birds (McMaster and Davis 2001) but, like CRP, provides limited benefits to native prairie specialists like Sprague's pipits (McMaster et al. 2005).

The USFWS, in coordination with Ducks Unlimited, Inc., also protects existing grassland through its grassland easement program. Lands enrolled in that program remain in private ownership and can be grazed or (after 15 July) hayed, but not cultivated. This program has permanently protected many thousands of hectares of grassland. Clearly those lands provide valuable habitat for species such as Sprague's pipit, chestnut-collared longspur, ferruginous hawk, grasshopper sparrow, marbled godwit, and many other species that are typically associated with native grassland (reviewed by Johnson et al. 2004).

As a disturbance-dependent ecosystem, grassland requires some form of manipulation. Fire, grazing, and haying are the management practices employed to mimic the natural processes under which grasslands evolved. These practices have both short-term and long-term influences on waterfowl and nongame species. In the northern mixed-grass prairie, for example, prescribed burning will provide habitat immediately for species such as upland-nesting shorebirds and horned larks (*Eremophila alpestris*) but reduce habitat suitability for bobolinks (*Dolichonyx oryzivorus*), grasshopper sparrows, and others which prefer grassland two to three years after a burn (Johnson 1997). Periodic application of prescribed fire enhances the value of grassland for many species (Madden et al. 1999). Haying has a similar mix of short-term and longer-term effects that differ among species (Owens and Myres 1973, Frawley and Best 1991, Dale et al. 1997, Horn and Koford 2000).

In any case, haying later in the season protects both nesting waterfowl and other grassland birds from direct mortality and immediate loss of habitat.

Grazing is much more complicated to assess than are the other management practices, in part because so many different regimes are used. There are also many confounding factors that are difficult to control (Briske et al. 2008). Short-term, intensive grazing removes most of the forage quickly and likely has effects similar to haying. Longer-term grazing will have different effects, depending especially on the intensity, timing, and duration of grazing (e.g., Kantrud and Kologiski 1982). Careful grazing management probably could be used to create a heterogeneous landscape suitable for a wide range of grassland species (Koper and Schmiegelow 2006).

Predation is widely recognized as the single major influence on reproduction and, in many cases, survival of waterfowl (Sovada et al. 2001) and other grassland birds (Davis 2003, Winter et al. 2004). Predator management involves a variety of techniques. Predator reduction, while often controversial, as well as predator exclusion by fencing, can dramatically increase nesting success of waterfowl (Sargeant et al. 1995, Garrettson et al. 1996) and shorebirds (*Charadrius melodus*; Larson et al. 2002). Effects on most nongame birds, however, may be very different (Dion et al. 1999). Reductions of larger predators such as red fox (*Vulpes vulpes*) or coyote (*Canis latrans*) may well lead to increases in numbers of smaller mammals such as ground squirrels and mice, which themselves can be predators of eggs of smaller grassland birds (Pietz and Granfors 2000, Renfrew and Ribic 2003).

Nesting structures are used to provide secure nesting sites for waterfowl. These include boxes for wood ducks (*Aix sponsa*) and hooded mergansers (*Lophodytes cucullatus*) and elevated nest baskets in wetlands for mallards (*Anas platyrhynchos*) and Canada geese (*Branta canadensis*). These structures seem to have little effect on nontarget species, especially grassland birds.

WHY DOES DUCK MANAGEMENT HELP GRASSLAND BIRDS?

Our review demonstrates that most management activities that target waterfowl have beneficial effects on many species of nongame grassland (as well as wetland) birds. The primary exceptions are predator management, whose influence on nongame species is problematic, and nesting structures, with inconsequential effects. The reason for the overlapping benefits of most management practices is rather simple: most dabbling ducks are grassland birds. Although dabbling ducks spend most of their time in the water, the key to their reproductive success—nesting—takes place in the uplands. And most ducks in the Great Plains nest in grassland. Hence, the importance of that habitat is greatly disproportionate to the amount of time they spend in it.

Further, ducks vary in their favored nesting habitat (e.g., Higgins 1977, Cowardin et al. 1985, Klett et al. 1988, Greenwood et al. 1995). Mallards and gadwalls (*Anas strepera*) prefer heavy vegetation, including brush. Blue-winged teal (*A. discors*) and northern shovelers (*A. clypeata*) favor shorter, grassier vegetation. Northern pintail (*A. acuta*) use even sparser vegetation, including heavily grazed prairie and sometimes cropland. Accordingly, meeting the habitat needs of a full suite of upland-nesting ducks requires a variety of grassland habitats, which in turn would support a wide array of nongame birds.

BUT DUCKS *ARE* DIFFERENT

The most obvious difference between ducks and grassland passerines is their size; ducks average about twenty times the size of most passerines. For this and other reasons, ducks are widely hunted. This fact has two important ramifications. First, waterfowl have strong constituencies which are very supportive of their species. Organizations such as Ducks Unlimited, Delta Waterfowl Foundation, and waterfowl associations in California, Minnesota, and many other states both carry out their own management and research activities and encourage state and federal agencies to do so as well. Support for nongame is not nearly as well organized. The large body size of waterfowl also makes it logistically easier to study most aspects of their ecology. They can, for example, carry radio transmitters that allow researchers to obtain much better estimates of vital rates and habitat use and also to understand long-distance movements and linkages among widely dispersed habitats.

The second consequence of the game status of waterfowl is that hunters, who each fall collect samples of birds, many of which have been banded, provide valuable data for determining survival rates and movement patterns of the various species. This information permits a much clearer understanding of the ecology and population dynamics of waterfowl, which in turn facilitates informed management. A duck that is shot benefits science, although not the individual duck itself. In contrast, very few bands of nongame birds are reported (Hobson 2003), and investigators must typically rely on resighting or recapturing birds in subsequent years at the sites where they were banded. Information from such recoveries confounds true survival rates with return rates and provide far less insight into population dynamics (Dinsmore and Johnson 2005: Williams et al. 2002.).

WHAT ELSE DO WE NEED?

We hope to have made it clear that most management activities directed toward waterfowl provide benefits to certain nongame species as well. That is important but not sufficient. Addressing the needs of the full range of grassland and wetland birds, especially priority species, in a region will require more diversity in habitats than waterfowl need. Although ducks differ somewhat in their preferences for wetlands (Stewart and Kantrud 1973) and for upland nesting habitat (Higgins 1977, Cowardin et al.

1985, Klett et al. 1988, Greenwood et al. 1995), as a group they are more homogeneous than are grassland birds in general. Grassland bird communities vary dramatically in response to the height, density, and patchiness of vegetation (Knopf 1996). So, for example, developing a luxuriant stand of vegetation suitable for duck nesting will enhance conditions for nongame birds such as Savannah sparrows (*Passerculus sandwichensis*) and dickcissels (*Spiza americana*) but will be of little value to species such as Baird's sparrow and Sprague's pipit, which favor shorter and native vegetation (reviewed by Johnson et al. 2004).

As conservation plans are developed to accommodate a broader variety of species, it will be important to recognize the specific needs of these species. This will likely require a diversity of natural and restored habitats, as opposed to large areas of monoculture, whatever value those might have for individual species. Habitat diversity need not be provided on a local scale, of course; it is more important that large patches of habitat suitable for a variety of species are available regionally. To maintain populations of grassland birds, it is essential that we better understand differences between their habitat needs and those of waterfowl. It is also important to recognize the spatial and temporal variation in habitat requirements and response to management of a broad array of species. Continued and enhanced efforts to determine how grassland birds—and other taxa—respond to management are a must. Careful assessment of management action will be essential to see whether the intended results are achieved and what unintended consequences might have occurred. An adaptive resource management philosophy (Walters 1986) can be adopted, by which management activities are taken with a specific objective of learning about the system being manipulated.

Finally, basic biological studies of the various species need to be conducted. Understanding the nature of the species of concern will help us anticipate how they will respond to both planned management activities and uncontrolled natural or human actions. Considering how little is known about many nongame species, natural history studies are still appropriate and valuable.

CONCLUSIONS

Most management activities conducted on behalf of waterfowl will provide benefits to many nongame grassland species of birds as well. For example, managing nesting habitat for gadwalls, which favor thick herbaceous cover, will have minimal benefits to northern pintails, which nest in more open situations (Bellrose 1980). Conversely, the nesting cover provided for gadwalls might indeed suit dickcissels and clay-colored sparrows (*Spizella pallida*), whereas extensive tracts of pintail-friendly nesting habitat might be used by chestnut-collared longspurs and Sprague's pipits (Johnson et al. 2004). The needs of individual species must be considered; broad-brush management for "waterfowl" or for "grassland birds" will likely benefit many species within both groups, especially the more generalist species, but species with specialized habitat needs will gain little.

In providing funding for avian conservation activities, hunters in general have been far more willing to open their wallets than have nongame advocates. Because of their activist role, they have influenced the course of wildlife management more than have their non-hunting counterparts. The activities they have encouraged and supported do indeed benefit many nongame birds as well. If nongame advocates desire changes in the direction of wildlife management, such as supporting species that do not benefit from management practices for waterfowl, they will need to become more active, financially and politically.

Integrated bird management, while a noble goal, will require explicit choices and tradeoffs. Decisions that favor one species will slight some others. But the dichotomy is not between waterfowl and nongame birds; it is between particular species. In the end, those decisions will likely be based on human value judgments.

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EVALUATION OF CURRENT AND ALTERNATIVE SPATIAL PATTERNS OF GRASSLAND IN THE LOESS HILLS

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Abstract: Half of Iowa's remnant prairie lies within the Loess Hills of western Iowa, yet development, woody encroachment, and agriculture continue to have an impact upon the size, shape, and quality of grasslands in the region. Given the limited resources available to manage grasslands, prioritizing restoration expenditures and targeting efforts to areas of greatest conservation potential are critical. To this end, we conducted an assessment of landscape patterns in the Loess Hills and developed a conservation priority index (CPI) to identify cropland with the greatest potential to promote connectivity of grasslands. Cropland parcels were given a CPI score between 1 and 100, with high values corresponding to areas having a low corn suitability rating and located close to prairie remnants, and with low values, the reverse. Over the entire Loess Hills landform, croplands dominate, comprising 47% of the land cover, while grasslands, forests, and developed areas comprise 23%, 20%, and 6%, respectively. Cropland patches tend to be contiguous and consist of large, relatively simple shapes, while the patches of remnant prairie are small, fragmented, and far apart. Our analysis also shows that grasslands are more abundant and more connected in the northern half of the Loess Hills. The CPI identified large portions of cropland with low overall agricultural production potential. If cropland areas scoring among the top 30% on the CPI were converted to native prairie, the total amount and connectivity of grasslands in the region would increase substantially, thereby buffering prairie remnants—regionally significant reservoirs of biodiversity—from conservation threats associated with development, woody encroachment, and row-crop agriculture.

Key Words / Search Terms: coarse-filter conservation, conservation priority index (CPI), Iowa, grassland, land use, remnant prairie, spatial pattern

INTRODUCTION

Over half of the remaining native prairie in the state of Iowa is found within the Loess Hills (Mutel 1989, NPS 2002), a region characterized by steep-sided bluffs and long, xeric ridge tops. Although prairie remnants are somewhat consolidated within the landform, conserving these remnants and their associated biodiversity remains difficult due to competitive land uses, such as row-crop agriculture, and the fragmentation that accompanies both rural and urban development (IDNR 2007). Given this situation, advocates of Loess Hills conservation are concerned about the current status of the remnant prairie and how best to carry out much-needed conservation efforts with limited resources. By evaluating the current land cover pattern in the landform, including the total area, shape, and connectivity of each land cover type, we can provide an important, broad-scale context for deciding how to apply limited resources toward on-the-ground conservation action.

Historically, the defining characteristic of grasslands, and specifically the mixed-grass prairies of the Loess Hills, was its continuously expansive, treeless character. Many grassland-obligate species in decline today once thrived in grass-dominated landscapes. Overall, 55 grassland species are threatened or endangered, and 728 species are candidates for listing (Samson and Knopf 1994). Forty-eight percent of grassland bird species within the United States are of conservation concern, and 55% have declining populations (NABCI 2009). In Iowa, 20% of the terrestrial species of greatest conservation need are dependent on warm-season grassland habitat (IDNR 2007). Grassland birds and other obligates are experiencing drastic population declines because of habitat loss and the related, indirect effect of habitat fragmentation (Knopf 1986, Herkert et al. 2003, IDNR 2007).

To accommodate obligate species that require expansive areas of grassland, landscapes must be managed to increase the size and decrease the fragmentation of the remnant prairie communities (Fletcher and Koford 2002, Shepherd and Debinski 2005, Walker 2005). In particular, the Iowa Wildlife Action Plan seeks to create grassland landscapes of 800 ha or more to benefit grassland-obligate species that require large areas (IDNR 2007). While individual pairs of grassland birds may successfully breed in smaller areas, large connected blocks of grassland habitat are required to maintain stable populations. For example, individual pairs of the grasshopper sparrow (*Ammodramus savannarum*) can be found in grassland patches of 30 ha; however, breeding sites measuring 800-1,400 ha in extent are likely required to support breeding populations (Vickery 1996). This example shows how species/habitat interactions are partially defined by the spatial characteristics of habitat over landscapes, such as the overall extent of a habitat type, but its connectivity, the shape of a habitat patch, or the amount of associated edge can also be important factors contributing to habitat quality (Strelke and Dickson 1980, Morgan and Gates 1982, Logan et al. 1985, McGarigal and Marks 1995, Turner et al. 2001). While species/habitat relationships are by their nature organism-specific and best studied on the level of the organism in question, insights can be gained by comparing influential metrics over landscapes and discussing the results in the context of specific organisms (Fischer et al. 2006, Lovell and Johnston 2009). Such a coarse-filter approach, in which the ecological status and value of broad-scale ecosystems and landscapes are assessed, is the only viable option where specific data on species' habitat requirements or response to changes in ecosystem processes are lacking, and in such cases constitute an efficient approach to conservation (Noss 1987, Hunter 1991).

This research focuses on Loess Hills prairie as an ecological community, with the assumption that a coarse-filter approach to conserving this community will benefit a large number of grassland-obligate species, even though the needs of some species are likely to be left unmet (Noss 1987, Groves 2003, Fischer et al. 2006). Grassland birds were selected as a primary example of grassland-obligate species and related species/habitat relationships because of their relatively well documented, rapidly declining populations (NABCI 2009) and for their value as indicators of habitat quality (Browder et al. 2002). Landscape characteristics that prominently and negatively affect many grassland bird species include decreased total area of habitat, decreased connectivity of habitat, and increased edge density (Fletcher and Koford 2002, Walker 2005).

For better or worse, the types of data available often determine the metric(s) used to quantify spatial pattern (Calabrese and Fagan 2004). In the Loess Hills, fine-grain biological data are lacking; however, newly acquired land cover data (Loess Hills Alliance 2008) provide spatially explicit information on vegetation types and arrangements, and are suitable for spatial pattern analysis. In using these land cover data, our objectives were to (1) define and quantify the landscape pattern of Loess Hills grasslands over multiple scales using multiple metrics, (2) develop a conservation priority index (CPI) to identify cropland parcels that would provide the greatest potential for promoting positive landscape characteristics if they were converted to grasslands (3) analyze potential increases in desirable spatial characteristics of grasslands within Special Landscape Areas using the newly developed CPI, and (4) interpret how landscape pattern might influence relationships between grasslands and grassland-dependent species, especially birds.

MATERIALS AND METHODS

STUDY AREA

The Loess Hills landform extends 321 kilometers from Holt County, Missouri, to Plymouth County, Iowa, along the eastern edge of the Missouri River, covering 279,776 ha (Figure 1). This distinctive geological landform is globally unique with respect to the deep deposits of loess, ranging from 18m to over 60m deep, and the highly dissected nature of the region (NPS 2002). The loess deposits are composed of sediment swept up from the nearby Missouri River floodplain and largely deposited over the last 30,000 years (Bettis 1990, NPS 2002).

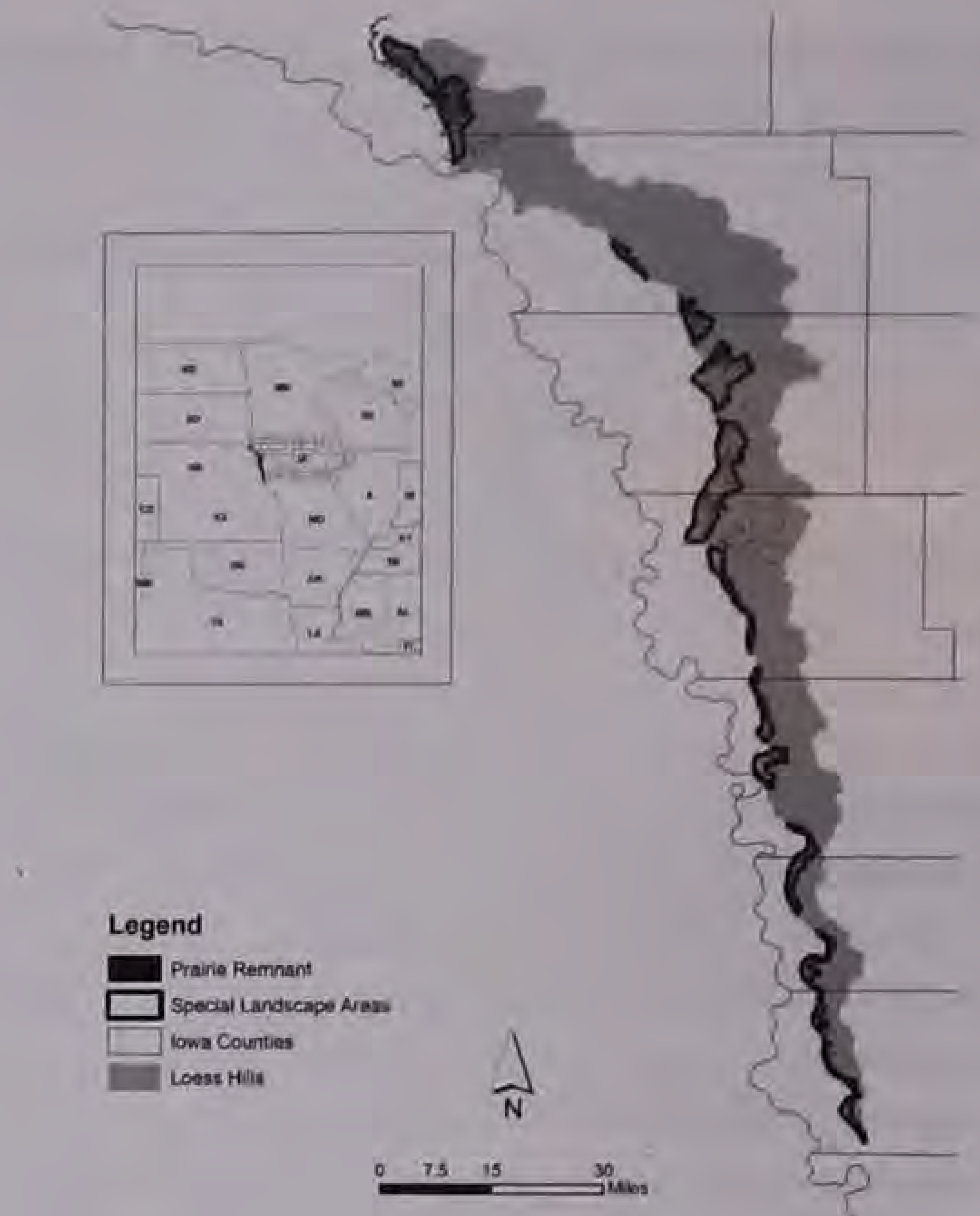

Figure 1. Map showing the location of Special Landscape Areas (SLAs) within the Loess Hills region of western Iowa, USA.

The average minimum and maximum temperatures for the Loess Hills region are -10.5° to 32.7°C (13°-91°F), respectively (US EPA 2009). Average annual precipitation ranges from 650.2 to 873.8 mm (25.6-34.3 in) (Bettis 1989), with higher precipitation levels in the southern hills. Well-drained aeolian soils in combination with steeply dissected topography create moisture-limited conditions on the upper portions of the slopes. Ridges, draws, and valleys also provide sheltered areas with moister microclimates, creating additional variability.

Historically, prairie vegetation was dominant throughout the Loess Hills, but areas with woody vegetation were patchily distributed within this matrix of open lands and were most often located in sheltered ravines and riparian areas (Mutel 1989, NPS 2002, Agren Inc. 2004, Stambaugh et al. 2006). Principle prairie species in the region include sideoats grama (Bouteloua curtipendula), little bluestem (*Schizachyrium scoparium*), big bluestem (*Andropogon gerardii*), heath aster (*Aster ericoides*), blue-eyed grass (*Sisyrinchium montanum*), Scribner's panic grass (*Dicanthelium oligosanthes*), whorled milkweed (*Asclepias verticillata*), Missouri goldenrod (*Solidago missouriensis*), leadplant (*Amorpha canascens*), and plains muhly (*Muhlenbergia cuspidata*) (Rosburg 1994). Bur oak (*Quercus macrocarpa*) dominated the majority of fire-tolerant savanna and woodland communities (Mutel 1989).

Loss of the historical fire regime (fire intervals of 2-7 years; Stambaugh et al. 2006), the temperate climate, and the topographic variability have led to an increase in woodland vegetation in the landform, where mesic deciduous forest and xeric prairie communities occur in relatively close proximity to one another. Today, 20% of the landform is in woodland or forest vegetation compared to presettlement estimates of 11%, and only 23% is in grassland vegetation, of which 3% is considered remnant prairie, compared to presettlement estimates of over 90% grassland (Farnsworth 2009).

In 2002, the National Park Service identified 12 Special Landscape Areas (SLAs) as "clusters of exemplary prairie and geological/topographic features based on past field surveys of prairie, forests and geological features" (NPS 2002) (Figure 1). These SLAs comprise 40,472 ha (100,000 ac) in 12 discrete areas spanning the Loess Hills, and contain greater than 80% of the region's biodiversity (NPS 2002). The SLAs are high-priority areas for a number of conservation organizations and thus are a focus of our research.

LOESS HILLS LAND COVER DATA

The primary source of data used in this study was a land cover classification performed by Saint Mary's Geospatial Services, Winona, Minnesota (Loess Hills Alliance 2008). Land cover units were classified and digitized as polygons using a 0.4 ha (1.0 ac) minimum mapping unit with one-meter-resolution National Agriculture Imagery Program aerial photographs (NAIP 2006) and one-meter-resolution 2002 false-color infrared imagery. Air photo interpretation was informed with field-based vegetation samples. A 1992 Gap Analysis Program (GAP) land cover dataset was used as ancillary data in conducting the classification.

Twenty-eight classes were identified in the classification following the National Vegetation Classification Standard for the Midwest, as modified by The Nature Conservancy (Iowa GAP Analysis Program 2001, Loess Hills Alliance 2008). For this analysis, we combined classes into broader land cover types and focused our analysis on those four classes that included the majority of the landform: croplands (i.e., corn, soybean), forests (e.g., deciduous woodland, red-cedar, grasslands (e.g., hay, brome, prairie), and development (e.g., residential, industrial). We calculated total area, mean patch size, edge density, nearest neighbor, and patch cohesion for these four major land cover classes to make more-specific vegetation comparisons and to provide an overall synopsis of Loess Hills land cover (McGarigal and Marks 1995, Turner et al. 2001).

GRASSLAND ANALYSIS

Land cover data for the Loess Hills (Loess Hills Alliance 2008) and a modified version of a Loess Hills remnant prairie data layer from The Nature Conservancy (TNC) were used as inputs for spatial pattern analysis of the four grassland categories: remnant prairie, warm-season grassland, cool-season grassland, and combined warm- and cool-season grassland. It should be noted, however, that the prairie remnant and warm-season grassland classes are not independent of one another. The prairie remnant class was extracted from the grassland land cover class using a version of the remnant prairie data layer obtained from TNC, modified from the original to improve its accuracy (Farnsworth 2009). There is roughly a 35% overlap between prairie remnant and warm-season grassland polygons. While this relationship compromises any statistical comparison among these classes, we found the information nonetheless useful to the overall understanding of landscape patterns in the Loess Hills.

For all data, we converted GIS vector-based data layers to raster format and overlaid the boundaries of the Loess Hills SLAs to extract land cover data per individual SLA. Grassland pattern analysis was conducted on the entire Loess Hills landform and within SLAs using ArcMap (ESRI 2009) and FRAGSTATS (McGarigal and Marks 1995), a computer software program designed to quantify the spatial characteristics of landscapes. Landscape metrics included total area, mean patch size, nearest-neighbor mean, nearest-neighbor coefficient of variation, patch cohesion, and edge density. These metrics were specifically chosen for their ability to describe the structural pattern of communities over landscapes while also inferring potential connectivity. Structural descriptors are best used when organism-specific information is lacking (McGarigal and Marks 1995). Total area, mean patch size, and edge density all relate to core area, which is an important spatial quality of habitat that affects population dynamics, specifically for grassland birds (Coppedge et al. 2001, Johnson and Igl 2001). While an important factor, core area is an organism-specific metric that could not be calculated here because of a lack of organism-specific information for the Loess Hills. Thus, this research evaluates landscape patterns relative to grassland-obligate species in general, and does not consider the specific spatial requirements of individual organisms. Nearest-neighbor and patch cohesion values are class metrics that can be used to evaluate the relative connectivity of a community type (McGarigal and Marks 1995). Connectivity is a major factor considered in preserve design and species conservation (Diamond 1975, Groves 2003). These metrics are also landscape structural attributes that can be defined without knowledge of organism-specific requirements following a coarse-filter approach.

CONSERVATION PRIORITY INDEX (CPI)

We developed a conservation priority index (CPI) to evaluate current areas of row-crop agriculture for their potential to contribute to grassland conservation. The CPI values a particular map pixel according to the distance to the nearest prairie remnant and its corn suitability rating (CSR). The additive value is weighted by the maximum score of the nearest remnant and CSR value, subtracted from one, and then multiplied by 100 to create a relative index (Equation 1).

Equation 1.
CPI = [1- ((CSR+distance) / maximum(CSR+distance))]*100

We hypothesize that restoration of cropland close to prairie remnants will disproportionately affect connectivity, patch size, and edge density in a positive direction from a grassland conservation perspective. Prairie remnants are often relegated to the most rugged terrain and consequently are less suitable for cultivation due to extreme slope and less-productive soils. Cropland closest to prairie remnants is hypothesized to have a higher chance of connecting to other nearby remnants or contributing to the patch size of an individual remnant patch if it were converted back to grassland.

Including CSR in the CPI provides additional benefit. The calculation of CSR takes weather, soils, slope, and other soil profile properties into consideration (ISU Extension 2005). A low CSR rating suggests that these areas are not well suited for the production of row crops. CSR thus can infer the potential for restoration from an economic perspective. If a parcel of land is less suitable for row-crop production (i.e., low CSR), it is likely to have lower market value in comparison to land with high CSR. Lower crop productivity can equate to a higher probability for restoring or conserving native vegetation.

Spatial data on CSR was obtained from an Iowa Soil Properties and Interpretations Database (ISPAID) and converted to raster format. ISPAID data were then extracted for cropland identified in the 2008 vegetation classification for the whole landform and then again by SLAs. CPI was calculated using this CSR raster layer and a raster layer showing distance to remnant constructed using the Euclidean distance tool in the ArcGIS (ESRI 2009) toolbox.

Increments of 10%, ranging from 0% to 100% of the total CPI raster score, were selected and made into separate data layers, merged with the modified remnant data layer, and analyzed in FRAGSTATS. These incremental layers were analyzed for potential connectivity using the patch cohesion metric in FRAGSTATS (McGarigal and Marks 1995). We calculated edge density by using Xtools extension in ArcMap (ESRI 2009) to determine the perimeter and area of polygons using the same incremental framework as patch cohesion calculation. At each increment, edge was calculated by taking the log transformed result of the perimeter divided by area (Equation 2).

Equation 2.
Edge density = ln(perimeter/area)*

*Results were log transformed and plotted against incremental CPI scores.

RESULTS

OVERALL LAND COVER ANALYSIS

Within the entire Loess Hills landform, croplands are dominant, comprising 47% of the land cover; grasslands, deciduous woodlands and forests (hereafter, forests), and developed areas comprise 23%, 20%, and 6% of the land cover, respectively (Table 1). The cropland class has the largest mean patch size, lowest edge density values, and largest cohesion. In other words, cropland patches tend to be contiguous, and consist of large, relatively simple shapes. By comparison, grassland and forest patches are much smaller (roughly one-sixth the size), tend to have a greater amount of edge, and be less well connected (Table 1). Forests tend to be further apart and have higher edge densities than all grasslands combined (Table 1). The land cover pattern in individual SLAs largely mimicked the patterns of the region as a whole (Farnsworth 2009); however, the six northern SLAs contained a comparatively larger proportion of grassland land cover (86%), while the six southern SLAs were occupied by much less grassland cover (14%).

GRASSLAND LAND COVER COMPARISON

The total area in prairie remnants was smaller than all other grassland vegetation types (Table 2). Total area and mean patch size increased between remnant, cool-season, and warm- and cool-season grassland combined. This trend of increasing values from prairie remnant to combined grassland land cover was consistent for metrics describing individual patches (i.e., total area, mean patch size), but metrics describing spatial arrangement (i.e., patch density, mean nearest-neighbor distance, and edge density) showed different results (Table 2). Using patch density and Euclidean distance nearest-neighbor values, remnant prairies were shown to be closer to each other than warm-season, cool-season, or combined warm- and cool-season grassland land cover types (Table 2). The combination of mean nearest-neighbor distance and coefficient of variation in this measure, which considers the standard deviation in nearest-neighbor distance relative to the mean, revealed that prairie remnants are closer to each other on average than other grassland cover types, but that this pattern is highly variable (Table 2). The edge density metric supported the dissected pattern among prairie remnants, with prairie remnants having high edge density in comparison to the other grassland cover types. The other grassland cover types were blocky in shape compared to the prairie remnants (Table 2). Overall, prairie remnants tend to be

Table 1. Quantitative comparison of spatial patterns among dominant land cover classes in the Loess Hills region of western Iowa, USA, based on 2006 imagery.

METRICS	CROPLAND	GRASSLAND	FOREST	DEVELOPED
Total area (ha)	132,585.6	65,473.7	55,205.2	16,843.0
Percent area (%)	47.4	23.4	19.7	6.0
Mean patch size (ha)	74.0	12.6	11.6	8.4
Edge density (m/ha)	366.6	477.5	531.2	360.2
Mean nearest neighbor (m)	115.7	114.4	127.3	382.7
Patch cohesion (index score)	99.7	97.7	97.8	97.8

small, fragmented, and have high amounts of edge compared to other grassland types (Table 2) as well as to croplands, forests, and developed areas (Table 1).

Among the 12 SLAs, those in the northern half of the Loess Hills consistently show higher amounts of grassland vegetation and greater connectivity. The second most northern SLA had the highest value of total grassland area (2,747 ha), second highest cohesion (99.6), and nearest-neighbor distance (86.7m) between grassland cover types. The southernmost SLA showed the opposite scores (grassland area = 250 ha; cohesion = 88.8; nearest-neighbor distance = 119.9 m).

CONSERVATION PRIORITY INDEX (CPI)

The overall distribution of CPI scores was skewed toward 100; in other words, toward values with high conservation potential (Figures 2 and 3). Eighty-three percent of all cropland within SLAs (33,591 ha) scored over 70 on the CPI (Figure 2), showing that much of the cropland within SLAs has a low CSR and is located proximal to prairie remnants (Figure 3). Cropland areas with a CPI score greater than 70 would account for 93% of the increase in the connectivity of prairie remnants if these areas were converted to grassland (Figure 4). We found similar results for edge density: croplands with a CPI score of >70 captured 92% of the potential improvement (low edge density) (Figure 5). Converting just 30% of the highest-scoring croplands to grassland, however, would result in substantial increases in patch cohesion and decreases in edge density (Figures 4 and 5).

DISCUSSION

The lack of in-depth knowledge on the status and trends of species and ecosystems, the extreme degree to which many natural systems have been altered, and the limited availability of resources for restoration often necessitates a coarse-filter conservation approach (Noss 1987, Groves 2003, Fischer et al. 2006, Seastedt et al. 2008). The evaluation and restoration of key landscape characteristics, with an eye toward a handful of desirable species for which the habitat requirements are better known, is a coarse-filter approach intended to preserve a significant portion of biodiversity, while acknowledging that some species will not be accommodated (Groves 2003, Fischer et al. 2006).

The Loess Hills of western Iowa is one such region where a coarse-filter conservation approach is needed, due to the high level of ecosystem alteration and lack of information on the needs of individual species. Although the region contains one-half of the remnant prairie in the state of Iowa, cropland still comprises the majority of land cover, followed by grasslands, forests, and developed areas (Table 1). While prairie remnants harbor much native biodiversity, they are small, fragmented, and largely relegated to the more rugged and agriculturally unfit portions of the landscape (Table 2).

Substantial opportunity exists to improve the structural characteristics of these prairie remnants, as well as surrounding grasslands, for the purposes of biodiversity conservation. Such improvement could be achieved in two ways: first, by improving the structural characteristics and management of existing nonnative grassland patches. At the patch level, many grassland obligates, especially birds, show a lower response to the species composition of grasslands than to their structural characteristics (Chapman et al. 2004). For example, dickcissels (Spiza americana) will occupy fallow fields and unmown hayfields in addition to native prairie, but the proximity to

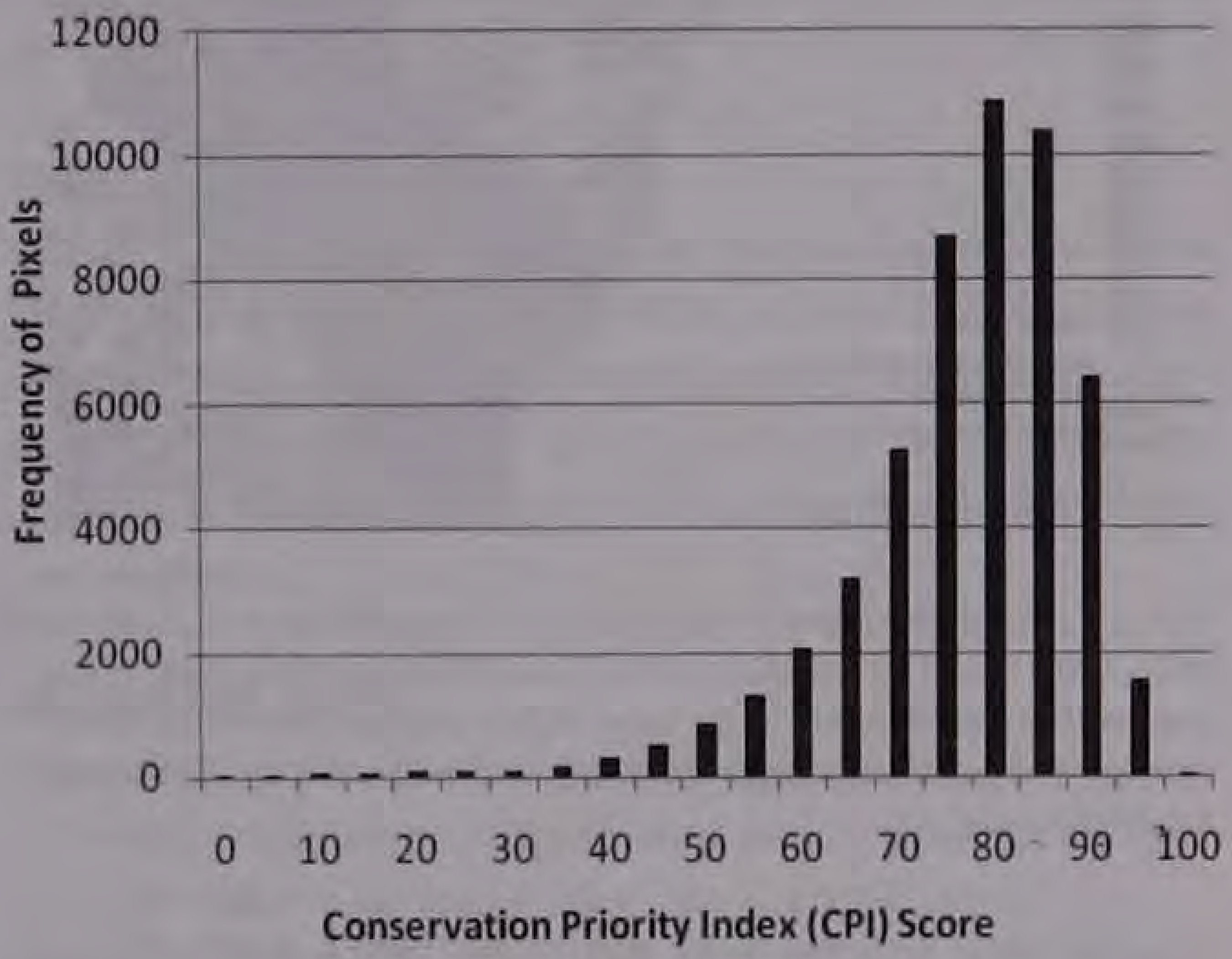

Figure 2. Frequency of cropland pixels by Conservation Priority Index (CPI) scores.

Table 2. Quantitative comparison of spatial patterns among grassland land cover classes within the Loess Hills of western Iowa, USA. Note that prairie remnants are not completely independent of the warm-season grassland class (see methods for details).

METRICS	PRAIRIE REMNANTS	WARM-SEASON GRASSLAND	COOL-SEASON GRASSLAND	WARM & COOL-SEASON GRASSLANDS COMBINED
Total area (ha)	8,574.8	14,144.3	49,391.9	63,536.2
Patch density (ha/patch)	26.3	13.4	8.4	7.2
Mean patch size (ha)	3.8	7.4	11.9	13.9
Mean nearest neighbor distance (m)	121.5	222.9	140.4	123.0
Nearest neighbor coefficient of variation (index score)	133.8	144.2	91.4	84.0
Patch cohesion (index score)	95.1	93.9	96.8	97.7
Edge density (m/ha)	285.8	221.2	177.6	167.8

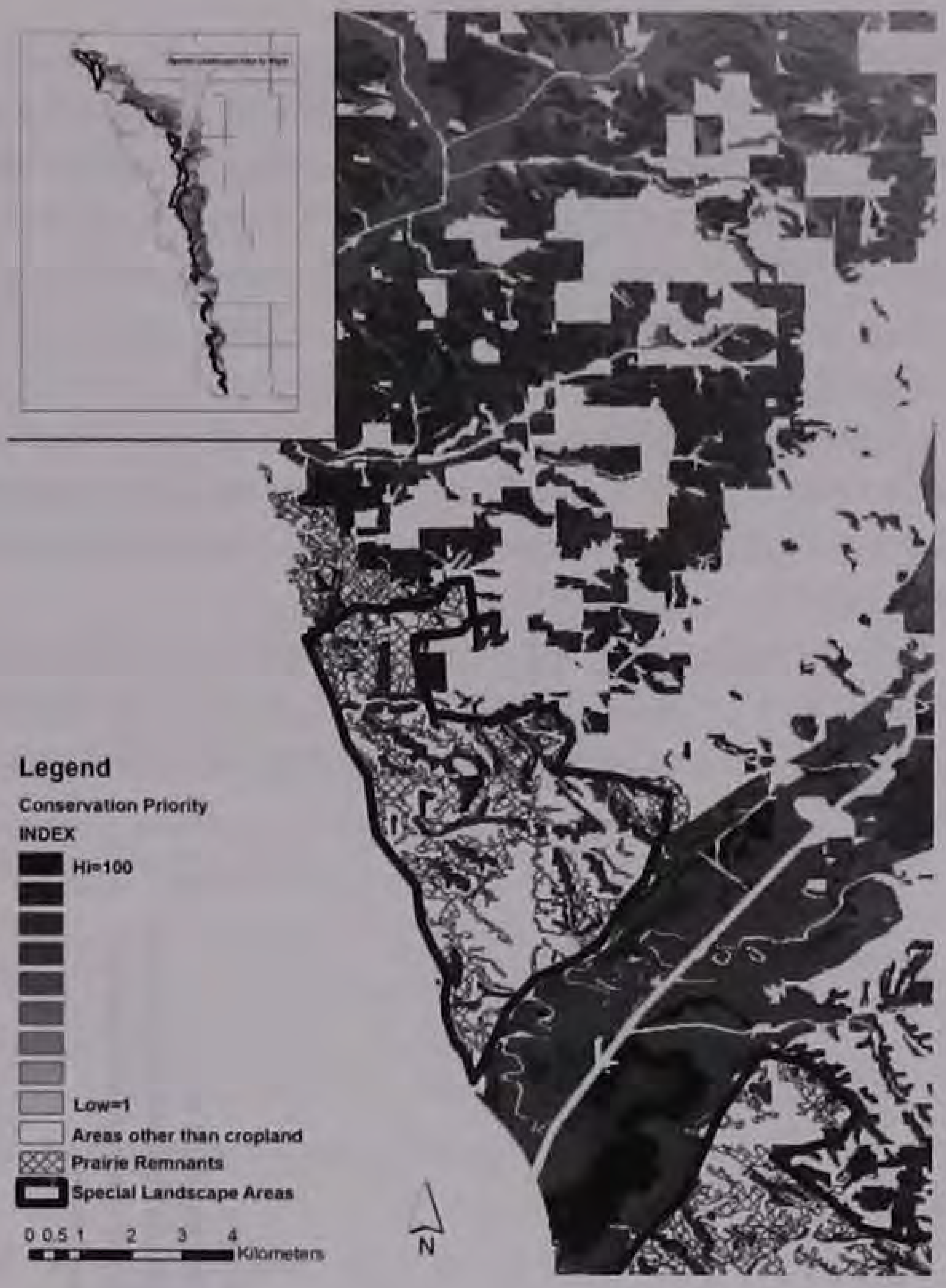

Figure 3. Spatial distribution of remnant prairies and Conservation Priority Index (CPI) scores for a portion of the Loess Hills in western Iowa. White areas within the boundary of the Loess Hills landform include non-prairie remnant and non-cropland cover types (i.e., other grasslands, forests, and developed areas).

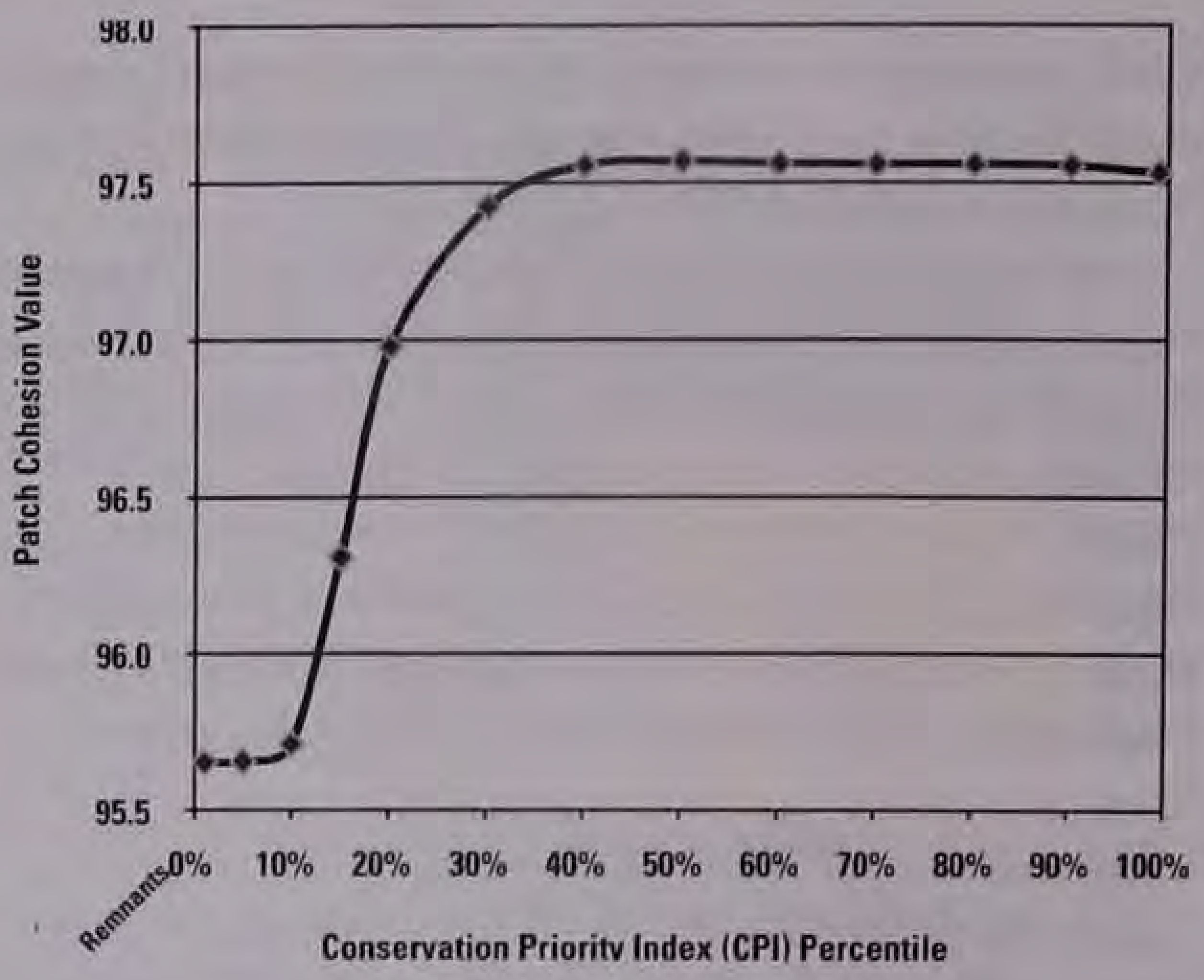

Figure 5. Potential decrease in edge density of prairie remnants if cropped areas within Loess Hills Special Landscape Areas (SLAs) displaying high values of a conservation priority index (CPI) were restored to native vegetation.

wooded areas decreases the quality of these habitats (Temple 2002). The fact that grasslands comprise a substantial proportion (26%) of the land cover in the Loess Hills represents a substantial opportunity; yet, most of this vegetation type consists of cool-season nonnative species, and includes pastures that are often grazed or hayed and grass plantings associated with the United States Department of Agriculture's Conservation Reserve Program (CRP). The habitat value of these cover types for grassland birds and other obligate-grassland species is highly variable (McCoy et al. 1999, Johnson 2000). Appropriately applied livestock grazing (Chapman et al. 2004), conservation strategies such as delayed haying (Horn and Koford 2000, Perlut et al. 2006), and other forage-reserve strategies can enhance the contribution of cool-season grasslands to conservation by increasing the available cover for breeding birds and other obligates (Patterson and Best 1996). Cool-season grasslands can also contribute to the overall extent of open, grass-covered habitat, decreasing the amount of edge and fragmentation associated with remaining remnant prairies. While cool-season grasslands might not meet the plant compositional needs of some grassland-obligate species (e.g., specific nectar- or fruit-bearing plants), they can meet some of the structural criteria (e.g., permanent cover, lack of perches for predators). Additionally, many critical ecosystem processes, such as the movement of organisms, the redistribution of nutrients, and the spread of natural disturbance, are linked to the size and spatial arrangement of patches (Johnson and Igl 2001, Turner et al. 2001, Shepherd and Debinski 2005, Fischer et al. 2006). Increasing the total area of grassland has the potential to improve the spread of fire by increasing the spatial extent of more flashy fuels (i.e., grass) and therefore improving the effectiveness of prescribed fire management in reducing woody encroachment.

Second, croplands of low economic value and located close to remnant prairie could be targeted for conversion to reconstructed prairie, thereby increasing the overall extent and connectivity of grasslands in the Loess Hills. This approach may be more cost-effective compared to focusing on restoration of overgrown areas commonly found within and around prairie remnants, because intensive woody vegetation removal can be quite costly, ranging from $620 to $2,500

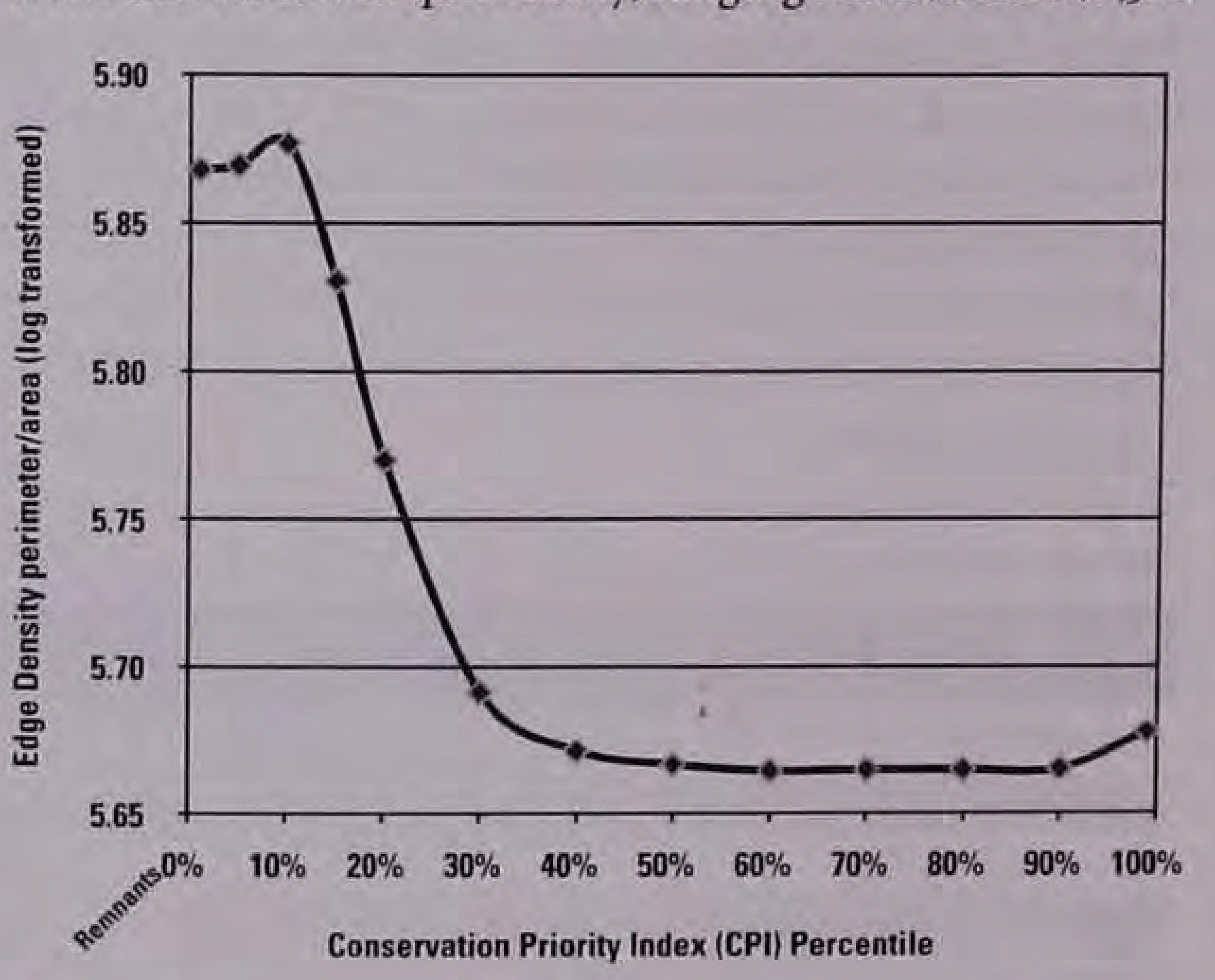

Figure 4. Potential increase in patch cohesion of prairie remnants if cropped areas within Loess Hills Special Landscape Areas (SLAs) displaying high values of a conservation priority index (CPI) were restored to native vegetation.

per hectare ($250-$1,000/ac). In many cases, we expect that the reconstruction of native grassland on former croplands will be more efficient at increasing core area for some area-sensitive species while lessening the impact of edge effect, than removing forest on lands once occupied by remnant prairie. We developed the conservation priority index (CPI) with this in mind.

We analyzed the CPI within SLAs, and our focus on them represents a further targeting mechanism—SLAs were previously selected by the Loess Hills Alliance, a multi-stakeholder conservation coalition—as focal conservation areas, thanks to the density of prairie remnants and the overall level of biodiversity contained within. The CPI further assists in the spatial prioritization of cropped areas that, when restored to prairie, might disproportionately affect the connectivity of remnants relative to their total extent. We found that the CPI revealed that a large proportion of land within SLAs was both near prairie remnants and had low suitability for growing corn. If croplands with high CPI values are converted to reconstructed prairie, dramatic increases in the total area and connectivity of grassland, along with a reduction in edge density, will result (Figures 4 and 5).

Challenges to both of these approaches remain, however. For example, sizable pockets of mature forest occur even within grassland-dominated areas of the Loess Hills today, compromising their potential to contribute to the effective conservation of grassland-obligate species in the region (Samson and Knopf 1994, Temple 2002, Walker 2005). Also, while grassland-dominated landscapes are larger and prairie remnants are relatively more connected in the Loess Hills compared to other regions of Iowa, they still may not be suitable for some species that are sensitive to edge effects and require large areas of core habitat, such as bobolinks (*Dolichonyx oryzivorus*) and northern harrier (*Circus cyaneus*) (Johnson and Igl 2001). Lastly, while the density of prairie remnants is higher within SLAs, the SLAs are 5.5 km apart on average and remain isolated from one another at the scale of the entire Loess Hills region (Figure 1). Thus, while efforts to alter the spatial characteristics within SLAs may prove successful to enhance the extent and connectivity of grasslands for grassland-obligate species, the conservation of these species could be still be thwarted by the lack of connectivity among SLAs.

CONCLUSION

While prairie remnants represent the richest pool of native biodiversity among land cover types in the Loess Hills, the landscape character of the region at present compromises their contribution to biodiversity preservation. Species dependent on grasslands require landscapes with large open spaces covered with perennial vegetation (Samson and Knopf 1994, Herkert et al. 2003). The Iowa Wildlife Action Plan is therefore seeking to create grassland landscapes of 800 ha or more to benefit grassland-obligate species that require large areas (IDNR 2007). Fulfilling the habitat requirements of these species necessitates the expansion of grassland habitat and an increase in its connectivity, if preservation goals are to be met. Effective methods of habitat expansion require a targeted approach to make the most of limited conservation funding. The future reconstruction of native prairie and the modification of nonnative grassland management should focus on portions of the landscape that provide the greatest potential conservation gains per expenditure of resources. We conducted our assessment with the goal of informing such a targeted approach and thereby revealed key landscape-level limitations associated with the current configuration of remnant prairie and other grassland types. In developing a Conservation Priority Index, we offered a mechanism to further prioritize the expenditure of restoration resources to improve the structural qualities of the Loess Hills landscape now and for the future.

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LESSONS LEARNED FROM A DECADE OF PROTECTING AND MANAGING PIONEER CEMETERY NATURAL AREAS IN ILLINOIS

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Abstract: The Illinois Natural Areas Inventory (INAI), conducted in the late 1970s, examined 3,923 pioneer cemeteries and identified 24 of these with prairie and savanna communities worthy of listing as high-quality natural areas. There are now 29 pioneer cemeteries on the INAI; 21 of these have received permanent legal protection. In spite of the high percentage of cemeteries protected, enormous management challenges remain as a result of the need to consider both the ecological and sociological impacts on these small sites, averaging 2.2 acres. The attitude of the cemetery owners and the local community greatly influences the type and extent of traditional prairie management practices which can be implemented within the cemetery. Illinois state law allows for the legal protection of natural vegetation communities through the Natural Areas Act in conjunction with the protection of the cultural resources within the cemetery. The Illinois Nature Preserves Commission (INPC) has employed various adaptive management strategies over the past decade on cemetery natural areas throughout Illinois. Management challenges and strategies to consider include public relations; multidiscipline cooperation; documentation of the condition and location of burial markers; cleaning, repairing, and resetting stones; fencing; trail maintenance; changes to hydrological drainage and soil erosion; loss of plant species diversity; alterations to the timing and extent of prescribed burns; use of herbicides to control invasive exotic and ornamental plants; and the expansion of buffer zones.

Key Words/Search Terms: pioneer cemetery, cultural heritage protection, prairie management

CEMETERY PRESERVATION IN ILLINOIS

The Illinois Nature Preserves Commission (INPC), through the Illinois Natural Areas Preservation Act (525 ILCS 30) has the ability to legally protect pioneer cemeteries with high-quality native remnant plant communities. The State of Illinois defines a cemetery as "a parcel of land devoted to, or at least a part of which has been used for, the interment of human remains "(525 ILCS 30/3.03, Ch. 105, par. 703.03)." Illinois does not provide a legal definition for "pioneer cemetery." The term is used here to describe those cemeteries which contain the earliest documented and visible burials in a region, with an understanding that additional burials might continue into the twenty-first century. (In contrast, the State of Iowa legally defines a pioneer cemetery as one with twelve or fewer burials in the past fifty years.)

During the late 1970s, as part of the Illinois Natural Areas Inventory (INAI), 3,923 pioneer cemeteries were examined for high-quality remnant natural vegetation. Twenty-four were identified as having high-quality prairie and savanna vegetation and were added to the INAI (Kerr and White 1981). There are currently 29 pioneer cemeteries recognized by the INAI (Illinois Department of Natural Resources). Of these 29 cemeteries, 21 have been permanently protected in the Illinois Nature Preserves System (Illinois Nature Preserves Commission). Many of these are small—less than 0.81 hectares (2 acres)—unregistered cemeteries in which the burial sites are scattered within the prairie or savanna community. These cemetery natural areas preserve some of the best examples of black soil mesic prairie left in Illinois (Betz 1972, Illinois Natural History Survey 1994). The scarcity of this habitat and an increased public awareness of prairies, in conjunction with an increased public interest in genealogical/ancestral heritage research, have resulted in numerous inquiries made to INPC regarding the care or management of these cemeteries. While the views on how cemeteries should be maintained often divide people, the preservation of both cultural heritage and natural heritage does not have to be divisive (Moorehouse and Hassen 2006). It is possible to preserve both at the same time.

The Illinois Natural Areas Preservation Act (525 ILCS 30/11; from ch. 105, par. 711 sec. 11) recognizes the status of cemeteries as burial grounds and protects the rights of continued "use of cemeteries for cemetery purposes." This includes the right of cemetery "maintenance, construction or preservation of markers, monuments or memorials, [and] the care of any burial space or the use of a burial space for an internment" for which there are "prior rights." The act declares cemeteries as "suitable for dedication" by the owner or cemetery authority and supports the protection of natural heritage values in the form of the native vegetation that has persisted since the time of settlement. Thus, for those cemeteries protected within the Illinois Nature Preserves System, cemetery preservation is not restricted to just the markers. Vegetation planted as part of the cemetery also should be preserved as part of the historic landscape. INPC is mandated to preserve both natural and cultural resources.

PRESERVATION AND MANAGEMENT CHALLENGES

While there is still a lot to learn about the impact of accepted prairie maintenance practices on fragile marble burial markers, there are a few things known about markers preserved within prairie and savanna vegetation. This vegetation typifies the landscape that was familiar to the pioneers buried in the cemetery. At the time pioneer cemeteries were established, "cemeteries were places where the beauty of unspoiled nature became part of the memorial for the dead" (Seidel 2003). Many pioneer cemeteries were made accessible through site selection. Most were established on drier sites, especially atop hills or bluffs, where there were few trees and the grassland vegetation remained shorter than in the surrounding landscape. Cemeteries were burned, removing accumulated vegetation. In some cases livestock grazing was also used to maintain a lower vegetation height. Within the last 70 years, since the use of the lawn mower became widespread (american-lawns.com), careless mowing has resulted in permanent chips and scratches on many stone markers. Forested cemeteries and those with heavy accumulations of decaying plant materials are more prone to accelerated wear. It is known that excessive moisture creates major deterioration to these stones (Strangsted 1993). Cemetery prairies which are regularly burned, where brush and multiple years' worth of plant biomass are not allowed to accumulate, are drier and may be less prone to deterioration from excessive moisture (Moorehouse and Hassen 2006).

Much of the discussion over how to maintain vegetation within pioneer cemeteries relates to access and aesthetics. Nearly everyone would agree that walking through a blackberry bramble patch is not pleasant. Land-use changes, and perhaps the microclimate related to those changes, have allowed for the invasion of unwanted plants in many of the cemetery prairies. Records within the Illinois Department of Natural Resources (IDNR) Natural Heritage Database and INPC files show key conservative species such as *Cirsium hillii* and *Lilium philadelphicum* have disappeared from several cemetery prairie natural areas as degradation by woody and weedy invaders has expanded (Illinois Department of Natural Resources, Illinois Nature Preserves Commission, Nuzzo 1985 and 1986). As blackberries, sumac, dogwood, and other woody invaders are clonal (spread by massive underground root systems), they are difficult to control and often annual fire is not sufficient to create suitable conditions for access. In such circumstances, mowed paths may be the best solution to facilitate visitation.

Creating aesthetics which are pleasing to all is difficult. Once the access issue is resolved, and either the brush is controlled or mowed paths are maintained, aesthetics become less important. Unwanted brush continues to be the single biggest problem for creating an attractive appearance within a cemetery natural area. As most cemeteries are visited in the spring, especially on Memorial Day, a variety of prairie and savanna wildflowers with various hues of white, pink, yellow, and blue are appealing to many visitors.

Often management conflicts over maintenance of vegetation within a cemetery come down to varying goals. Whose opinion shall take precedence? Does the opinion of the local community and those who visit the cemetery, especially those with ancestors buried in the cemetery, have greater influence than those who must also protect the valued plant community, and vice versa? It is important to listen to all sides. Part of the solution is through education, whereby the local community understands both the original character of the environmental landscape at the time the cemetery was created and the goal of the INPC, which is to seek a balance between the mandates to preserve both natural and cultural resources. INPC and IDNR have done extensive work to document the cultural history and develop management plans for the protection of cultural resources for many cemeteries within the INPC system and cemeteries owned by IDNR. Sharing information retrieved from inscriptions on the burial markers as well as maps and photographs of the stones can be greatly appreciated. Doing so also helps INPC and IDNR to maintain a record of the condition of the stones and allows us to monitor changes which may be occurring as a result of our management activities. Assisting visitors in searching for the graves of ancestors buried in these sites and allowing for better access through temporary or permanent pathways into the cemetery are also helpful.

Once a cemetery is properly documented, the next step is to restore the grave markers to their original position and condition, to the extent possible, while preserving their historical value. The Illinois Historic Preservation Agency (IHPA) is responsible for the protection of unregistered cemeteries more than 100-years-old as well as graves not located within cemeteries (Hassen and Cobb 2008). The Human Skeletal Remains Protection Act (20 ILCS 3440; 17 IAC 4170) protects all unregistered graves, grave artifacts, and grave markers. The act requires that persons receive a permit from IHPA before they can "disturb or allow the disturbance of human skeletal remains, grave artifacts or grave markers." IDNR in cooperation with IHPA has been offering workshops, led by certified professionals, to train participants on proper restoration techniques. IDNR and IHPA have published a valuable tool for cemetery preservation titled "Illinois Historic Cemetery Preservation Handbook: A Guide to Basic Preservation" (Hassen and Cobb 2008).

INTRODUCTION

CEMETERY NATURAL AREAS IN ILLINOIS

INPC is charged with the responsibility of preserving "adequate examples of all significant types of natural features found in the state" within the Nature Preserves System. These "natural features include landforms and geological formations, soils, streams and lakes, terrestrial and aquatic communities of plant and animals, and archaeological sites" (Illinois Nature Preserves Commission 1972). INPC guidelines equate archaeological sites with "important cultural assets," or what we currently refer to as cultural heritage resources, including pioneer cemeteries as well as Native American settlements and burial mounds, all of which are protected within the Illinois Nature Preserves System. Dealing with the cultural sensitivity of a hallowed cemetery is essential. However, equally important to the goals of the INPC is preserving the diversity of high-quality prairie flora and fauna that exist within these protected sites. Maintaining the on diversity of conservative prairie flora, especially on the outer edges of these tiny cemetery natural areas, is a battle we are losing (John Taft, personal communication; Illinois Nature Preserves Commission).

Out of 28 hectares (69 acres) identified by the INAI as pioneer cemetery natural areas (Illinois Department of Natural Resources), 17.43 hectares (43 acres) are protected with the Illinois Nature Preserves System (Illinois Nature Preserves Commission). These protected sites receive management to preserve the historic vegetation, yet the diversity and quality of the prairie and savanna plant communities continue to decline, most likely attributable to multiple factors including small size, woody encroachment, exotic species, and herbicide drift (Phillippe et al. 2010, Corbett and Corbett 2002). The ability of a small cemetery natural area to maintain a high-quality floristic community does not appear to be related to any single management practice or timing its or frequency over the past decade.

METHODS

Plant species data were collected for all 29 pioneer cemetery natural areas in Illinois (Figure 1). Seventeen cemeteries, for each of which there were a minimum of 55 documented plant species, were analyzed. The 17 cemetery plant lists were compared to consider the following factors: size of natural community, location within the landscape, management type and frequency, disturbance (mowing), edge effect, and soil or community type. Information was obtained on the community classification for all 29 INAI cemetery natural areas and notable plant preferences for community types were noted.

RESULTS

Of these 17 cemeteries analyzed (Table 1), an average of 103 vascular plant species was reported. There was a cumulative total of 525 plant species, 90% of which were native plants. The top natives reported from the majority of cemetery natural areas included *Amorpha canescens, Andropogon gerardii, Ceanothus americanus, Euphorbia corollata, Rosa carolina, Schizachyrium scoparium,* and *Sorghastrum nutans.*

The most frequently documented invasives were *Ambrosia artemisiifolia, A. trifida, Helianthus grosseserratus, Prunus serotina, Rhus glabra, Rubus allegheniensis,* and *Solidago canadensis.* The top invasive exotics included *Achillea millefolium, Daucus carota, Melilotus alba, Morus alba, Pastinaca sativa,* and *Poa pratensis.* Of the remaining 10% of non-natives reported from the 17 sites, 10-15% of these species were introduced as ornamental cemetery plantings. The most frequently occurring ornamentals were *Euphorbia cyparissias, Hemerocallis fulva, Iris germanica, Ornithogalum umbellatum, Picea alba, Pinus strobus,* and *Syringa vulgaris.*

Floristic Quality Indices (FQI) (Taft et al. 1997) ranged from 16 to 45; an FQI value of 45 or greater generally indicates a high-quality natural area; a value over 35 indicates a site which may have regional significance; a value over 20 is thought to be degraded with some potential for recovery. The FQI values of those cemeteries analyzed ranged as low as 15.6; for cemeteries protected within the Illinois Nature Preserves System the range was from 21 to 45 (Table 2). The range of mean C for all plants and the range of mean C for all natives recorded from each cemetery varied greatly. This was partly due to the under-reporting of non-native plants at some sites.

Figure 1. Map of cemetery prairie/savanna natural areas in Illinois.

In comparing all 29 cemetery natural areas found in Illinois (Table 1), the community types broke down thusly: 17 mesic black soil prairies, 7 savannas, 3 hill prairies, and 2 sand prairies. As expected, conservative plant species were found at larger sites (> 2 hectares; > 5 acres). *Schizachyrium scoparium* was found at every site. *Euphorbia cyparissias* was found in both sand prairies. *Bromus inermis, Pastinaca sativa, Poa pratensis, Solidago canadensis,* and *Helianthus grosseserratus* were frequent in mesic prairies. Sporobolus heterolepis occurred in most hill prairies and sand prairies. *Amorpha canescens* and *Lithospermum canescens* were commonly reported in all communities except savannas. *Ceanothus americanus, Sorghastrum nutans,* and *Euphorbia corollata* were common in all but hill prairie communities.

The average size of these cemeteries is 1.07 hectares (2.4 acres); range 0.2-3.84 hectares (0.5-9.5 acres). Those species largely restricted to larger cemeteries (> 2 hectares; > 5 acres) included conservative prairie forbs: *Aster laevis, Commandra umbellata,*

Table 1. Illinois pioneer cemetery natural areas plant community data.

SITE NAME	COUNTY	HECTARES PROTECTED	HECTARES INAI	FQI	NO. PLANT SPECIES	MEAN C ALL	MEAN C NATIVES	NO. EXOTICS
Weston Cemetery Prairie	McLean	2.02	2.02	38	116	3.5	4.1	16
Beach Cemetery Prairie	Ogle	1.01	1.01	45	59	5.8	5.8	0
Prospect Cemetery Prairie	Ford	2.02	2.02	33	130	3.4	3.4	32
Brownlee Cemetery Prairie	Mercer	1.21	0.57	41	120	4	4.5	14
Loda Cemetery Prairie	Iroquois	1.38	2.02	31	103	3.4	4.1	19
Temperance Hill Cemetery Prairie	Lee	0.32	0.4		2			N/A
Munson Township Cemetery Prairie	Henry	2.02	2.02	43	160	3.7	4.4	25
Spring Grove Cemetery Prairie	Warren	0.45	0.45	34	122	3.4	4.2	23
Brookville Lutheran Cemetery Prairie	Carroll	0.3	0.2	30	57	4	4.1	1
Tomlinson Pioneer Cemetery Prairie	Champaign	0.4	0.4		32			8
Fairchild Cemetery Prairie	Vermillion	0.56	0.61		6			N/A
Roberts Cemetery Prairie	Montgomery	0.69	0.65		9			N/A
Byler Cemetery Savanna	Adams	0.4	0.4	34	94	3.7	4	7
Greenlee Cemetery Prairie	Henry	0.49	0.49	27	71	3.6	4.5	14
Hetzler Cemetery Prairie	Bureau	0.4	0.53		35			0
Short Pioneer Cemetery Prairie	Grundy	0.53	0.53	31	169	2.8	3.6	43
Mt. Palatine Cemetery Prairie	Putnam	0.61	0.61		10			N/A
Root Cemetery Savanna	Peoria	1.01	1.01	35	129	3.3	3.7	15
Vermont Cemetery Prairie	Will	0.4	0.4		N/A			N/A
Voight Pauper Cemetery Prairie	LaSalle	0.4	0.2	21	69	3	4.2	20
Chandlerville Cemetery Hill Prairie	Cass	0.81	0.85		N/A			N/A
Scotch Cemetery Prairie	Knox		0.69	41	109	4.1	4.5	11
St. Mary Cemetery Hill Prairie	Peoria		1.29	37	135	3.5	4.2	22
Springdale Cemetery Hill Prairie	Peoria		2.43	38	160	3.2	3.6	20
Afton Cemetery Prairie	DeKalb		3.84		N/A			N/A
Clyde Cemetery Prairie	Whiteside		0.36	27	55	3.9	4.4	6
Sandy Town Cemetery Prairie	Whiteside		0.81	16	56	2.4	3.3	15
Pellsville Cemetery Prairie	Vermillion		0.45		N/A			N/A
Winchester Cemetery Prairie	Scott		0.81	25	60	3.5	3.9	8
TOTAL		17.43	28.07					

Eryngium yuccifolium, Gentiana puberulenta, and *Potentilla arguta.* Plants commonly found on small sites (< 0.8 hectares; < 2 acres) included weedy forb species: *Heliopsis helianthoides, Lespedeza capitata, Monarda fistulosa, Physalis heterophylla, Rudbeckia hirta,* and *Solidago juncea.* Urban cemeteries contained more exotics, such as *Alliaria petiolata, Robinia pseudoacacia, Poa pratensis,* and *Rosa multiflora*; however, surprisingly, ornamentals were not more prevalent within cemeteries in urban environments versus those in rural areas. Rural cemeteries adjacent to cropland had more weedy natives such as *Apocynum cannabinum, Helianthus grosseserratus, Ratibida pinnata,* and *Verbena stricta.* Sites near woodlands or pasture had more shrubs and vines.

Exotics and weedy species such as *Achillea millefolium, Bromus inermis, Convolvulus arvensis, Morus alba, Panicum virgatum,* and *Poa pratensis* as well as conservative natives, such as *Echinacea pallida, Eryngium yuccifolium,* and *Viola pedatifida* were more common in sites which were frequently burned. Those plants more common in rarely burned sites included taller grasses and savanna trees and shrubs: *Andropogon gerardii, Corylus americana, Prunus serotina, Rubus* spp., *Salix humilis,* and *Sorghastrum nutans.*

Sites with large mowed areas were more likely to have rhizomatous shrubs such as *Rubus* spp. and *Salix humilis.* Nuzzo (1985) noted that at Munson Township Cemetery Prairie Nature Preserve in Henry County, the areas near paths that were frequently mowed had the greatest abundance of woody and herbaceous invasive plants as compared to areas where no mowing occurred. This observation was also made by the author at Scotch Cemetery Prairie in Knox County (Illinois Nature Preserves Commission). Curiously, disturbance species such as *Poa pratensis* and *Melilotus alba* were also abundant in unmowed cemeteries. As mowing often occurs frequently along roadsides adjacent to many of these cemeteries, the impact of mowing within the cemeteries may no longer be notable.

DISCUSSION

Analysis of the plant species data revealed an astonishing diversity of plants documented within cemetery natural areas statewide. In spite of losses to diversity and natural community quality, cemetery natural areas remain critical sources of genetic diversity (Corbett and Corbett 2002) and are excellent sources of seed for restoration projects. While some cemetery natural areas remain highly diverse, the actual size

Table 2. Floristic quality comparisons between select cemetery natural areas in Illinois.

FQI	CEMETERY NAME	MEAN C ALL	CEMETERY NAME	MEAN C NATIVES	CEMETERY NAME
45.0	Beach	5.9	Beach	5.7	Beach
42.8	Munson Township	4.1	Scotch	4.5	Prospect
40.9	Brownlee	4.0	Brownlee	4.5	Brownlee
40.2	Scotch	4.0	Brookville Lutheran	4.5	Greenlee
37.7	Weston	3.9	Clyde	4.5	Scotch
37.6	Springdale	3.7	Munson Township	4.4	Munson Township
37.3	St. Mary	3.7	Byler	4.4	Clyde
35.1	Root	3.6	Greenlee	4.2	St. Mary
34.2	Byler	3.5	Weston	4.2	Spring Grove
33.8	Spring Grove	3.5	St. Mary	4.2	Voight Pauper
33.3	Prospect	3.5	Winchester	4.1	Weston
31.0	Loda	3.4	Loda	4.1	Loda
30.1	Brookville Lutheran	3.4	Prospect	4.0	Brookville Lutheran
30.0	Short Pioneer	3.4	Spring Grove	4.0	Byler
27.2	Clyde	3.3	Root	3.9	Winchester
27.1	Greenlee	3.2	Springdale	3.7	Root
25.0	Winchester	3.0	Voight Pauper	3.6	Short Pioneer
20.9	Voight Pauper	2.8	Short Pioneer	3.6	Springdale

of the high-quality areas is decreasing; the edges are dominated by weedy plant species (Illinois Department of Natural Resources; Illinois Nature Preserves Commission; Nuzzo 1985 and 1986; Taft, personal communication). Management on cemetery natural areas across Illinois has been inconsistent, often due to controversy over maintenance of the burial sites. Better methods of monitoring the community dynamics of these small sites will be necessary to document and determine the best long-term management solutions to halt and reverse the trend of declining floristic quality while also protecting the cultural heritage of the cemeteries.

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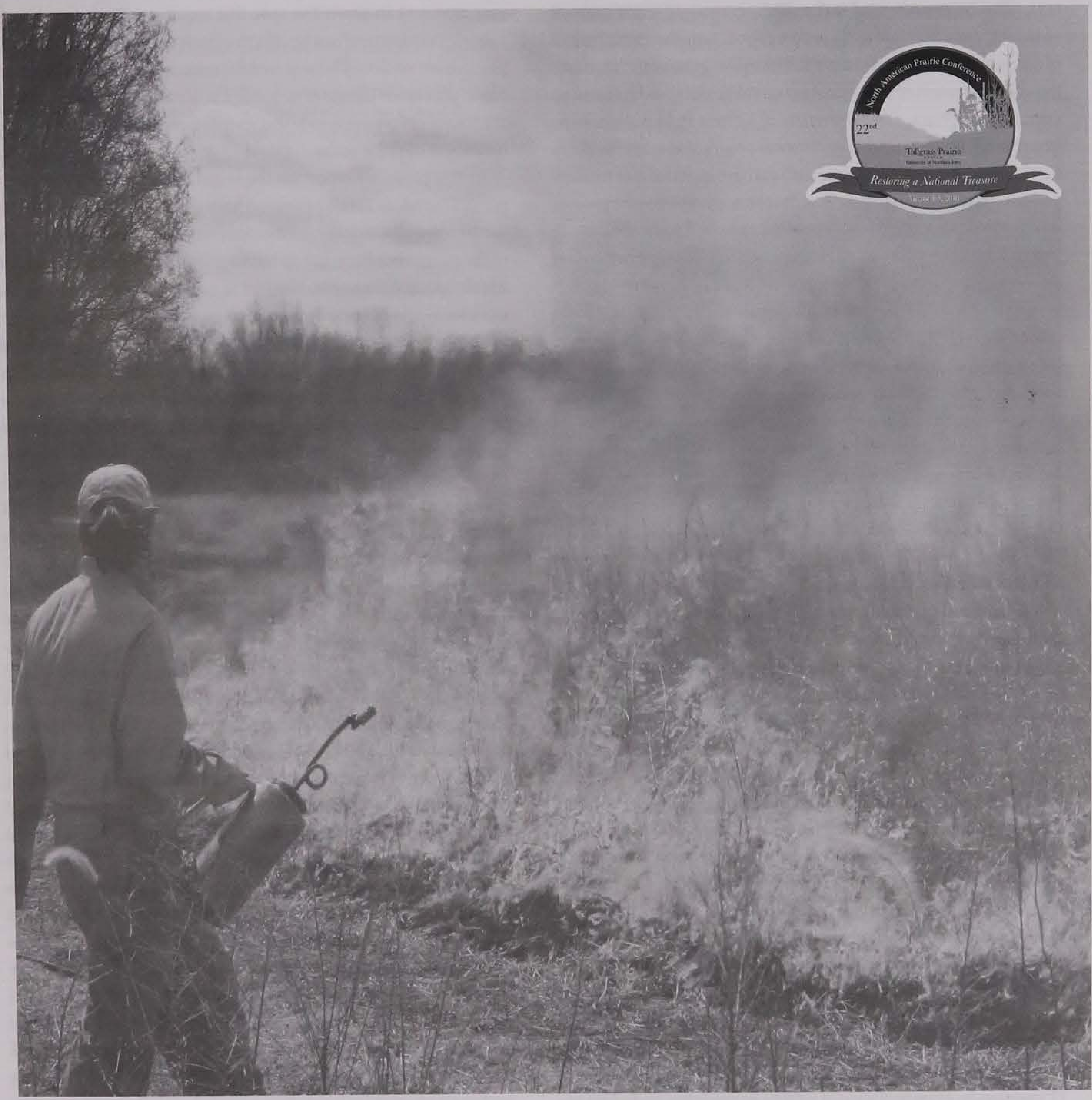

RESPONSE OF HERBACEOUS PLANT COMMUNITY AFTER REMOVAL OF WOODY VEGETATION IN A TALLGRASS PRAIRIE RESTORATION

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Abstract: One of the major mechanisms of change in the North American prairie landscape has been an increase of woody vegetation. This increase has led to the disappearance of a vast amount of habitat once dominated by native grasses and the resulting loss of native biodiversity. Conservation of native grassland communities and restoration of functional grassland landscapes are therefore critically important. The University of Kansas Field Station is engaged in a long-term prairie restoration program at a 293-ha site in northeastern Kansas. Restoration efforts have included tree removal and prescribed burning to increase the connectivity of native prairies and to encourage expansion of native vegetation. Landscape metrics were calculated to determine changes in the landscape before and after restoration activities. Permanent vegetation monitoring plots were sampled in 2005, prior to restoration activities, and again in 2008 and 2010 to assess changes in plant community composition. Our surveys in 2010 showed greater grassland habitat size and connectivity in this historic prairie-forest landscape and an increase in warm-season native grasses and native forbs. Both non-native forbs and cool-season grasses have declined, and woody species have not recolonized the areas surveyed.

Key Words/Search Terms: prairie restoration, prescribed fire, woody invasion

INTRODUCTION

The tallgrass prairie that once dominated the Great Plains has been greatly diminished in the last two centuries (Sampson and Knopf 1994), and in northeastern Kansas less than 1% of the high-quality native prairie remains (Kindscher et al. 2005). This is a result of human occupation that has altered the landscape through activities such as plowing, grazing of domestic livestock, introduction of non-native species, and widespread suppression of fire (Whitney 1994, Kettle et al. 2000, Briggs et al. 2002b, Briggs et al. 2005). One of the major mechanisms of change has been an increase of woody vegetation, which is known to invade and outcompete native prairie plant communities in the absence of fire, mowing, or other management (Fitch and Hall 1978, Kettle et al. 2000, Briggs et al. 2002b, Heisler et al. 2003). Woody encroachment into grasslands is thus a major cause of the loss of native biodiversity and alteration of ecosystem function (Briggs et al. 2002a, Jackson et al. 2002, Heisler et al. 2004, Briggs et al. 2005, Lett and Knapp 2005, Knapp et al. 2008, McKinley and Blair 2008). In addition to this direct loss, the few prairies that remain have become increasingly isolated, often through expansion of woodlands. Fragmentation of native communities, including native prairie, is associated with a host of negative outcomes (Wilcox and Murphy 1985, Fahrig 2003). Grasslands heavily invaded by trees may represent an alternative state that requires perturbation before historic management can be effective (Suding et al. 2004); for example, tree removal may be required for restoring fire management in mesic grasslands (Briggs et al. 2005). Thus, restoration efforts aimed at removal of invasive woody species may benefit tallgrass prairie restoration at multiple scales.

In this paper, we report on the initial four years of a long-term project to restore native prairie on a site in northeast Kansas. This site had been historically dominated by tallgrass prairie, but much of the prairie has been lost to both agriculture and the invasion of woody species (Fitch 1965, Kettle et al. 2000). Restoration methods included cutting of woody vegetation and prescribed burning. The preliminary restoration goals were to (1) increase the expanse and connectivity of native grasslands, and (2) increase native, herbaceous plant cover in degraded areas. Landscape metrics, including connectivity and expanse of grassland parcels, were established prior to and after removal of woody vegetation by GIS analysis of aerial photography. Vegetation monitoring plots that were established and surveyed prior to restoration activities will remain for long-term monitoring of changes in woody, exotic, and native vegetation cover. In this paper, we report on the impact of the removal of woody vegetation on landscape features and on herbaceous vegetation in the first years following restoration activities.

METHODS

STUDY SITE AND RESTORATION MANAGEMENT

This project was conducted on a 293-ha portion of the University of Kansas Field Station in northeast Kansas (Jefferson County) (Figure 1). Prior to European settlement in the mid 1850s, the area was dominated by tallgrass prairie but contained some upland oak-hickory stands (Fitch 1965, Kettle et al. 2000). The site contains a variety of grassland habitats including small isolated prairie remnants, lands reseeded to prairie in 1957 and undergoing different treatments (Fitch and Hall 1978, Kettle et al. 2000), lands seeded to Conservation Reserve Program (CRP) plantings in 1988, and a mix of other grasslands varying in their degree of native character. Trees and shrubs had become established at various densities and assorted spatial configurations across the site (Figures 2, 3, 4). Woody vegetation had been out-competing herbaceous prairie plants and reducing connectivity among grassland sites. For these reasons an aggressive program of restoration was undertaken in 2005.

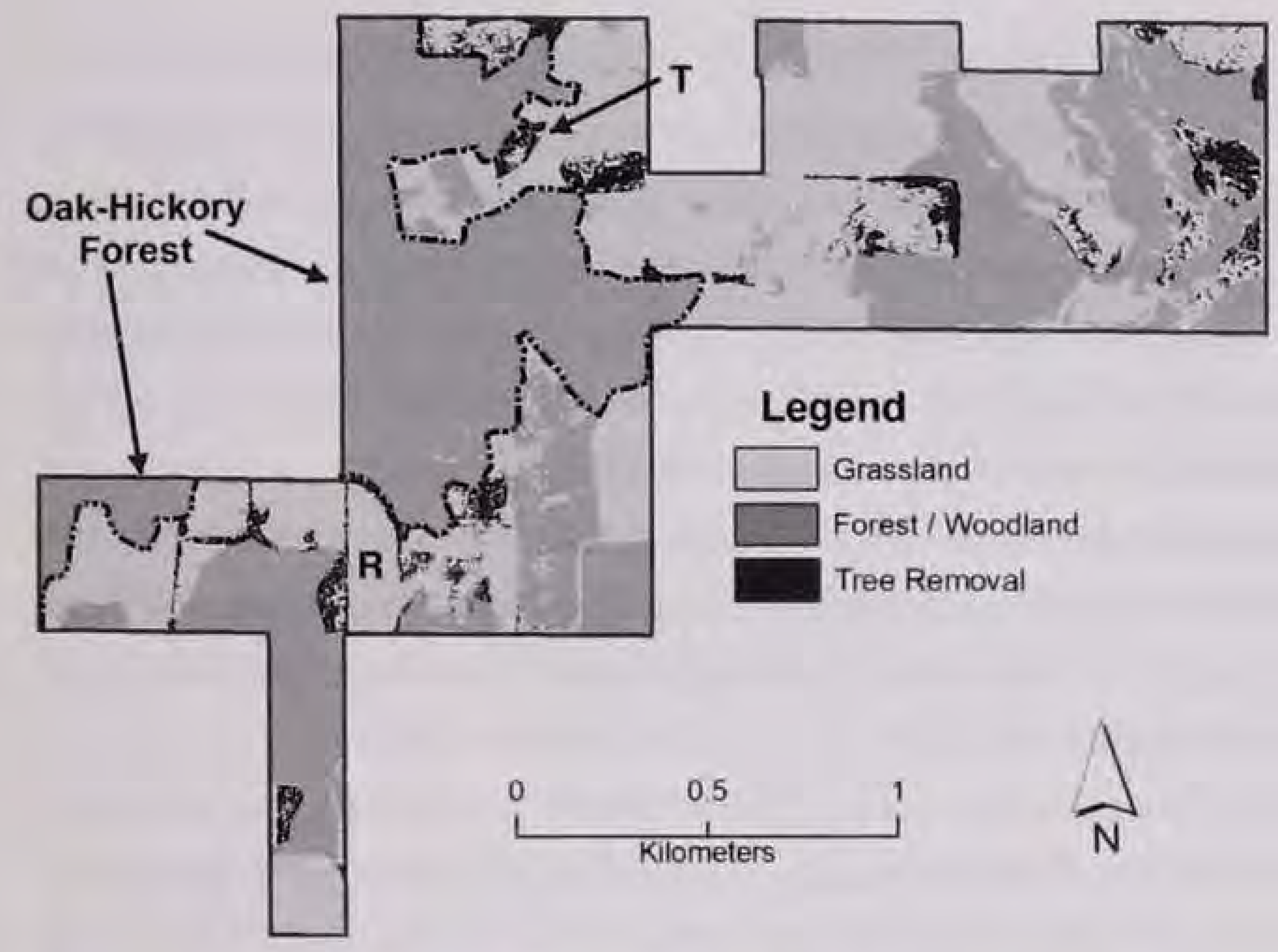

Figure 1. Map of the 293-ha restoration site showing areas where woody vegetation was removed, and locations of the 4-ha Rockefeller Native Prairie (R) and the Tomanthera Prairie (T; point of arrow indicates the centroid of the 1.6-ha prairie). Two areas of oak-hickory forest are identified on the figure; other closed canopy areas are successional woodlands that developed on former prairie areas.

The centerpiece of the restoration was planned as a 30-ha corridor of grasslands with the high-quality 4-ha Rockefeller Native Prairie (RNP) at its heart (Figures 1, 2). This continuous corridor of grassland habitat (Figure 2) connects former CRP land to the west of the RNP and prairie restoration units to the east of the RNP that had been seeded to prairie grasses in 1957. The RNP was managed as a hay meadow for decades, and from the late 1950s to 2005 by springtime burning at 1-3-year intervals (Kettle et al. 2000). It was bordered on the west by a treed fence line; the northern portion of this fence line separated the RNP from CRP lands planted in 1988 (Site 1, Figure 2) and the southern portion of the fence line separated RNP from an area dominated by exotic cool-season grasses (Site 2, Figure 2). On the east side of RNP, woody vegetation had developed on formerly tilled land that was seeded to warm-season grasses in 1957 (Fitch and Hall 1978, Kettle et al. 2000) (Site 3, Figure 2). The RNP contains ~200 native prairie species including two federally threatened plants: Mead's milkweed (*Asclepias meadii*) and western prairie fringed orchid (*Platanthera praeclara*). There is evidence at RNP that proximity to the native prairie results in greater colonization of seeded grasslands by native species (Campbell 1996, Kindscher and Tieszen 1998) and greater seed rain (Schott and Hamburg 1997). It is expected that the elimination of woody species in areas separating the RNP from planted grasslands will encourage movement of species from the native prairie into the seeded grasslands. To create the corridor, trees were removed from areas bordering the RNP, from treed fence lines separating CRP fields, and throughout the seeded land east of the RNP (Figures 1, 2).

Another native prairie at the site is the 1.6-ha Tomanthera Prairie (Figure 1), a site degraded by erosional gullies and invaded heavily by woody species, especially eastern red cedar (*Juniperus virginiana*) (Figure 4). This prairie is not known to have been plowed and has not been burned for at least 35 years. Although it was a degraded site, we anticipated that the limited understory of natives, or possibly some native species in the seed bank, would aid in recovery of plant diversity at the site. Restoration consisted of tree removal and prescribed burning.

Trees throughout the 293-ha site were cut using hydraulic shears mounted on skid loaders or by hand with chainsaws. Stumps of cut trees, except for eastern red cedar, were treated with herbicide (picloram) to prevent resprouting. Debris and cut material were moved and piled for burning. Trees were removed from the area surrounding the RNP during January 2006, from the Tomanthera Prairie in September 2006, and from the general 293-ha area between January 2006 and December 2009.

Prescribed burning (mid-March to mid-April) was used in conjunction with all areas of tree removal in the 293-ha restoration site during 2006-2010. For purposes of evaluating the data from the four vegetation monitoring locations (Figures 1, 2), Site 1 was burned in 2007, 2008, and 2010; Sites 2 and 3 were burned in 2006-2008 and 2010; and the Tomanthera Prairie was burned in 2009 and 2010.

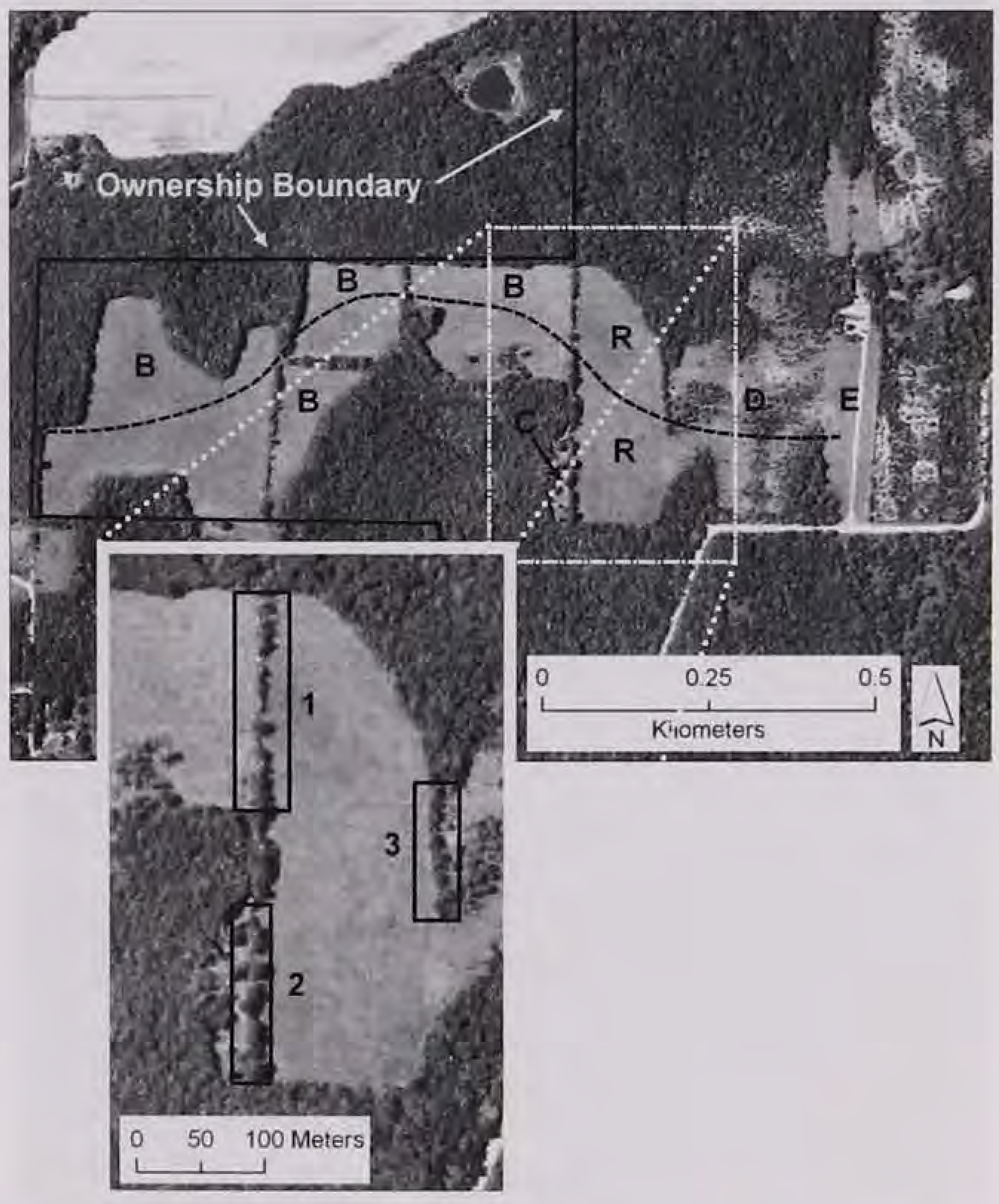

Figure 2. In 2005, before restoration, the Rockefeller Native Prairie (R) was bordered on the west by lands seeded in CRP in 1988 (B) and a small area dominated by exotic cool-season grasses (C). On the east, the native prairie was bordered by two areas seeded to warm-season grasses in 1957 (for details see Fitch and Hall 1978, Kettle et al. 2000): one area was unmanaged (D), and the other area has been mowed annually (E). Woody vegetation in patches and fencerows was removed between 2006 and 2009 (refer to Figure 1). Enlarged view shows general locations for sample sites for vegetation before and after tree removal included areas within the former fence line (Sites 1 and 2) and within the area seeded in 1957 (Site 3). Dashed line indicates a continuous 30-ha grassland corridor after removal of trees. The ownership boundary between the University of Kansas Field Station and private property is indicated. Base image is 2005 NAIP photography.

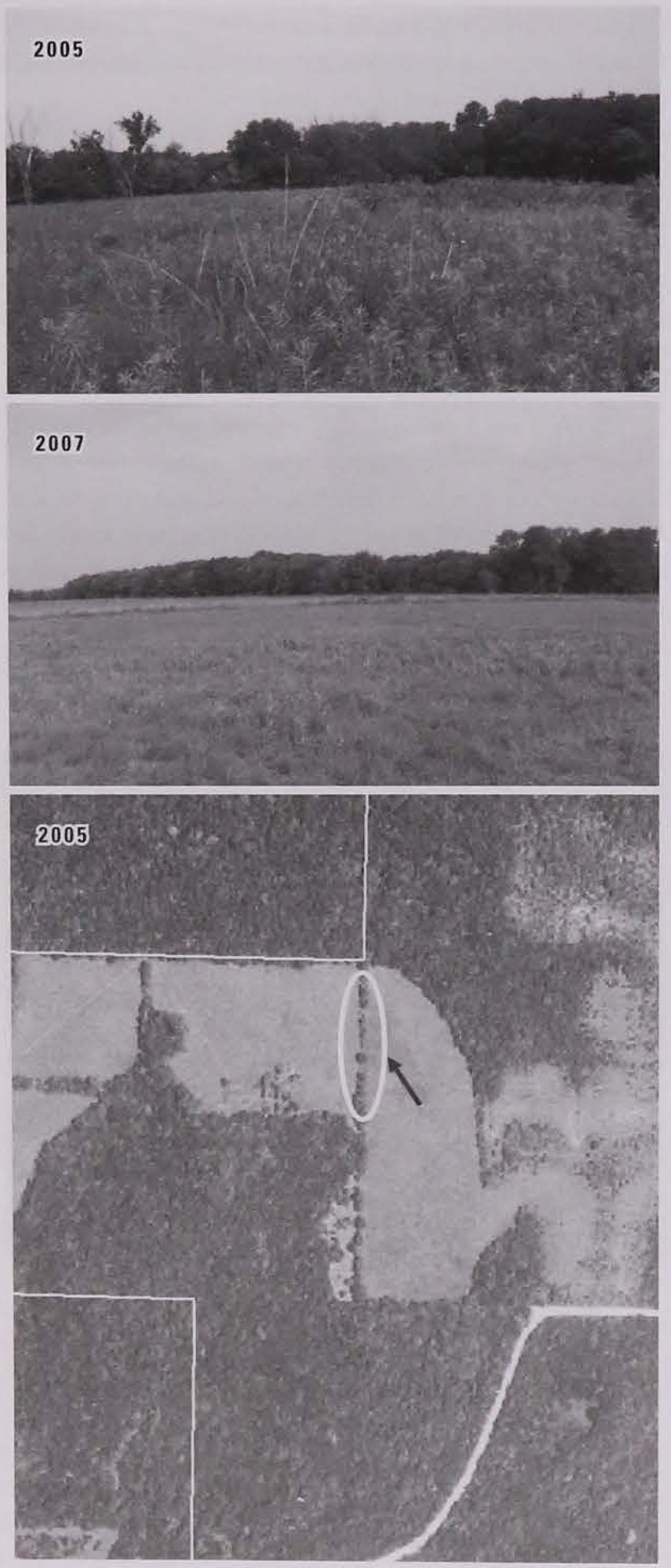

Figure 3. Ground-level repeat photographs taken from the same location in 2005 and 2007 show results of removing trees in the fence line on the west side of the Rockefeller Native Prairie in 2006. The reference image provides an aerial view (2005 NAIP photograph), for orientation and scale refer to the area of tree removal (oval) shown in photograph series; the arrow base is the approximate location of the camera in the Rockefeller Native Prairie with the orientation to the northwest.

LANDSCAPE METRICS

We assessed physical changes in the site using GIS (geographic information system) analysis of aerial photography taken in 2005 (before restoration activities) and in 2008 (after the majority of restoration was completed). A few areas were cleared of trees after the 2008 imagery was taken, and for these we digitized the estimated extent of tree removal using aerial imagery and measurements taken in the field. Classified imagery from these sources was used to perform spatial pattern analysis for four landscape metrics (patch number, edge, shape, and connectivity) using FRAGSTATS (McGarigal and Marks 1995). The threshold distance for connectivity was set at 100 m. The analysis of landscape metrics summarized changes from pre-restoration (2005) to post-restoration (2009) across the entire 293-ha site.

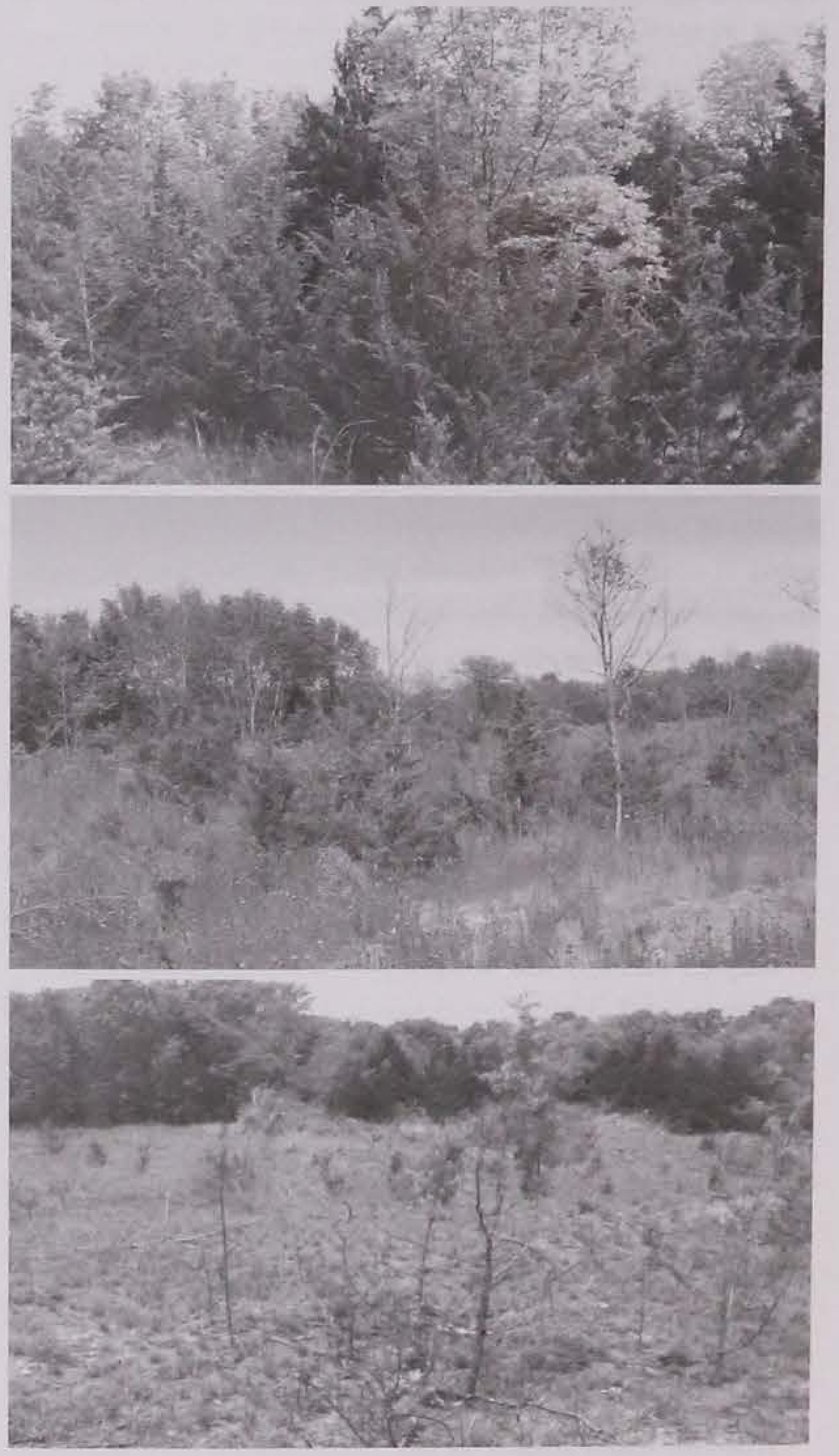

Figure 4. Tomanthera Prairie before and after restoration. Ground-based repeat photographs taken from the same location in 2005 and 2008 show the reduction in larger trees that were removed mechanically. The 2009 photograph was taken on May 15 following a prescribed burn on April 8. Many of the smaller red cedar trees that were not removed mechanically were killed by the fire.

VEGETATION MONITORING

To assess the impact of restoration activity (tree removal and prescribed burning) on herbaceous plants, we established seventy-two 5 x 5 m permanent sample plots in 2005 (before restoration). Plot corners were marked with iron bars, and GPS readings were taken to facilitate relocating plots in subsequent years. The presence and a visual estimate of the aerial percent cover of all plant species within each plot were recorded. Visual estimates of plant cover were based on a continuous scale and could exceed 100%. Plots were sampled in July of 2005 before tree removal, and at two times after tree removal in June of 2008 and June of 2010. For the present study, we report the results from 23 plots at four locations: one location at the Tomanthera Prairie (6 plots) and three locations adjacent to the RNP (referred to as RNP Sites 1, 2, and 3) (Figure 2). Six plots were monitored at RNP Sites 1 and 2, and five plots were monitored at RNP Site 3, although in 2010 only four of these plots could be located. Plots surrounding the RNP were placed linearly (along old treed fencerows) with distances between plots being equal. Plots in the Tomanthera Prairie were established to achieve uniform distribution across the site. To assess changes in vegetation following restoration activities, we compared the 2005 cover of native and exotic forbs, native and exotic grasses, and woody vegetation with that found in 2008 and 2010 across each study site. Data were transformed by taking the arcsine square root of the average absolute cover values for each species. A paired sample t-test was then used to detect significant differences in cover values from 2005 to 2010 (McDonald 2009). In addition to comparing cover values, we calculated and compared the average coefficient of conservatism (mean $C = \Sigma C/N$) (Freeman and Morse 2002) for each monitoring site as well as the Floristic Quality Index ($FQI = \text{mean } C \times \sqrt{N}$). We used a single factor ANOVA to detect any significant changes in the average coefficient of conservatism among monitoring years. Nonnative taxa were excluded from the calculation of mean C and the FQI. Data from all plots in each site were combined and analyzed using Microsoft Excel (2007) and PASW Statistics 18 (2009).

RESULTS

LANDSCAPE METRICS

Extensive mechanical removal of trees and shrubs increased use of prescribed burning in existing grasslands, and maintenance of recent tallgrass plantings resulted in an increase in grassland habitat size and connectivity from 2005 to 2010 (Figure 1 and Table 1). In total, 14.7 ha of trees (canopy cover) were removed across the 293-ha site, with 4.3 ha removed in the 30-ha restoration corridor centered on RNP, and 0.6 ha removed in the Tomanthera Prairie. Within the 293-ha site, the number of grassland patches declined by 45.6 percent (Table 1). The total amount of habitat edge in our restoration declined 38.3% (Table 1). The landscape shape index, a quantitative measure of landscape complexity, declined by 39.0% and the connectance index increased by 36.9% (Table 1).

VEGETATION MONITORING

Species richness in RNP Sites 1, 2, and 3 increased from 2005 to 2008 and then declined in 2010, although species richness was still greater in 2010 than in 2005 (Table 2). Only in RNP Site 1 did the average coefficient of conservatism increase from 2005 to 2010. No changes in the average coefficient of conservatism were significant ($P > 0.05$). The Floristic Quality Index doubled for Site 1 and improved only slightly for Site 3. Table 3 summarizes the changes in vegetation cover from 2005 to 2010 for each site. In RNP Site 1, the cover of native forbs increased while the cover of woody vegetation declined ($P < 0.01$). In RNP Site 2, the cover of woody vegetation significantly declined ($P < 0.01$) from 2005 to 2010. In RNP Site 3, both the cover of native warm-season grasses and native forbs significantly increased ($P < 0.05$) while woody cover significantly declined ($P < 0.01$). The dominant species in both 2005 and 2008 at RNP Sites 1 and 2 were, respectively, *Bromus inermis* (smooth brome) and *Bromus arvensis* (Japanese brome) (Table 4). By 2010, these cool-season grasses had been replaced with *Sorghastrum nutans* (Indiangrass), a native warm-season grass. RNP Site 3 was dominated by *Fraxinus pennsylvanica* (green ash) in 2005, *Sporobolus compositus* (composite dropseed) in 2008, and *Symphyotrichium praealtum* (willowleaf aster), in 2010.

Table 1. Summary of changes in landscape metrics for grassland and prairie before and after restoration.

METRIC	PRE-RESTORATION (2005)	POST-RESTORATION (2009)	% CHANGE
Number of patches	756	411	-45.6
Patch density (no./100 ha)	259.0	140.3	-45.6
Total edge (m)	94038	58036	-38.3
Edge density (m/ha)	320.3	197.7	-38.3
Landscape shape index	22.5	13.7	-39.0
Connectance index (percent)	3.46	4.72	36.9

Results at the Tomanthera site were similar to those at the RNP sites. Species richness increased from 2005 to 2008 and then declined in 2010, although species richness was still greater in 2010 than in 2005 (Table 2). The average coefficient of conservatism declined, but not significantly ($P > 0.05$) and the FQI increased from 2005 to 2010. The cover of native forbs and native warm-season grasses increased significantly ($P < 0.01$) from 2005 to 2010 (Table 3). In 2005 the Tomanthera Prairie was dominated by eastern red cedar with undergrowth consisting mostly of *Sorghastrum nutans*, *Liatris squarrosa* (hairy gayfeather), and *Schizachyrium scoparium* (little bluestem). After tree removal these undergrowth species declined with the native cool-season grass *Dichanthelium acuminatum* (pointed dichanthelium) becoming dominant. By 2010 *Ambrosia artemisiifolia* (common ragweed) (Table 4) was dominant.

Table 2. Species richness, average coefficient of conservatism, and Floristic Quality Index (FQI) for each study site in each monitoring year. Data from all plots in a study site were combined.

STUDY AREA	YEAR	SPECIES RICHNESS	AVERAGE COEFFICIENT OF CONSERVATISM	FQI
Rockefeller Native Prairie Site 1 (n=6)	2005	28	2.0	9.8
	2008	66	2.2	16.5
	2010	63	2.6	18.9
Rockefeller Native Prairie Site 2 (n=6)	2005	57	2.3	16.0
	2008	69	2.3	17.0
	2010	59	2.2	15.3
Rockefeller Native Prairie Site 3 (n=5, 2005 and 2008) (n=4, 2010)	2005	59	2.9	20.8
	2008	95	3.4	30.1
	2010	68	2.8	21.6
Tomanthera Prairie (n=6)	2005	44	3.2	20.2
	2008	81	2.9	25.0
	2010	73	3.0	24.0

DISCUSSION

All landscape metrics indicated a trend toward greater expanse and connectivity of prairie habitat following removal of woody vegetation. Reduction in the number of grassland patches by 45.6% resulted from the removal of woody vegetation. Reduction of edge by 38.3% indicates a simplification of landscape configuration, in this case by removing the edge produced by patches of woody vegetation. A decline in the landscape shape index by 39% indicates a reduction in the complexity of the landscape. Lastly, an increase in the connectance index for grassland by 36.9% indicates former barriers to grassland connectivity had been removed. The actual ecological significance and relevance of landscape metrics differs by species, ecosystem, and other variables, and interpretation can be difficult (McGarigal and Marks 1995; Turner et al. 2001). However, it is known that increasing or restoring connectivity can be a critical element of restorations (Suding 2004). For our purposes, the landscape metrics served as a coarse indicator of effectiveness in providing greater grassland expanse and connectivity (following restoration by eliminating patches of woody vegetation) and progress toward our goal.

As for our second goal, to increase native herbaceous plant cover in degraded areas, it appears that the removal of woody vegetation followed by annual (or nearly annual) burning for sites across the restoration area has produced the desired results. Four years after the initiation of restoration activities there has been an increase in native forbs and warm-season grasses, a decrease in nonnative forbs and cool-season grasses, and little recolonization of woody species. Invasions of woody species can reduce the biomass and diversity of herbaceous vegetation, and this is known to affect the effectiveness of fires by reduced fuel load under progressively greater woody stands (Briggs et al. 2002a, Heisler et al. 2004, Lett and Knapp 2005, Briggs et al. 2005). In our restoration, mechanical removal of trees, followed by increased herbaceous fuel load was required as an intervention before fire could operate as a historical disturbance; this

Table 3. Mean absolute cover values and relative cover values for six vegetation categories in each of the three monitoring years. Vegetation cover values for monitoring plots in each study site were combined.

STUDY AREA	VEGETATION TYPE	2005	2008	2010	2005	2008	2010
		RELATIVE COVER (%) (STANDARD ERROR)			MEAN ABSOLUTE COVER (%)		
Rockefeller Native Prairie Site 1 (n=6)	Native forbs	12.4 (4.9)	45.7 (15.5)	76.9[1] (12.1)	7	34	48
	Exotic forbs	0.4 (0.2)	0.8 (0.3)	0.3 (0.1)	0	0	0
	Woody vegetation	116.8 (16.1)	14.7 (2.9)	5.3[1] (1.8)	67	11	3
	Native warm-season grasses	0	10.5 (4.0)	64.8 (12.4)	0	8	41
	Native cool-season grasses	0	0.3 (0.2)	0.1 (0.1)	0	0	0
	Exotic cool-season grasses	44.4 (19.8)	64.0 (17.6)	12.2 (8.3)	26	47	8
Rockefeller Native Prairie Site 2 (n=6)	Native forbs	34.8 (16.3)	22.4 (6.6)	30.5 (9.9)	20	15	26
	Exotic forbs	16.7 (9.6)	2.3 (0.8)	0.3 (0.2)	10	2	0
	Woody vegetation	72.4 (3.7)	11.8 (5.1)	11.3[1] (4.4)	42	8	10
	Native warm-season grasses	2.4 (1.4)	9.2 (2.2)	48.1 (10.5)	1	6	42
	Native warm-season grasses	2.4 (1.4)	9.2 (2.2)	48.1 (10.5)	1	6	42
	Native cool-season grasses	0.4 (0.2)	1.5 (0.9)	0.3 (0.2)	0	1	0
	Exotic cool-season grasses	47.4 (18.9)	104.0 (3.4)	25.4 (11.3)	27	69	22
Rockefeller Native Prairie Site 3 (n=5, 2005 and 2008) (n=4, 2010)	Native forbs	38.2 (10.3)	65.0 (11.5)	82.9[1] (22.9)	21	40	53
	Exotic forbs	2.3 (1.3)	37.2 (11.8)	1.5 (0.8)	1	24	1
	Woody vegetation	126.0 (12.4)	10.5 (3.2)	6.8[1] (3.4)	68	7	5
	Native warm-season grasses	19.0 (9.1)	35.6 (9.1)	48.1[1] (20.7)	10	22	38
	Native cool-season grasses	0.3 (0.1)	3.8 (3.0)	0.4 (0.2)	0	2	0
	Exotic cool-season grasses	0.3 (0.1)	7.8 (3.5)	2.9 (1.5)	0	5	2
Tomanthera Prairie (n=6)	Native forbs	18.8 (3.8)	96.0 (12.5)	102.0[1] (12.6)	24	50	69
	Exotic forbs	0.2 (0.3)	4.2 (4.0)	0.4 (0.2)	0	2	0
	Woody vegetation	47.8 (18.0)	28.4 (9.7)	5.3 (2.1)	61	15	4
	Native warm-season grasses	10.8 (6.2)	27.7 (15.3)	28.9[1] (9.3)	14	15	20
	Native cool-season grasses	0.3 (0.1)	33.0 (12.8)	9.6 (3.0)	0	17	6
	Exotic cool-season grasses	0.1 (0.1)	0.8 (0.6)	1.2 (0.8)	0	0	1

[1] Difference in mean absolute cover values were significantly different (P<0.05) between 2005 and 2010.

also has been suggested as a necessary step for restoring grassland ecosystems (Briggs et al. 2005). Degraded systems of many kinds may require intervention before historical disturbances can be effectively implemented (Suding et al. 2004). It appears that our sites are recovering sufficiently well and will be able to support fire in the future. While these results are promising, we are still in the initial stages of a long-term restoration process; continued strategic monitoring of vegetation will serve to document the progress of the restoration and allow for adaptive management.

Plant responses to restoration activities are tracked to assess restoration progress and enable science-based adjustments in treatment levels. Potential applications of adaptive management include alternative mowing and burning schedules (including changing the seasonality and frequency of management), pacing the removal of trees and shrubs, and incorporating the results of research. We have gradually removed woody vegetation bordering grassland habitat. Monitoring the vegetation response to tree and shrub removal will allow us to determine a pace for tree removal. For example, a sharp increase in weedy or invasive species in the cleared (disturbed) areas could be detrimental to neighboring grassland parcels if undesirable species are not controlled. If we continue to observe recovery of native species following removal of invasive trees and shrubs, we will proceed in clearing larger areas of successional forest.

While the goals for the initial stage of the restoration project were to reduce woody cover and increase native forb and grass cover, we would like to take the restoration a step further and attempt to restore a more representative native vegetative community. This attempt will be aided by the presence of the Rockefeller Native Prairie, which can serve as a reference community. Future attention to species composition and diversity in comparison to the on-site reference community will continue to improve the ecological value of the restored areas as well as allow continued testing of restoration techniques. Recovery of species diversity is often limited by species availability, and seeding may increase species richness (Myers and Harms 2009), including areas for prairie restoration (Foster 2001, Foster and Tilman 2003, Foster et al. 2007). While seeding these areas with native forb species may ultimately be recommended, for the near future regular burning appears to be encouraging native warm-season grasses and reducing woody growth as desired.

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USE OF LOW-LEVEL AERIAL PHOTOGRAPHY FOR DELINEATION OF BIOLOGICAL AND PHYSICAL FEATURES OF TALLGRASS PRARIE

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Abstract: Delineating biological and physical features of large tracts of prairie communities is challenging, especially when annual monitoring is needed and major disturbances (natural or human-induced) are frequent. On the Fort Riley Military Installation in eastern Kansas, frequent disturbance is normal because of major training events that involve wheeled and tracked vehicles. Our objective was to document long-term disturbance trends to the tall grass prairie ecosystem. We used a low-level aerial photography system (a remotely controlled camera system suspended from a tethered blimp) to obtain high-resolution digital photographs to determine whether delineation of major plant forms (i.e., woody vs. nonwoody vegetation) and tracks created by military vehicles could be achieved for 1-ha (100 x 100 m) study plots. Field methodology developed included identifying placard/field marker design and placement to enhance geo-referencing, the efficient transportation of equipment to facilitate site-to-site transport of an inflated blimp, and demarcation of digital images from plot to plot without field review of images. Protocols for post-field processing of images refined included identifying > 1 quality image per plot, inspection for key features (e.g., tracks, vegetation patches, and placard placements), and file labeling to facilitate subsequent analysis and long-term archiving. Aerial photography may produce more-accurate results, be more efficient, and provide more information than a traditional ground-based sampling method can. The option to archive the digital photographs for future reference and evaluation—including for other features currently not considered—provides an added benefit not possible with nonphotographic ground-based sampling methods.

Key Word/Search Terms: aerial photography, disturbance assessment, vegetation monitoring, digital image processing

INTRODUCTION

Intensive monitoring of prairie landscapes can provide both insight into the dynamics of disturbance, restoration, and invasive species and input to management. Methods commonly used to document biological and physical features of prairies are ground-based and focus on assessing vegetation and bare ground at micro-scales (e.g., line transects of 25-, 50-, or 100 m length, plots of 0.25, 0.50, or 1.00 m2 size) (Stohlgren 2007). All are labor-intensive, seldom exceed 20 x 50m in plot size because of the time required to sample that large an area, and provide no assurance that the same exact area can be resampled with near 100% accuracy for followup surveys. Most are non-photographic. Another drawback to nonphotographic ground-based methods is that spatial characteristics of the landscape are not easily, documented. Recently, remote sensing (spaceborne) has become a viable option for obtaining information on landscapes, and provides an excellent spatial resolution in most instances (Miyamoto et al. 2004). However, remote sensing data generally require supporting field data to delineate vegetation communities or other features that are not well defined or ascertained from the images, especially if personnel evaluating photos are unfamiliar with the site. Thus, ground-truthing remains an important aspect of any such approach. High costs and scheduling challenges also exist. Scheduling is likely to remain the most problematic because the timing of a disturbance is often unpredictable, and accessibility to satellites in position to take images on short notice is limited.

Low-level aerial photography systems (LLAPS) offer flexibility of scheduling, relatively low costs, and high-quality digital images, in comparison to remote sensing and ground-based approaches (Caylor 2000, Aber 2004). These tethered systems are significantly less expensive to operate and lack the high-frequency vibration associated with camera units mounted to helicopters and fixed-wing aircraft. Capturing current conditions in just a few hours field time is possible. Regardless of the LLAPS system used, delineation and classification of vegetation (Miyamoto et al. 2004, Aber et al. 2006), restoration/reclamation of damaged military maneuver areas, and bare ground (personal observation), invasive plant populations (Blumenthal et al. 2007), or open water conditions (Aber et al. 2006) are possible from such high-resolution images. The option to archive the digital photographs for future reference and evaluation—including for other features currently not considered—provides an added benefit not possible with nonphotographic ground-based sampling methods. In addition to data (e.g., percent of area in vegetation, bare ground, open water, tracked, etc.) that can be obtained from photographs, the photographs are usually informative in and of themselves, regardless of the level of training or expertise of personnel. This includes pictures acquired in both vertical and oblique vantages (Aber et al. 1999).

Accurate mapping from digital photographs using a low-level aerial photography system can be achieved by placing survey markers (i.e, ground-control points) within or on the boundary of the area of interest (Aber et al. 1999, Rocchini and Di Rita 2005). Survey marker coordinates acquired with high-performance submeter GPS units can provide precise data necessary for geo-rectification (hereafter referred to as rectification) (Novak 1992, Seang and Mund 2006). Coordinate data is also useful for revisiting sites to take additional photographs and can eliminate the need for placement of pins or stakes that are subject to removal or displacement.

Herein, we describe the LLAPS equipment, field protocols, and post-acquisition processing of high-quality digital photographs that we have used to obtain quantitative data that are relatively inexpensive, repeatable, and useful for documenting disturbance and landscape features at one or more times. The equipment recommendations and protocols developed are based on more than 125 days of operating a LLAPS over a seven-year period under a variety of field conditions.

METHODS AND MATERIALS

EQUIPMENT: LLAPS COMPONENTS

A tethered LLAPS consists of four basic components: the camera unit, the camera platform, the aerial support system, and a remote control (R/C) unit with transmitter and video monitor (Figure 1). The support system can be a kite (Aber et al. 1999), a blimp (Murden and Risenhoover 2000) or balloon (Miyamoto et al. 2004). Blimps and balloons are nonrigid airships with their shape maintained by internal gas pressure. Complete systems are commercially available (2010 pricing: $3,500-$6,500), and we generally recommend purchasing them rather than individual components from multiple vendors because complete off-the-shelf systems minimize both the need to combine and test components to maximize system efficiency and the need for project personnel to be experts in electronics and engineering, thus allowing the focus to be on application rather than development of the system. A limited number of companies sell complete systems; they can be identified through Internet search engines using terms such as "aerial photography systems," "blimp photography systems," or "balloon aerial photography systems." Considerations for individual components are discussed below.

Figure 1. Major components of low-level aerial photography systems. Left: tethered lift platform (blimp). Center: camera platform. Right: ground-based remote-control unit with monitor.

CAMERAS

Camera designs suitable for aerial photography range from inexpensive digital formats (<$200) to moderately expensive (>$3,000) large-size digital SLR (single-lens reflex) units. Most provide high-resolution (> 10 megapixel) capabilities that ensure very good to excellent image quality. Depending on lift capabilities of the aerial support system, camera system weight might be a consideration, in which case point-and-shoot cameras (typically < 1.4 kg; 3 lbs) are the only option vs. mid- or large-sized digital units (typically 1.4kg; 3-7 lbs). Most commercial vendors offer a few choices of brands and models based on control designs that affect powering the camera, video output, and focus and zoom of the lens.

Most standard digital camera lenses provide sufficient quality. A wide-angle lens offers both advantages and disadvantages. If the blimp/balloon is capable of lifting the camera platform and tether line 122-152 m (400-500 ft.), as much as 1 ha (2.5 acres; 100 x 100 m) of area can easily be photographed within a picture frame using a standard lens. (Note: Federal Aviation Administration (FAA) regulations stipulate that moored balloons and kites not exceed an altitude of 152 m (500 ft.) without notification). If lift capacity limits the photography system to < 122 m (400 ft.) in altitude, then a wide-angle lens can ensure photos including up to 1 ha. The disadvantage to using a wide-angle, particularly for detailed analysis of landscape features, is that some distortion will be incorporated into the photo.

CAMERA PLATFORM

This component includes 1) the attachment equipment to the aerial support system, 2) the transmitter (for camera video) and receiver (for R/C-to-camera operation including zoom and trigger, pan, and tilt), and 3) batteries to operate noncamera electronics (Figure 1). The total weight of these components, along with the camera and of tether weight determines the payload, and can dictate the balloon, blimp, or kite size required to lift the unit to the necessary altitude. For high wind conditions, more tether line is required (up to 15%-20%) to launch a blimp, balloon, or kite than during relatively calm conditions. Therefore, weight of tether line is variable even if the desired altitude for taking photographs remains relatively constant from one photography session to the next.

Pan control (rotating the camera up to 360°) enables the operator to center photos along the long or short axis of the picture frame (Figure 2). Tilt control (positioning the camera straight down or at an angle providing a landscape view) permits varying the angle relative to the horizon.

Figure 2. Direct overhead photograph (left) vs. direct landscape photograph (right). View determined by pan and tilt controls based on video feed from camera to R/C unit.

AERIAL SUPPORT SYSTEM

The camera and camera platform units are held airborne using a tethered kite, blimp, or balloon. Blimps and balloons (Figure 3), which can generally carry heavier payloads and are more stable in windy conditions, are the focus of this paper. Altitude, ambient temperature, attack angle, payload, wind conditions, air density, and other factors determine the final lifting force available. The sizes of these nonrigid airship designs typically range from 3.7 to 6.1 m (12-20 ft.) in length, and they are filled with helium to provide lift. Airship materials usually consist of nylon, urethane, polyurethane, or some combination of these. Mounting patches on the belly allow for tether and camera/camera platform attachment (Figure 1, top center).

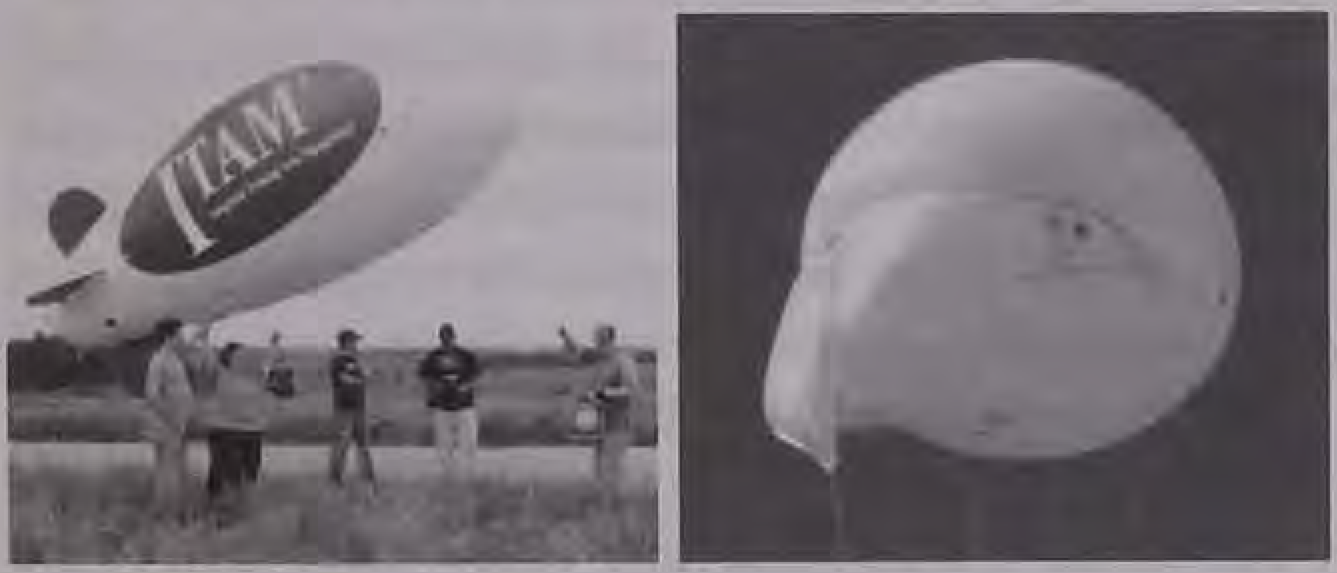

Figure 3. Aerial support systems. Blimp (left) shown is 5.9 x 1.8 m (18 x 6 ft.) at longest and widest points, respectively. Balloon (Aerostat Kingfisher™ model, right) is approximately 1.7 x 2.1 m (6.8 x 5.4 ft.). Both systems can support mid-sized SLR digital camera-equipped platforms.

Tether line is typically high-strength (500-3,000-pound test) nylon, polyester, or polyethylene fiber. Retrieval systems can be hand-cranked (Figure 4) or powered rewind reels; the former usually result in quicker retrieval and significantly less cost. Hand-cranked portable units also eliminate the need to drive vehicles into remote or difficult-to-access areas, thereby reducing the impact of sampling on the landscape. Hand-cranked units also allow for quicker repositioning of the blimp or balloon over the middle of the plot or area of interest during variable wind conditions.

REMOTE CONTROL UNIT

The R/C unit permits on-the-ground, wireless control of camera angle, focus, and triggering. These lightweight units' basic components include a transmitter and omnidirectional antenna to send signals to the camera platform receiver, a receiver for video feed from the camera, and a video monitor to display that video data (i.e., to see what the camera sees). Video monitors are best viewed with a hood to minimize glare (Figure 1, right).

Figure 4. Hand-crank reel used to release and retrieve tetherline to aerial platform.

SUPPLIES AND TRANSPORT EQUIPMENT

Helium in pressurized tanks is required for a blimp or balloon. Helium is an inert, colorless, odorless, tasteless, and nonflammable gas. In addition to being lightweight (seven times lighter than air), helium is relatively safe to transport and use. However, it can be hazardous when pressurized and acts as a suffocant if inhaled. Approximately 5.6-9.9 cubic meters (200-350 cubic ft.) of helium (typically 2-3 large tanks) is required for initial inflation. Depending largely on the status of the balloon or blimp (i.e., inflated, partially deflated, or uninflated) during transport, more helium may be required during an operational day. If the unit can be stored inflated (full or near-full), helium cost can be minimized when the unit is used over a 2-3-week period. However, even partially filled units need to have all gas/air evacuated periodically, because helium will be lost and to some degree replaced by oxygen, thereby reducing payload capacity. Using 2010 prices, a typical 18-foot blimp fully inflated would cost approximately $150 to inflate.

An inflated blimp or balloon can be transported from site-to-site in one of two basic ways: either stored in a box trailer or tied down to a specially designed rack atop a vehicle or small trailer (Figure 5). Box trailers can also serve as more perma-

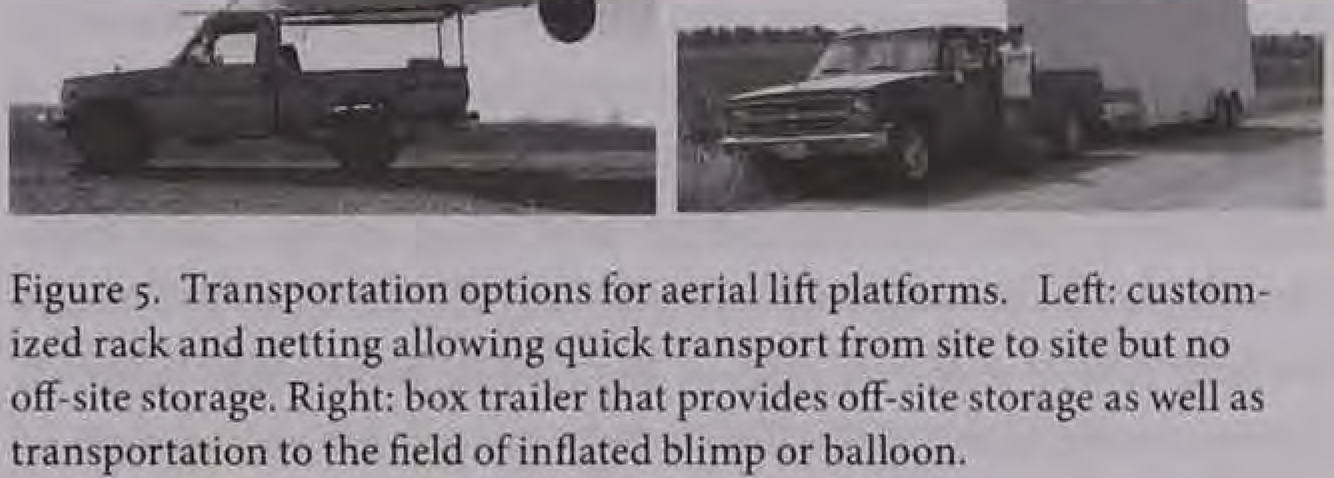

Figure 5. Transportation options for aerial lift platforms. Left: customized rack and netting allowing quick transport from site to site but no off-site storage. Right: box trailer that provides off-site storage as well as transportation to the field of inflated blimp or balloon.

nent storage but generally add cost (~$1,000-4,000) to start up expense, including the possible cost of acquiring/identifying a vehicle capable of pulling a large trailer.

Both the electronic components of the camera platform (including the camera) and the R/C unit are typically powered by rechargeable batteries. Fully charged batteries usually can permit 30-60 minutes of continuous operation. At least two complete sets of fully charged batteries should be available for any photography session and a third set is highly recommended. Although recharging of batteries in the field (using a converter unit connected to a vehicle's electrical system) is an option, that process may take >60 minutes per battery pack, thereby delaying resumption of the photography session past acceptable weather conditions. Backup batteries should be stored out of direct sunlight and ideally in a cool location (<24°C; 75°F) until needed.

Safe transport of the camera platform components and the R/C unit can be accomplished using a lightweight, solid-wall case (Figure 6). The case should be waterproof, crushproof, and dust-proof when closed. If room is available, battery packs may transported in the same unit. Special attention should be paid to avoid excessive bending, pinching, or twisting of wires or damaging of connectors during the packing and closure of the transport case.

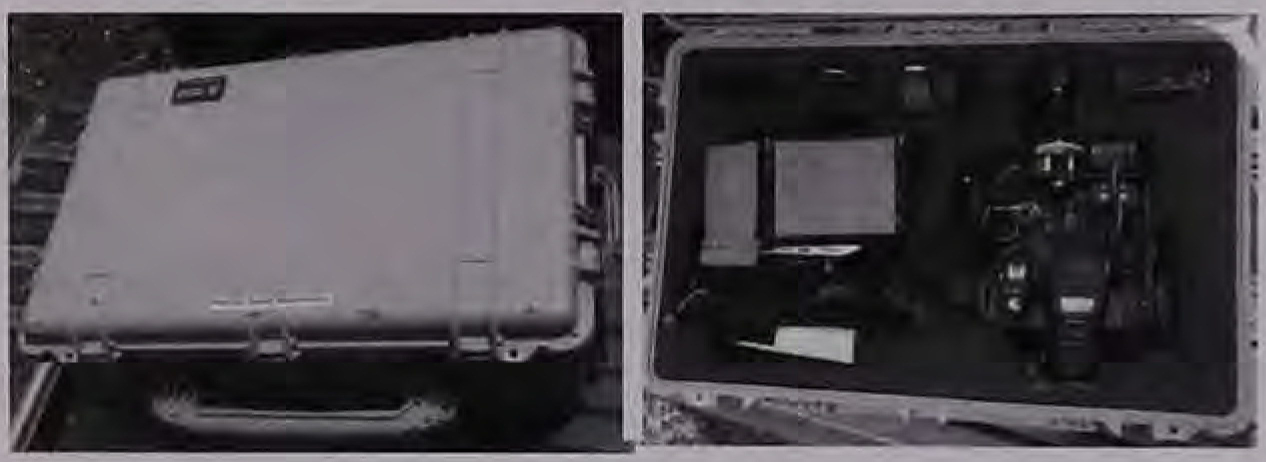

Figure 6. Solid-wall transport case used for storing and transporting camera platform, R/C unit, and batteries.

EQUIPMENT:
FIELD SURVEY MARKERS, GPS UNITS, AND RANGE FINDER

Points of reference (ground control points; GCP) on the landscape are important during a photography session because they aid the R/C unit operator's ability to position the blimp and therefore the camera platform over the area of interest. GCPs are also helpful in post-session processing and analysis of photos. If return visits to the site are anticipated, obtaining GPS coordinates during the initial photography session is also highly recommended.

Using survey markers (Figure 7) and obtaining GPS coordinates of those marked points are relatively simple. Markers can be used to outline predefined plots (Figure 8) or natural landscape features. Generally, survey markers need to be at least 20 x 20 cm (8 x 8 in), white, and easily set up in the field. Consideration for vegetation height may dictate that markers be anchored atop a tripod (Figure 7) or other adjustable support. Survey markers should be lightweight enough for easy transport into the field yet sturdy enough to withstand moderate winds at ground level to ensure they remain upright during photography sessions. A minimum of three survey markers per site should be positioned so as to appear in each photo and their locations recorded using a GPS unit. For square or rectangular plots, at least four survey markers (on the corners) should be properly positioned with perhaps a fifth centered within the plot. If rectification (Georgic and Wagner 2009) of digital aerial photos is projected, at least five survey markers should be positioned and GPSed (Figure 7). If only four GCPs are utilized, accurate spacing and evenness across the site are very important for any rectification/georeferencing procedures carried out. When more GCPs are used, one does not have to be as concerned about proper spacing. However, care should be taken not to position all GCPs in a straight line. Consideration for the roughness of the terrain (i.e., slope and elevation) and lens distortion, which may be confounded in oblique photos (from a severe camera tilt toward the site), may dictate the need for additional survey markers to minimize geometric distortions (Rocchini and Di Rita 2005).

Figure 7. Field markers can serve as easily identifiable reference points in the aerial photographs as well as ground control points when GPS coordinates are obtained. GPS coordinates are necessary for geo-rectification and detailed analysis. The field marker on the left incorporates a tripod support, which allows for adjustment of the marker relative to vegetation canopy (e.g., grasses and forbs).

Survey-grade GPS units (i.e., providing differentially corrected precision to at least +/- 1.0 m) are recommended for acquiring coordinates if rectification is needed. If the need is only to return to the approximate area, then recreational-grade GPS units may be adequate.

Figure 8. Low-level aerial digital photograph of a tracked-vehicle disturbance treatment in progress. Field markers of known position/distance apart (circled in white) provide ground-control points for photo interpretation and analysis. Markers were 20 cm x 20 m and photo was taken from an altitude of approximately 90 m (300 ft.).

A laser range finder with accuracy +/- 1 m (3.3 ft.) is optional equipment that may be helpful to determine approximate altitude of the low-level photography system during the photography session. This information may be helpful for taking photographs at a consistent distance above the ground (e.g., 91-98 m [300-320 ft.] vs. 107-113 [350-370 ft.]). This consistency is important to achieve comparable resolutions between sites or time periods when using image-processing software (i.e., pixel-based processing software).

FAA REGULATIONS AND LOCAL CONSIDERATIONS

Federal Aviation Regulations (FAR) exist for moored (i.e., tethered) balloons, kites, unmanned rockets, and unmanned free balloons. FARs can be accessed via the Internet at eCFR, part 101, at http://ecfr.gpoaccess.gov/cgi/t/text/text-idx?c=ecfr&sid=2077d1fab30096f92995797745b00e5a&rgn=div8&view=text&node=14:2.0.1.3.15.2.9.2&idno=14.
This section of the eCFR should be reviewed and understood by all field crew prior to the initial aerial photography session. The regulations include the following:

1. No moored balloon or kite can be operated more than 500 feet above the surface of the earth.
2. No moored balloon or kite can be operated from an area where the ground visibility is less than 3 miles.
3. No moored balloon or kite can be operated within 5 miles of the boundary of any airport.

On military installations, additional regulations may apply. Field crew should communicate with support and/or command centers to determine access to training areas and possible conflicts with both ground and aerial training activities. Communication prior to going to photography sites and prior to launching of the aerial system is important.

FIELD PROTOCOL

STARTUP/SETUP

Local weather ultimately dictates acceptable conditions for launching a tethered low-level aerial photography system. Time of day becomes a consideration if there is little to no cloud cover; 10:00 A.M.-2:00 P.M. is best to minimize shadowing.

Survey markers should be placed in advance of the launch of the blimp/balloon and, ideally, be GPSed at this time. Having at least one crew member available to complete this task speeds the setup and the retrieval of the markers once photographs have been obtained.

Two persons are typically required to ready a low-level photography system. Preparation includes inflation of the blimp/balloon, readying of the camera platform, and testing the R/C controls before launching.

Manufacturer guidelines must be followed for inflating the blimp or balloon. If air temperatures (in the range of 32°-38°C [90°-100°F]) increase significantly during a session, crew should check to see whether some helium should be released to minimize the risk of over-inflation from heating of the gases inside.

PHOTOGRAPHING

The operator of the R/C unit and the crew anchoring the tether reel must work in coordination to position the blimp at the desired altitude and over the landscape feature/area of interest. This is as much an art as it is a science, as wind conditions may change considerably. Usually, conditions are more stable above 91 m (300 ft.) than they are near the ground, so minute-to-minute repositioning of the aerial support system is usually minimal. Also, with sufficient altitude one does not have to be concerned with fitting the area of interest perfectly within the picture frame.

Orientation of the camera frame is optional. For many landscape shots, approximating the long axis of the camera frame on a north-south or east-west orientation is helpful for subsequent review and analysis of photos. However, key landscape features may dictate orientation. For example, this might be justified for photographing stream segments and/or crossings, extensive vehicle tracking, or research plots that matched slope and/or soil conditions (Figure 9). As noted, sufficient altitude to allow for imperfect centering and/or orientation of the area of interest in the photo increases operational efficiency.

Figure 9. Low-level aerial digital photographs highlighting various natural features and human-induced disturbance by orienting picture frame to key landscape features. Left: small creek (meandering from left to right) bisected by a designated military vehicle stream crossing (top to bottom). Right: controlled burn on series of study plots evaluating effects of tracked-vehicle disturbance and fire.

Field experience coupled with subsequent review and analysis of photos will indicate how many photos of an area are needed. Under ideal conditions (i.e., calm or low wind speeds, consistent lighting), two or three photos are likely adequate. It is recommended that at least one true landscape (vs. directly vertical camera position photo) be taken between sites to help identify photography sessions (Figure 2, right). Capturing a part of the blimp/balloon in such a photo at the end of photography at a site serves as a good separator. This is especially important when sites have similar or nondescript landscape features, or when photos will be reviewed, archived, and/or evaluated by individuals who did not participate in the field effort. It is also recommended that one crew member record in a field notebook the approximate start and stop times of individual photography sessions, as that information can be cross-checked with time-and-date stamps on individual photo files if there are questions about the location of individual photos. Other items that may be helpful to record for post-photography-session processing are the locations of individuals, equipment, or vehicles relative to the plot or landscape feature of interest (e.g., north, south, east, west, or specific corner, tree, bend in the stream, etc.).

POST-PHOTOGRAPHY

One virtue of digitally saved photo images is that they can be examined once the camera unit has been retrieved. A laptop computer with a basic photo editor or viewing software package allows field crew to be sure that images of sufficient quality (both in focus and capture of the area of interest) have been obtained. In-the-field image-checking is optional.

Before leaving a site, the field crew should be sure GPS coordinates have been acquired for survey-marker positions, permanent markers have been put in place, and field notes taken about approximate start and stop times of the photography session. One might also consider taking ground-level digital photos of some of the features captured in the aerial photography session to augment interpretation and archives for future review.

POST-FIELD PROCESSING OF IMAGES WITHIN FIRST 24 HOURS

Within 24 hours of obtaining digital photos, the images should be reviewed, sorted, properly labeled (i.e., file name that includes information about location and date). Doing this initial processing shortly after returning from the field ensures that (a) acceptable photos have been obtained (if not reviewed in the field) and (b) they can be archived as well as duplicated. This process can be aided by having the field crew and field notes available for this review. It is recommended that all original image files be saved to a second media storage device. Those identified as keepers should be properly filed electronically in at least two locations.

Some digital cameras can save files on the media storage card in raw format as well as .jpg format. The raw formats are generally manufacturer-specific and identified by extensions such as .arw (Sony), .sfr (Sony), .sr2 (Sony), .crw (Canon), .nef (Nikon), .nrw (Nikon), .raf (Fuji), .ptx (Pentax), and .pef (Pentax). Some cameras provide images in only .jpg (Pentax). Raw files are so named because they have not been processed, meaning the image is not ready to be used by a bitmap graphics editor or printed. Depending on analysis requirements, only .jpg formats may be needed. If given an option, save image files in both formats (raw and .jpg) when the photos are taken, as future image-processing software packages may permit analysis not currently available or easily processed. Raw file formats are recommended to be used with complex image-processing suites. Any compressed image formats (e.g., .jpg) can generally be created from raw format files. The only drawback with raw files is that they require more space per image (4-5 times more) on the storage media than .jpg-formatted files. Therefore, additional media storage cards may be needed in the field. The declining cost of media storage cards offsets any significant increase in digital photo acquisition and storage expenses.

GPS coordinate data should also be processed and appropriately labeled and/or filed to be cross-referenced easily with digital photos. Standard procedures used for differentially correcting coordinate data should be completed. If rectification of aerial photo images is necessary, further processing using Geographic Information System (GIS) software will be necessary. GIS suites such as ArcGIS 9.x (ESRI) are capable of carrying out the rectification process using simple first-, second-, or third-order polynomial transformations. Higher-order transformations will require a larger number of GCPs. GIS personnel can provide the expertise for this task. To carry out specialized spatial analysis conducted within GIS software, georectification is necessary. In addition, geo-rectified images may be useful just as background layers in a GIS database (Georgic and Wagner 2009).

PHOTO INTERPRETATION AND ANALYSIS

If a picture is worth a thousand words, then a high-resolution digital image is worth a thousand data points, so to speak. In most instances, key landscape features are very evident in digital images displayed on standard computer monitors or color laser printers. Basic viewing and editing of files can be accomplished using software such as Adobe Photoshop, ArcSoft Photostudio, or Corel Paint Shop Pro. If metrics (e.g., area, distance/length, object size analysis) are needed to quantify features such as the number or size of bare-ground patches (Figure 10), shoreline length, surface area covered by water, amount of area disturbed by recent traffic, amount of area of distinct vegetation communities/patches, etc, then image-processing software will be required. Currently available specialty packages include but are not limited to Image Tool (free), Image Pro-Plus (Media Cybernetics), and Motik Imaging Software (Motik Instruments Inc.). If rectification procedures have been carried out, then GIS software such as ArcGIS 9.x may be used for digitization of landscape features into polygon features. These polygon features can then be used within the GIS to carry out more-complex spatial analysis such as in diversity and evenness metrics, fragmentation analysis, or patch characteristic analysis. In addition, once in a GIS, properties of the polygonal features of interest may be correlated or compared to other geo-referenced landscape features (e.g., soil types, large-scale remotely sensed data, LIDAR).

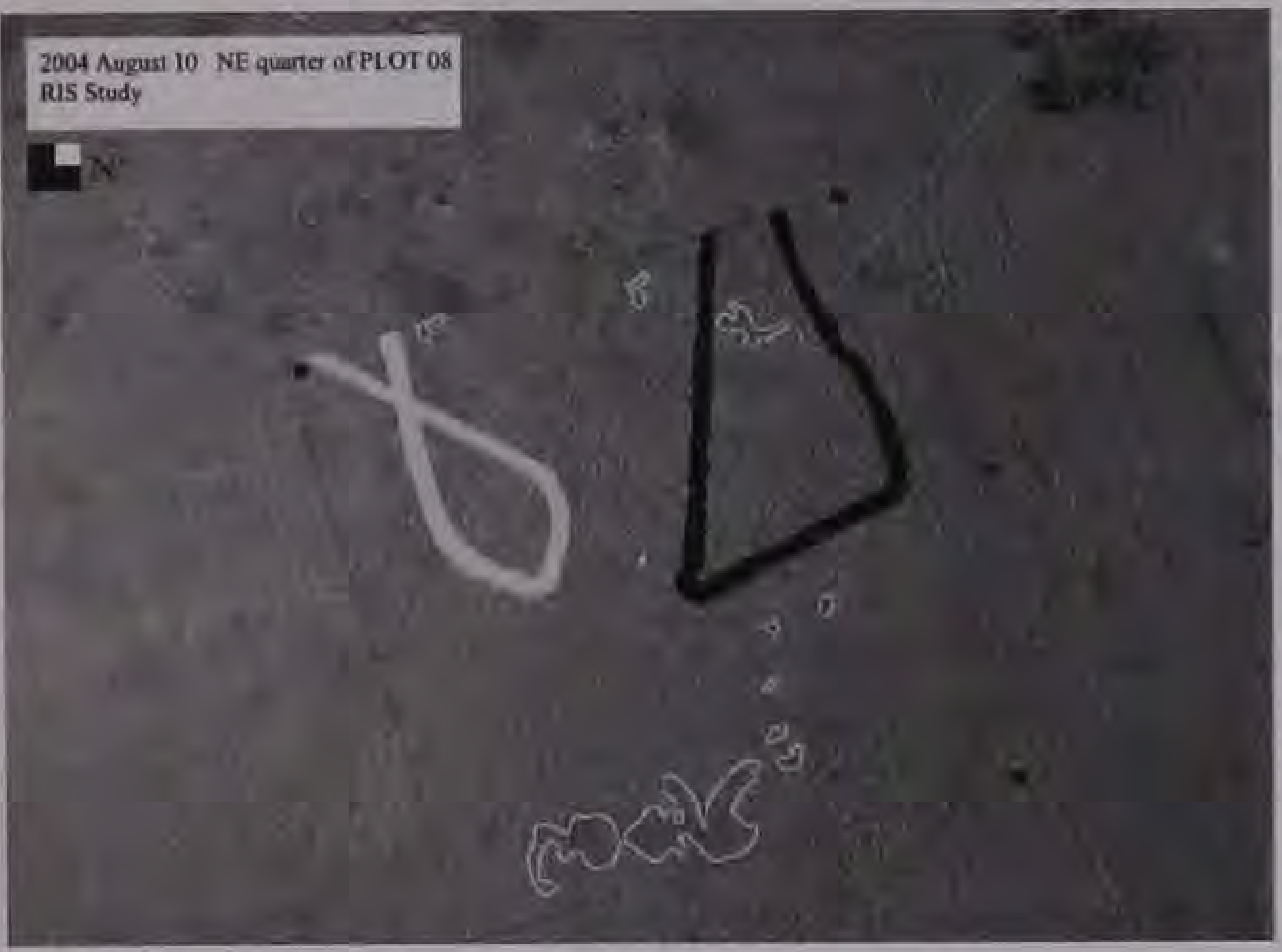

Figure 10. Digital photograph acquired with LLAPS that was analyzed for key physical and biological features using a software image processing program. Features highlighted included demarcation of the 1 hectare (100 m x 100 m) study plot (perimeter denoted by black lines and corner by small black squares), recent tracked vehicle traffic (solid white and black areas), and non-woody vegetation patches (areas outlined in white). Amount of area for each parameter was calculated by the software after being outlined manually by the software user.

RESULTS AND DISCUSSION

The aerial photography method described can provide a compromise between ground-based sampling and remote sensing (satellite) image acquisition (Murden and Risenhoover 2000). The initial investment in equipment is greater than that required for probably any ground-based approach. Thus, multiseason, multiyear data-acquisition objectives are normally required to justify acquisition of a LLAPS. At a minimum LLAPS can produce high-quality images that capture current conditions, document species, growth forms, bare ground, and disturbance within large plots, and provide a record for future evaluation and review. Our latest camera equipment yielded resolution of 6 cm per pixel when capturing the 100 m x 100 m plots per image. Although it is unlikely that this degree of resolution can enable identification of individual plants or measurement of below-canopy bare ground, patches of invasive species, exposed soil/rock, or disturbance > 1m^2 are readily identifiable in such images. This potential is further realized when ground-truthing is incorporated into the post-photography sessions to confirm the presence of physical and biological features of interest. A LLAPS is easily used in most tallgrass prairie landscapes because nonwoody plant forms dominate (i.e., few trees to snag the tetherline), topography is typically flat to rolling, riparian zones are narrow (allowing one to float the aerial platform over a zone during wind conditions), and disturbances on a large scale (fire, vehicle traffic, mowing, haying, etc.) are common.

We have used images from the LLAPS on Fort Riley to document tracked-(i.e, Humvees and tanks) and wheeled-vehicle disturbance on training landscapes. This included obtaining images within minutes of vehicle traffic for controlled research plots, recent (<24 hours) military training exercises, and in-progress land rehabilitation efforts involving grading and reseeding. Subsequent monitoring (1-24 months later) of such disturbed areas—as well as undisturbed areas-yielded images revealing changes and/or recovery of vegetative cover as well as residual effects of vehicle traffic.

PROS AND CONS

There are advantages and disadvantages to using the LLAPS approach. Pros include being able to acquire data for variable plot sizes—including large plots (0.25-1.0 ha)—local scheduling (versus acquisition of satellite imagery), multiple photo sessions of individual plots/areas within just a few hours (progress of a burn; see Figure 9) or days or weeks at little additional costs (versus satellite imagery), flexibility of resolution and scale of photos, and minimum crew size of two (although three-four is probably the optimum crew size). Cons include initial startup costs, weather conditions (snow, rain, high winds, high temperatures) that may preclude day-to-day use, and need to modify one or two vehicles for field operations.

CONCLUSION

The acquisition of high-resolution digital images using the tethered LLAPS we have used in a tallgrass prairie landscape is most likely less labor-intensive than ground-based methods and provides temporal and spatial documentation that can be easily archived. Whether for rapid assessment or more detailed post acquisition analysis, this approach offers a viable alternative to both traditional ground-based sampling and satellite imagery to document landscape conditions and changes over time. At a minimum, use of LLAPS can complement these other traditional methods where quantifying physical and biological features at various scales is helpful to monitoring and management efforts.

ACKNOWLEDGMENTS

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RESTORATION AND RECONSTRUCTION

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EFFECTS OF MOWING ON ABUNDANCE AND PERSISTENCE OF TALLGRASS PRAIRIE FORBS SEEDED INTO AN ESTABLISHED STAND OF PRAIRIE GRASSES: TEN YEARS AFTER SOWING

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Abstract: In fall 1998, 23 forb species were broadcast-seeded into a 25-year-old planted stand of warm-season prairie grasses at the University of Northern Iowa in Cedar Falls, Iowa. To determine their effect on forb establishment, mowing treatments were applied in 1999 and 2000 and compared with no-mow controls. In the summer of 2008, the plots were resampled to assess the effect of ten years' time on forb abundance and diversity. We hypothesized that the forb abundance, richness, and diversity would be greater in plots that were mowed, over the no-mow controls. Our results showed that both the number of forb shoots ($p=0.004$) and the number of forb species were significantly greater ($p<0.001$) in mowed plots over no-mow controls. We also found that the number of warm-season grass shoots was significantly greater ($p=0.026$) in no-mow plots. The Simpson diversity index in mowed plots was significantly ($p=0.002$) higher (0.650) than the unmowed plots (0.243). This study shows that frequent mowing in the first season after sowing novel forb species into an established grassland can have a profound impact on the plant community well into the future: increased forb abundance, increased forb richness, and a more diverse plant community.

Key Words / Search Terms: forb enhancement, mowing, species-poor grassland, species richness, prairie reconstruction, tallgrass prairie

INTRODUCTION

Since the first prairie conference, held in Galesburg, Illinois, in 1968, there have been increased efforts to use native tallgrass prairie species for conservation reserve lands, state/county rights-of-way, pasture, and biofuels (Smith 2010, Schramm 1970). Many of the early prairie plantings resulted in plant communities dominated by warm-season prairie grasses with few to no forbs (Williams 1999). The ecological benefits of increasing the species diversity of grasslands with native prairie forbs are well documented. The benefits include maintaining above-ground growth through weather extremes; increased habitat for and diversity of other living organisms, including habitat-sensitive species; and reduction of noxious weed invasions (Tilman and Downing 1994, Reis et al. 2001, Tilman 1997).

The negative effects of repeated clipping of prairie grasses is well documented. Biswell and Weaver (1933) found that clipping big bluestem (*Andropogon gerardii*), switchgrass (*Panicum virgatum*), and sideoats grama (*Bouteloua curtipendula*) at a 5 cm height, every two weeks during one growing season, significantly reduced their root and shoot growth. Previous research has demonstrated that novel prairie species can be successfully added into species-poor grasslands by frequently mowing the stand for a growing season without destroying the established vegetation (Williams et al. 2007, Snyder 2010). However, studies to date have been time-limited, usually to one or two years after the experimental treatments. The fate of the introduced plant species over time is not known. Some resource managers have speculated that forb abundance would eventually decrease over time as the forbs are out-competed for resources by the more dominant native prairie grasses.

A 1998-2000 research project on a reconstructed tallgrass prairie on the campus of the University of Northern Iowa determined that one season of frequent mowing of a stand of tallgrass prairie promoted establishment of forbs seeded into the grass stand (Williams et al. 2007). The plots of this 1998-2000 experiment were resampled in 2008. The plots were fall-burned twice and spring-burned twice between 2000 and 2008, but no additional mowing treatments were applied to them.

The objectives of this study were to (1) assess and compare abundance, richness, and diversity of forbs in mowed and unmowed plots seeded in 1998, sampled in 2000, and resampled in 2008; and (2) assess and compare native grass abundance in mowed and unmowed plots ten years after the initial seeding. We hypothesized that the forb abundance, richness, and diversity would be greater in plots that were mowed than in the no-mow controls.

This research project was designed to determine whether forbs added into an established stand of prairie grasses would persist in the plant community ten years after seeding and mowing treatment. Information from this research can be used to revise or develop alternative management plans to increase diversity of species-poor grasslands.

METHODS AND MATERIALS

SITE DESCRIPTION

The reconstructed tallgrass prairie study site is part of the University of Northern Iowa campus tallgrass prairie preserve in Cedar Falls, Iowa. It is an alluvial bench of about six hectares along a creek, and the soil type is a mixture of loamy sand and gravel (Fouts and Wisner 1982). Prior to 1973, the site was a hayfield that consisted of nonnative pasture grasses and legumes (Smith 2010). In spring of 1973, the hayfield was plowed, disked, and seeded with 25.7 kg/ha of cultivated varieties of native warm-season grasses. The seed mix consisted of big bluestem, little bluestem (*Schizachyrium scoparium*), Indiangrass (*Sorghastrum nutans*), switchgrass, and sideoats grama. The site was divided into four burn sub-units and each unit burned every 2-3 years. In 2008 as in 1998, most of the aboveground growth on the research site consisted of big bluestem and Indiangrass. Kentucky bluegrass (*Poa pratensis*) and smooth brome (*Bromus inermis*) were also present.

SEED MIX/SEEDING/MOWING

Twenty-three forb species were hand-broadcast-seeded in November 1998 (Table 1). All seed was tested for viability by an independent seed-testing lab (Hulsey 1999). Seeding rate for each species was calculated from pure live seed (PLS) value and seed weight estimates of Henderson and Kern (1998).

Table 1. Forb species seeded in fall 1998. Individual seeding rates were based upon Henderson and Kern (1999) seed weight estimates. Each species was tested for pure live seed by an independent seed-testing lab (Hulsey Seed Laboratory Inc. 1999).

SPECIES	VIABLE SEEDS SOWN/M^2
New England aster (*Symphyotrichum novae-angliae*)	134.7
Black-eyed susan (*Rudbeckia hirta*)	56.7
Stiff goldenrod (*Oligoneuron rigidum*)	28.1
Gray-headed coneflower (*Ratibida pinnata*)	22.2
Purple prairie clover (*Dalea purpurea*)	13.5
Wild bergamot (*Monarda fistulosa*)	13.1
Prairie coreopsis (*Coreopsis palmata*)	12.8
Thimbleweed (*Anemone cylindrica*)	10.0
Prairie blazingstar (*Liatris pycnostachya*)	8.6
Sweet coneflower (*Rudbeckia subtomentosa*)	7.6
Round-headed bush clover (*Lespedeza capitata*)	6.1
Flowering spurge (*Euphorbia corollata*)	4.1
Rough blazingstar (*Liatris aspera*)	4.1
Pale purple coneflower (*Echinacea pallida*)	4.0
Ox-eye sunflower (*Heliopsis helianthoides*)	4.0
Ohio spiderwort (*Tradescantia ohiensis*)	3.7
Illinois bundleflower (*Desmanthus illinoensis*)	3.6
Leadplant (*Amorpha canescens*)	2.7
Canada anemone (*Anemone canadensis*)	2.7
Butterfly milkweed (*Asclepias tuberosa*)	2.5
Golden alexanders (*Zizia aurea*)	2.5
Showy tick trefoil (*Desmodium canadense*)	2.4
Compassplant (*Silphium laciniatum*)	0.5

In 1999, plots designated for mowing were mowed weekly from mid-May to late August at a height of 10 cm. In fall 2002, as part of another experiment, ten additional forb species were broadcast-seeded into the mowed plots of the 1998-2000 experiment. Forb species seeded in 2002 were excluded from the 2008 mow and no-mow comparisons.

EXPERIMENTAL DESIGN/DATA ANALYSIS

The 1998-2000 experiment was a randomized block design with two blocks and two mowing treatments: weekly mowing and no-mow controls. There were six randomly assigned 15 x 20 m plots in each block and each treatment was replicated three times per block (Figure 1).

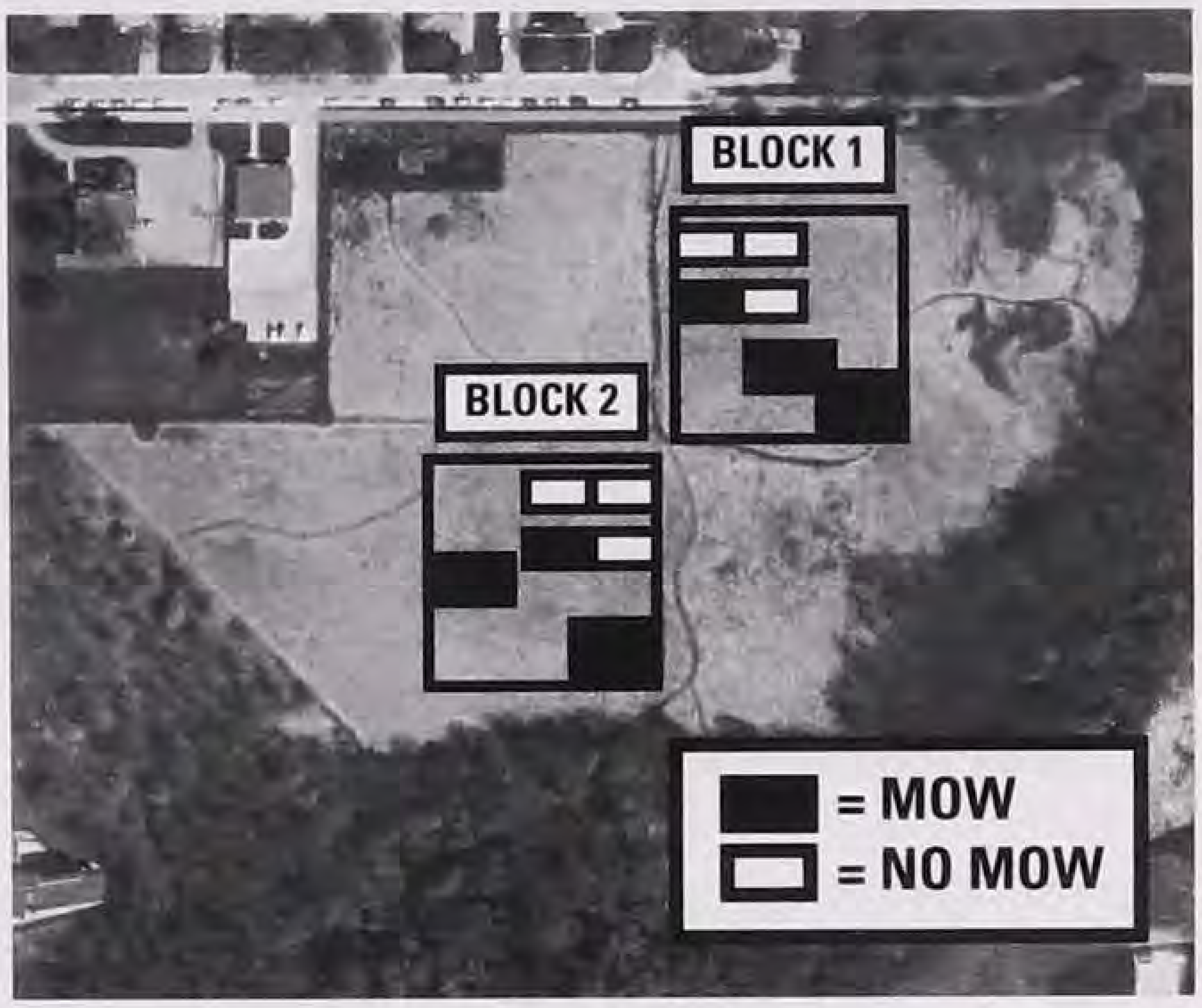

Figure 1. Aerial view of the research site at the University of Northern Iowa Biological Preserves. The experimental study plots are part of a 1973 reconstructed prairie that was initially dominated by warm-season grasses. The experiment was a randomized block (60 x 60 m) design with two blocks and six 15 x 20 m plots per block.

Data collected in June 2008 was compared to the final data set collected in September 2000 of the original experiment. All data were analyzed using a two-way analysis of variance (ANOVA) or repeated measures ANOVA to evaluate the changes over time. To test for the assumptions of ANOVA, skewness (g1) and kurtosis (g2) and homoscedasticity (Bartlett's test) were first calculated for all data sets. A student's t-test (alpha=0.05, with infinite degrees of freedom) was conducted to determine if the data had significant skew or kurtosis from zero (Wilkinson 1989). Data sets were square root-transformed to run the ANOVA, and the means were back-transformed to report the data. To determine whether mowing had an effect on species diversity, a Simpson index of diversity (1 – D) was calculated for each plot and analyzed with a one-way ANOVA (Brower et al. 1998).

VEGETATION SAMPLING

The vegetation was sampled in late June 2008. Five 0.25m^2 quadrats were randomly placed in each plot. Within each quadrat, all forb and grass species were identified and the number of shoots were counted and recorded for each species. Data analysis was limited to the forb species seeded in the 1998-2000 experiment.

RESULTS AND DISCUSSION

GRASS/FORB ABUNDANCE

In 2008, forb abundance was significantly higher in mowed plots over no-mow controls. Forb shoots had increased significantly (P<0.001) in mowed plots from 2000 to 2008 and only slightly increased in no-mow controls during this same time (Figure 2). In addition, there were significantly (p=0.004) more forb shoots in mowed plots as compared to no-mow controls (Figure 3). The increase in forb abundance may be due to new recruitment of individuals by seed and/or increased vegetative growth of established plants (Williams et al. 2007). Forb recruitment by seed was illustrated by black-eyed Susan (*Rudbeckia hirta*) in this experiment. In 2000, there were thirty-one times more flowering black-eyed Susan plants in mowed plots than in no-mow controls (Williams et al 2007). From 2000 to 2008, black-eyed Susan abundance had declined nearly fifteen-fold in no-mow plots and declined by only 14% in mowed plots (Table 2). Because of black-eyed Susan's annual, biennial, and short-lived perennial life cycle, it is clear that plants present in 2008 were recruited by seed from the progeny of the original 2000 black-eyed Susan plants (Gleason and Cronquist 1991). In addition to being more numerous in the mowed plots, individual plants appeared to be much larger than in no-mow controls (Figures 4 and 5). It appears that the effects of mowing, which promoted increased forb growth and maturity in the original experiment continued in subsequent years. Conversely, forbs that established in the original experiment, but didn't have the benefits of establishment mowing, were less likely to expand by seed and/or by vegetative spread.

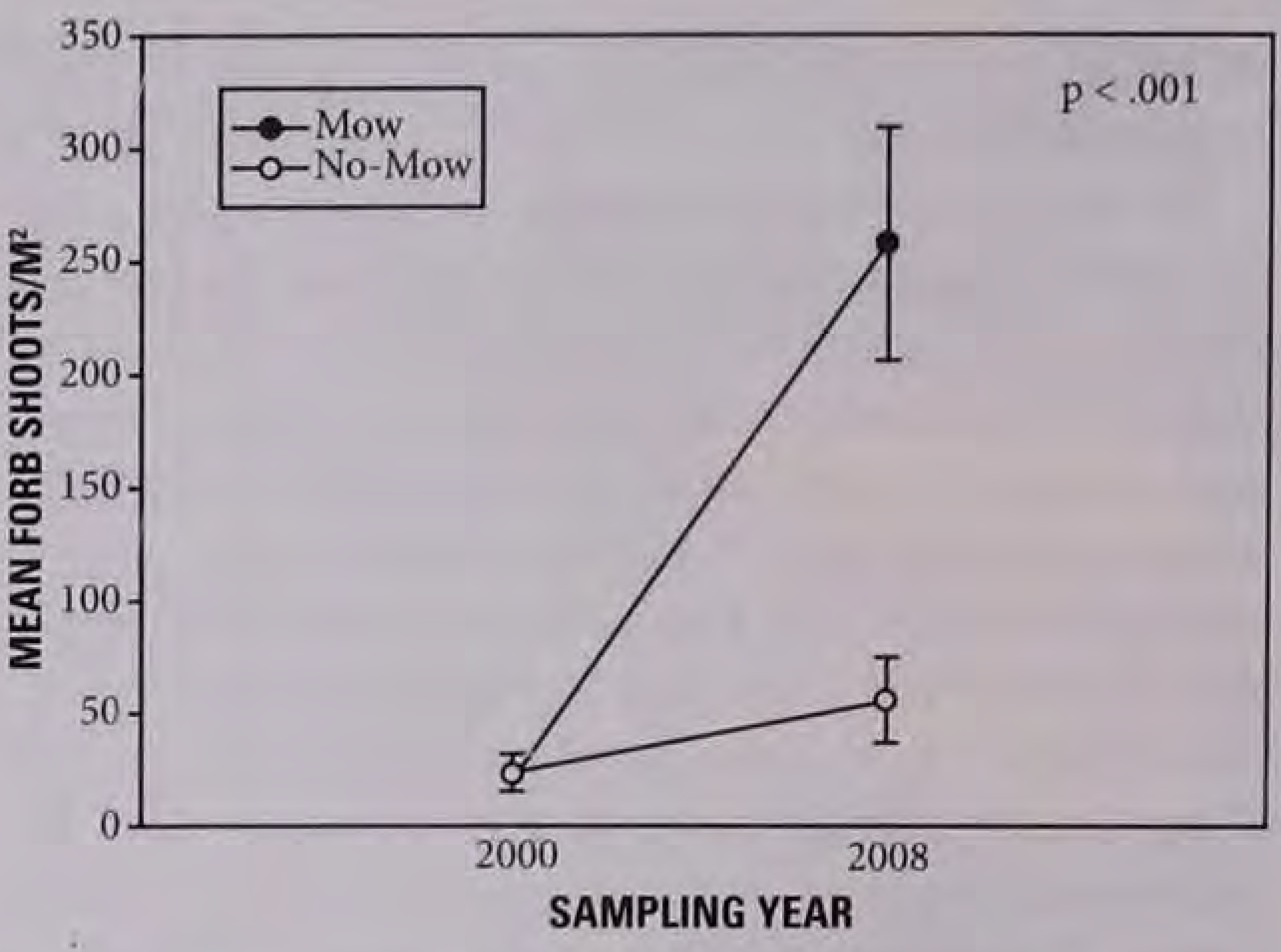

Figure 2. Mean number of forb shoots sampled in mow and no-mow plots in September 2000 and June 2008. Forb shoots increased significantly (p<0.001) in the mow plots from year 2000 to year 2008. Data sets were analyzed with a repeated measures ANOVA. Means were back-transformed to report the data.

Table 2. Mean shoots/m^2 of forbs sampled in mow and no-mow plots in 2000 and 2008. Forbs were seeded in fall 1998 and mowing was conducted in summer of 1999. All species except Illinois bundle flower (*Desmanthus illinoensis*) were detected on the site in 2000.

COMMON NAME	**SCIENTIFIC NAME**	**MOW**			**NO-MOW**		
		2000	**2008**	**GAIN(+) LOSS(-)**	**2000**	**2008**	**GAIN(+) LOSS(-)**
Black-eyed susan	*Rudbeckia hirta*	10.0	8.8	-	12.7	0.8	-
Butterfly milkweed	*Asclepias tuberosa*	0.0	0.4	+	0.3	0.0	-
Canada anemone	*Amenone canadensis*	0.0	0.0	no change	0.0	0.0	no change
Compass plant	*Silphium laciniatum*	0.0	0.3	+	0.2	0.5	+
Flowering spurge	*Euphorbia corollata*	0.2	0.0	-	0.2	0.0	-
Golden alexanders	*Zizia aurea*	0.0	1.1	+	0.3	7.3	+
Gray-headed coneflower	*Ratibida pinnata*	4.0	120.3	+	4.4	4.1	-
Illinois bundle flower	*Desmanthus illinoensis*	0.0	0.0	no change	0.0	0.0	no change
Leadplant	*Amorpha canescens*	0.0	0.8	+	0.0	0.0	no change
New England aster	*Symphyotrichum novae-angliae*	1.7	4.1	+	1.1	0.3	-
Ohio spiderwort	*Tradescantia ohiensis*	0.0	0.4	+	0.0	0.7	+
Ox-eye sunflower	*Heliopsis helianthoides*	1.0	5.1	+	0.8	0.0	-
Pale purple coneflower	*Echinacea pallida*	0.3	3.3	+	0.3	2.5	+
Prairie blazingstar	*Liatris pycnostachya*	0.0	0.3	+	0.0	0.0	no change
Prairie coreopsis	*Coreopsis palmata*	0.0	4.7	+	0.0	9.6	+
Purple prairie clover	*Dalea purpurea*	0.0	1.5	+	0.0	0.0	no change
Rough blazingstar	*Liatris aspera*	0.0	0.3	+	0.0	0.0	no change
Round-headed bush clover	*Lespedeza capitata*	0.2	0.3	+	0.1	0.0	-
Showy tick trefoil	*Desmodium canadense*	0.1	5.5	+	0.0	0.0	no change
Stiff goldenrod	*Oligoneuron rigidum*	2.0	64.9	+	1.2	11.7	+
Sweet coneflower	*Rudbeckia subtomentosa*	0.0	1.3	+	0.0	0.0	no change
Thimbleweed	*Anemone cylindrica*	0.4	8.8	+	0.7	0.0	-
Wild bergamot	*Monarda fistulosa*	2.4	16.8	+	1.7	14.7	+

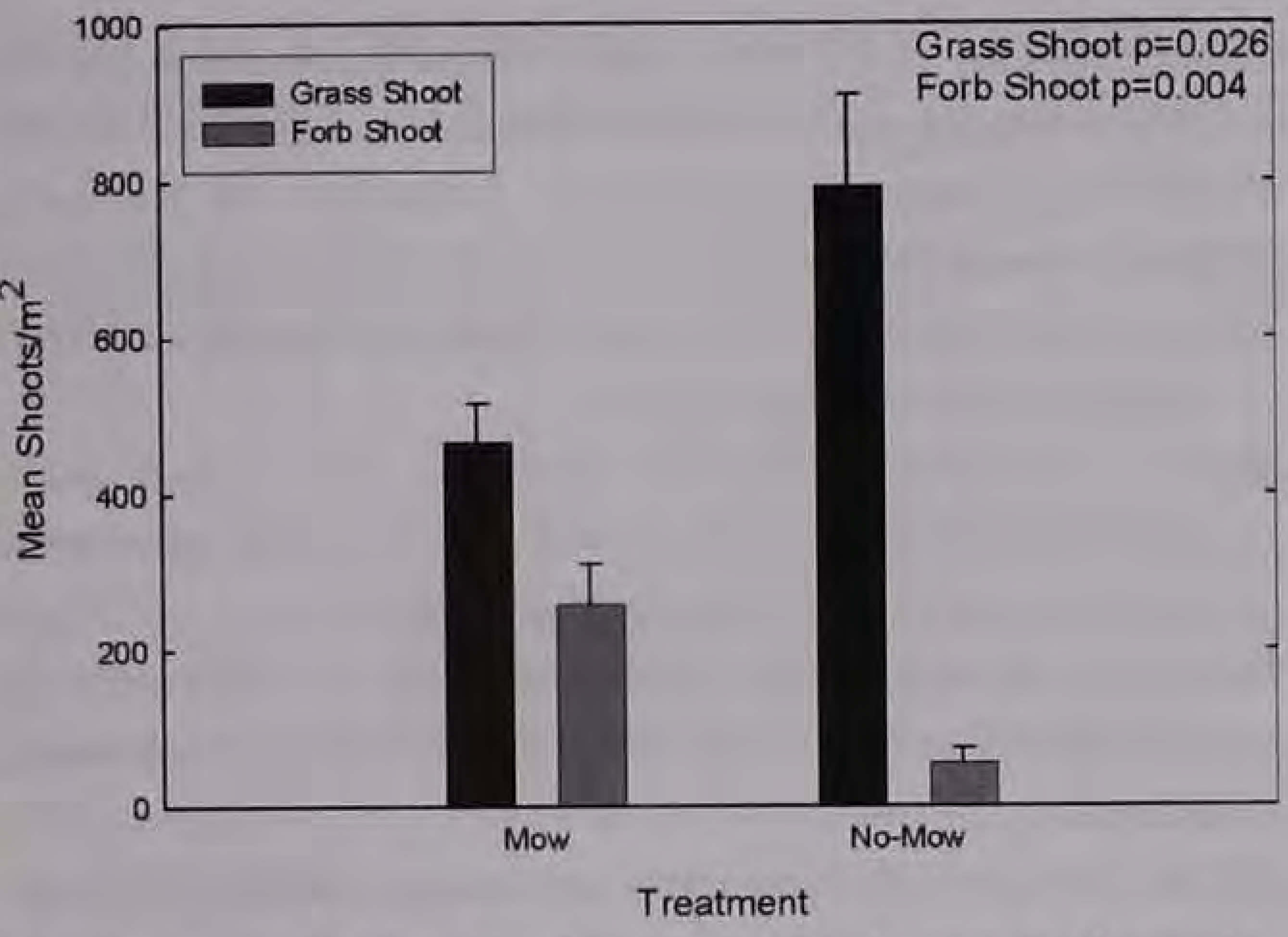

Figure 3. Mean forb and grass shoots sampled in June 2008. The number of forb shoots was significantly (p=0.004) greater and the number of grass shoots was significantly (p=0.026) lower in mowed plots when compared with no-mow controls. Grass and forb shoots were analyzed independently with a one-way ANOVA. Means were back-transformed to report the data.

Native grass abundance in 2008 was significantly lower in mowed plots than no-mow controls (Figure 3). This result can, in part, be attributed to two factors: extensive growth of forbs in mowed plots in 1999, and suppression of above- and below-ground growth of the grasses by frequent mowing in 1999 (Williams et al. 2007, Biswell and Weaver 1933). In the original experiment, forb shoot and root biomass were significantly greater in mowed plots as compared to the forbs sampled in no-mow controls (Williams et al 2007). It appears that if forbs develop extensive root and shoot growth as a result of suppression of the established grasses in the first growing season, they will expand and occupy more above-ground space without additional suppression of the grasses during subsequent growing seasons. As forbs occupy more space above ground, it is reasonable to assume they also occupy more below-ground space. Another possible explanation for increased forb expansion from 2000 to 2008 is that the prairie grasses, which were mowed weekly in the summer of 1999, had not fully recovered nine years later.

Figure 4. Forbs were seeded into a 25-year-old warm-season grass stand in fall 1998. This plot was mowed weekly in 1999. Note the compassplant (*Silphium laciniatum*) flower stalks developing in the right-hand corner of the photograph. Photo was taken in late June 2008.

FORB RICHNESS

Species richness increased significantly (p=0.001) in mowed plots and significantly decreased in no-mow plots from 2000 to 2008 (Figure 6). Twenty-three species were seeded into the research plots in 1998. Ten years later, all of the original twenty-three species, except Illinois bundle flower (*Desmanthus illinoensis*), were present on the research site. Twenty of the original twenty-three species were present in quadrat samples in the formerly mowed plots in 2008 while only ten were in samples in the no-mow plots (Table 2). Furthermore, nineteen of twenty-three species sampled in the formerly mowed plots had increased in abundance from

Figure 5. Forbs were seeded into a 25-year-old warm-season grass stand in fall 1998. This plot was not mowed in 1999. Note the two large compassplant (*Silphium laciniatum*) leaves without flower stalks. Photo was taken in late June 2008.

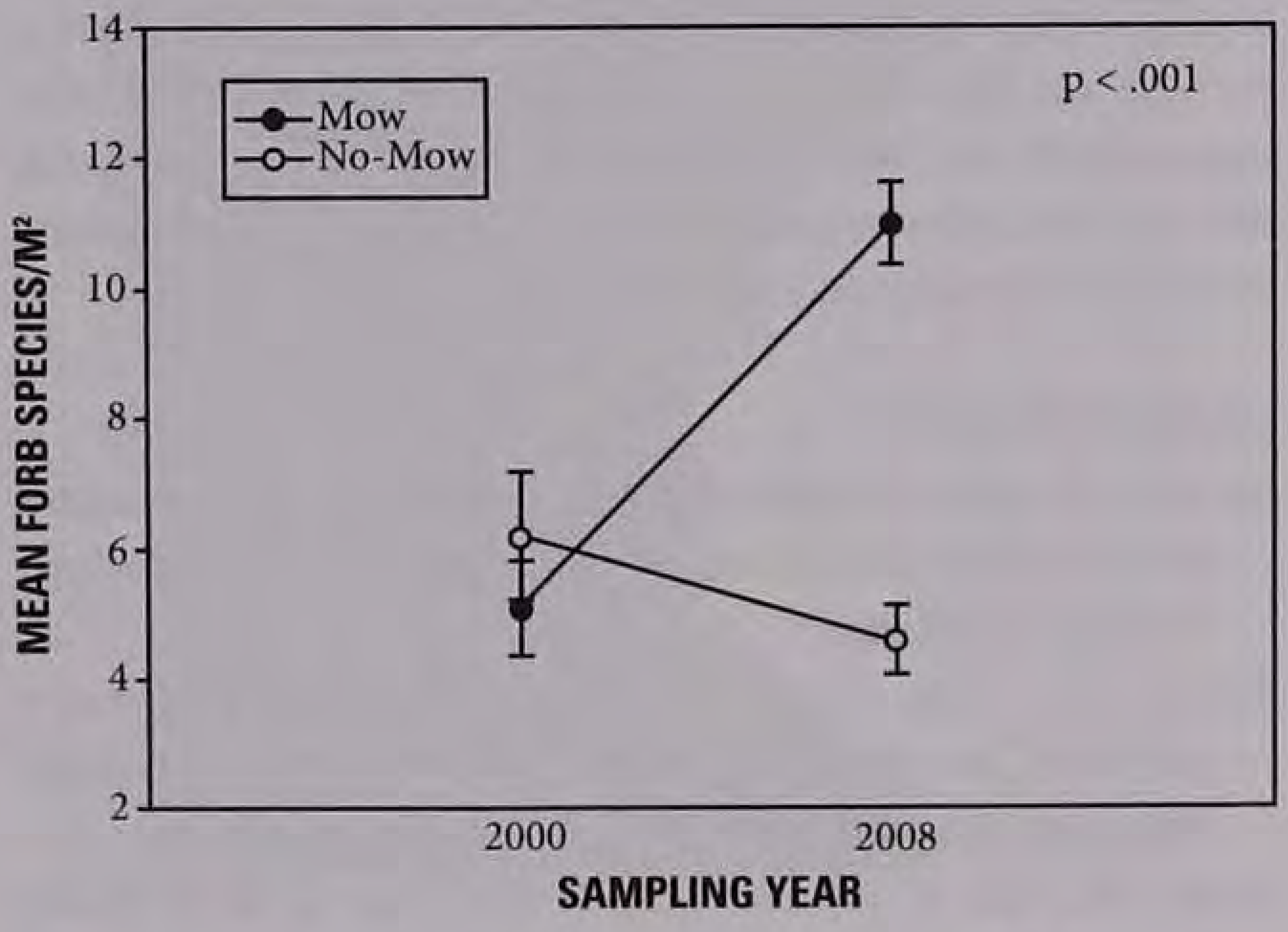

Figure 6. Mean number of forb species sampled in mow and no-mow plots in September 2000 and June 2008. Forb species significantly (p<0.001) increased in the mow plots from 2000 to 2008 and significantly (p<0.001) decreased in the no-mow plots from 2000 to 2008. Data sets were analyzed with a repeated measures ANOVA. Means were back-transformed to report the data.

the 2000 sample, whereas only seven of the twenty-three species in the no-mow plots increased in abundance in the same time span (Table 2). Only one species, flowering spurge (*Euphorbia corollata*), that was detected in mowed plots in 2000 was not found in mowed plots in 2008 (Table 2). By contrast, five species detected in no-mow plots in 2000 were not found in these plots in 2008 (Table 2). By late June in 2008, warm-season grasses were already developing a closed canopy in no-mow plots as compared to mow plots (Figures 4 and 5). Many forbs observed in no-mow plots, except compassplant (*Silphium laciniatum*), appeared to be below the grass canopy (Figure 5). Low light levels created by the warm-season grass canopy may be responsible for the decline of forbs in no-mow plots. It appears that the successful invasion by a novel species into a highly competitive stand of prairie grasses is more likely to occur if the grasses are suppressed (Williams et al. 2007, Synder 2010). On the other hand, novel species that are able to establish in prairie grasses that are not suppressed are more likely to decline in abundance or disappear over time (Table 2).

SPECIES DIVERSITY

In 2008, Simpson's Index of Diversity (1-D) species diversity was significantly (p=0.002) greater in mowed plots (0.650) over no-mow controls (0.243). Research has shown that there are many ecological benefits of diverse plant communities as compared to less diverse stands (Tilman and Dowling 1994, Tilman 1997, Ries et. al 2001). Clearly, it is possible to convert a species-poor stand of warm-season prairie grasses into a species-diverse plant community that can persist for the long term.

CONCLUSION

Mowing can be an effective tool to increase diversity in species-poor grass stands. Forbs can establish and persist when broadcast-seeded into a stand dominated by prairie grasses without any disturbance to the grasses. However, forb seeding coupled with one season of frequent mowing after sowing will significantly increase forb abundance and richness of the stand both initially and nine years later.

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GENERAL LAND OFFICE SURVEY VEGETATION ANALYSIS FOR EFFIGY MOUNDS NATIONAL MONUMENT, ALLAMAKEE AND CLAYTON COUNTIES, IOWA

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Abstract: The development of technologies like GIS and digital orthophotography has made it possible to use nineteenth-century U.S. General Land Office survey data for creating maps of presettlement vegetation in Iowa and elsewhere. Such maps become invaluable both for recognizing and understanding changes in the U.S. landscape over the last 150 years and more, and for creating a basis for registering ongoing developments. The survey notes themselves provide insights into the goals and methods of the surveys, as well as information about the men who did the original surveying.

Key Words/Search Terms: GIS/surveyors, orthophotos, plat maps

INTRODUCTION

The U.S. General Land Office, now succeeded by the Bureau of Land Management, commissioned surveys of U.S. territory beginning early in the nineteenth century. Intended to determine the feasibility of settlement—primarily the establishment of farms—the surveys also included notes on "bearing trees" and considered the availability of construction materials for farms and for town buildings. These were not vegetation surveys, but they have, since the 1920s at least, become the groundwork for highly detailed maps of presettlement vegetation. Digital orthophotos and GIS now make it possible to digitize, rectify, and map these surveys with great accuracy, as well as to provide a visual base. Bearing trees are digitized using species and diameter as well as any distinguishing features. Plat line data are mapped using a code and a description field. Line notes use fields denoting vegetation type, trees, undergrowth, rating, description, and notes. Plat point data are mapped using a code and description. With this data, usable broad-based information can be obtained, even though the surveys were not a scientific vegetation study but a tool to be used by settlers during the westward expansion. Knowledge of who the surveyors were and where they came from, the terminology of the time, and the soil and vegetation communities of the area surveyed aid in survey map interpretation. The recent mapping of Effigy Mounds National Monument in Iowa is just one example of ongoing mapping projects.

G.L.O. SURVEY BEARING TREES

"Bearing tree" as a descriptor is general in nature and not part of a vegetation survey; however, trees that were so designated have characteristics which were chosen to make them easy to relocate. According to surveyor instructions, bearing trees were to be located in predetermined directions from the section corners, with the azimuth and distance, species, and diameter to be recorded during the survey. Some short-lived species were ignored. Trees were also of certain age classes, although young trees were ignored unless they were the only ones in the area. Trees on section lines and those at the quarter-section corner were also recorded. Figure 7 shows how a bearing tree was mapped.

EARLY G.L.O. SURVEY MAP USAGE

Working with the notes made by the surveyors of the Public Land Survey of Minnesota conducted from 1847 to 1907, a research assistant in the USDA Bureau of Agricultural Economics in Washington, D.C., drew and colored a map he called *The Original Vegetation of Minnesota* (Figure 1). The map's originator, Francis J. Marschner, had never been to Minnesota, but his meticulous, highly detailed, five-foot-tall paper creation made between 1929 and 1931, has become significant enough to be known simply as "Marschner's map." At some point the original disappeared, but a copy was discovered by Miron "Bud" Heinselman at the USDA North Central Forest Experiment Station

Figure 1. Reconstruction of Francis J. Marschner's map, *The Original Vegetation of Minnesota*, 1929–31, based on General Land Office survey notes.

in St. Paul, reconstructed, and printed in 1974 at the original scale of 1:500,000. On it Marschner identified 18 classes of vegetation:

0 Undefined
1 Prairie
2 Wet prairie
3 Brush prairie
4 Aspen-oak land
5 Oak openings and barrens
6 Big woods-hardwoods (oak, maple, basswood, hickory)
7 River bottom forest
8 Aspen-birch (trending to hardwoods)
9 Mixed hardwood and pine (maple, white pine, basswood, etc.)
10 White pine
11 Mixed white pine and red pine
12 Jack pine barrens and openings
13 Pine flats (hemlock, spruce, fir, white pine, aspen)
14 Aspen-birch (trending to conifers)
15 Conifer bogs and swamps
16 Open muskeg
17 Lakes (open water)

Working years after Marschner, John T. Curtis (1959) published *The Vegetation of Wisconsin: An Ordination of Plant Communities*, and J. William Trygg (1964) completed the Trygg Composite Maps of Minnesota, Wisconsin, Michigan, and Iowa; both used G.L.O. survey information. Two years after the publication of the restored Marschner map, Robert W. Finley (1976), with the USDA Forest Service, created yet another Wisconsin map, *Original Vegetation Cover of Wisconsin* (Figure 2), also using land survey notes from the mid-1800s. It is of significance that Curtis and his students (Cottam and Curtis 1956) used the bearing trees recorded in the survey notes to determine tree density in forests; the distance from a section corner to a bearing tree was thought to be an indication of density. However, this may not be the case, as bearing trees had predetermined criteria for selection (i.e., diameter, direction from the section corner, and species).

Figure 2. Finley's Presettlement Vegetation of Wisconsin.

MAPPING AND ADVANCED TECHNOLOGY

Minnesota's adaptation of historical data to geographic information systems (GIS) began in 1991 when the Department of Natural Resources obtained the first GIS programs, ARCInfo Workstation with a digitizing table and ArcView 1.0, a viewing program. The Division of Forestry digitized the Marschner map from the reproduction of the original 1:500,000 paper map. Because of the size of the map, many small polygons were not captured. With the development of GIS coverage by the Minnesota Department of Natural Resources, the state's Division of Parks and Recreation set up a project to redigitize the original notes to a better scale. The original notes were obtained from the Wilson Library, University of Minnesota, for each of the state parks, which cover from 300 acres to 34,000 acres. The project was contracted with TAIGA (Technical Associates in Geographic Analysis), Elizabeth Hobbs, and John Lunde. The notes and plat maps were digitized to one mile outside each individual park's statutory boundary. Parks completed, with their city/town location and size, include the following (Figure 3):

1. Banning, Sandstone, 6,000 acres
2. Charles A Lindbergh, Little Falls, 300 acres
3. Crow Wing, Brainerd, 1,200 acres (Figures 4, 5, and 6)
4. Cuyuna Country State Recreation Area, Crosby-Ironton, 5,000 acres
5. Father Hennepin, Isle, 300 acres
6. Interstate, Taylors Falls, 300 acres
7. Lake Maria, Monticello, 1,500 acres
8. Mille Lacs-Kathio, Onamia, 10,000 acres
9. St. Croix, Hinckley, 34,000 acres
10. Savanna Portage, McGregor, 16,000 acres
11. Wild River, Almelund, 7,000 acres

MAPPING IN IOWA: EFFIGY MOUNDS NATIONAL MONUMENT

Paul Anderson at Iowa State University has been conducting research to digitize the maps of the Iowa survey, which was completed between 1832 and 1859. Effigy Mounds National Monument obtained the General Land Office notes

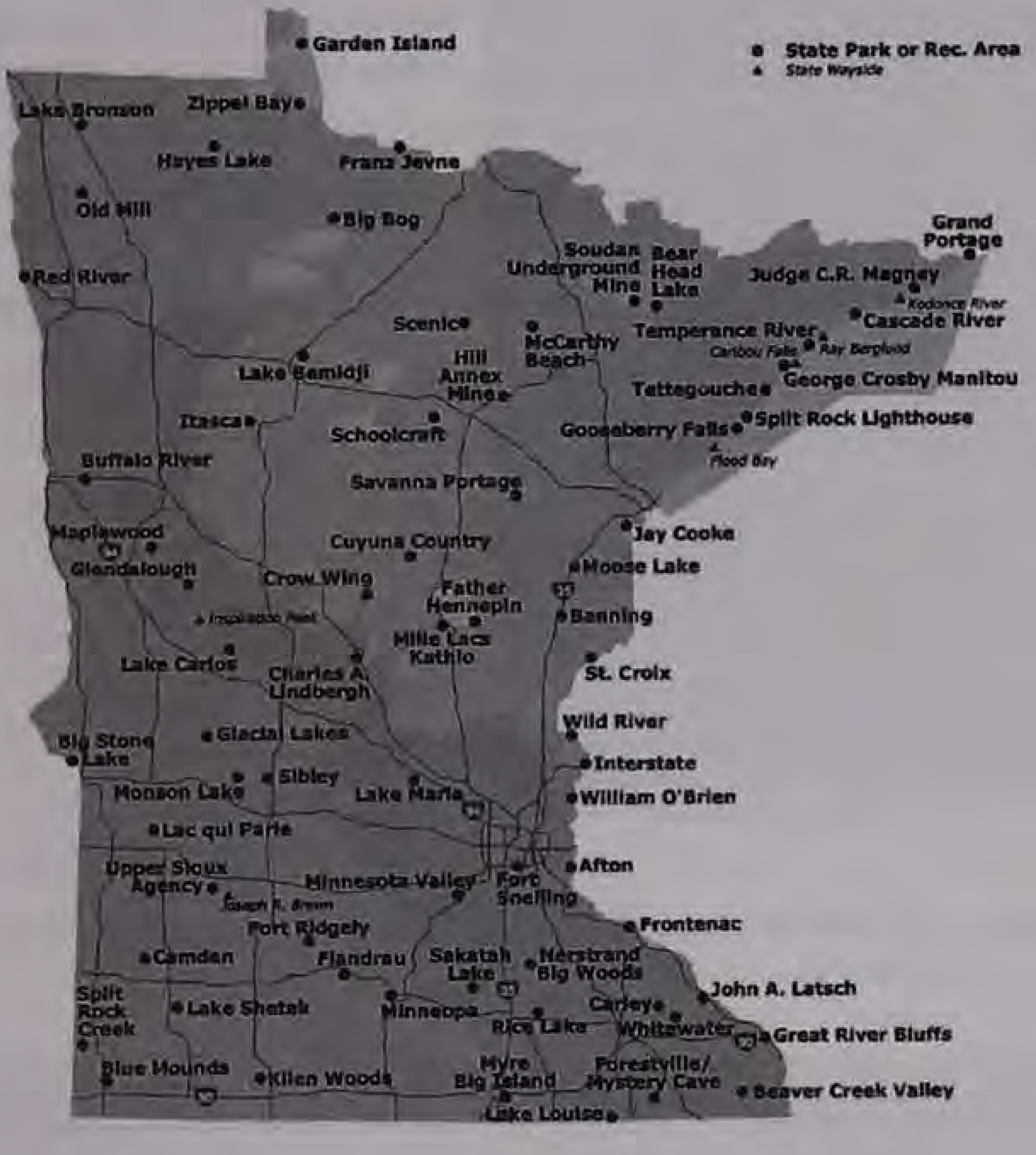

Figure 3. Minnesota State Parks.

for Allamakee County, T96N, R3W (Fairview Township); Clayton County, T95N, R3W (Mendon Township); T94N, R3W (Clayton Township); and T93N R3W (Garnavillo Township). The notes, which are on microfilm in the State Historical Society of Iowa Library and Archives in Des Moines, have also been photographed by Paul Anderson (1992–2010) at Iowa State University and may be available upon request.

WHO WERE THE EFFIGY MOUNDS SURVEYORS?

William Gordon surveyed Garnavillo Township (T93N R3W, 5th Principal Meridian, Wisconsin Territory) from May 15-May 25, 1838. Little is known about William Gordon. William Gordon served as Deputy Surveyor; the chain bearers were B. D. Lee, Ira Davis, and Robert S. Silcox; the flag-bearer was Daniel Adams.

Ira B. Brunson was born in Fowler, OH. In 1838, Mr. Brunson was an appraiser for Chippewa County, Wisconsin. From August 1-9, 1838, his crew mapped Mendon Township (T95N, R3W, 5th Principal Meridian, Iowa Territory). From May 29-August 31, 1839, his crew mapped Clayton Township (T94N, R3W, 5th Principal Meridian, Iowa Territory. In 1847, Mr. Brunson platted St. Paul, Minnesota. During 1853 and 1854, he served as Clerk of Court, Register of Deeds, and County Surveyor of Crawford County, Wisconsin. In 1856, Mr. Brunson surveyed the proposed site of the village of Nezeka at the site of the Effigy Mounds visitor center (Andreas Atlas 03). Ira B. Brunson served as Deputy Surveyor; the chainmen were H.D. Lee, Ian Davis; and the marker was C. Pettibone.

Guy H. Carleton was born in Bath, New Hampsire, November 1, 1819. He came to northern Michigan in 1845 to

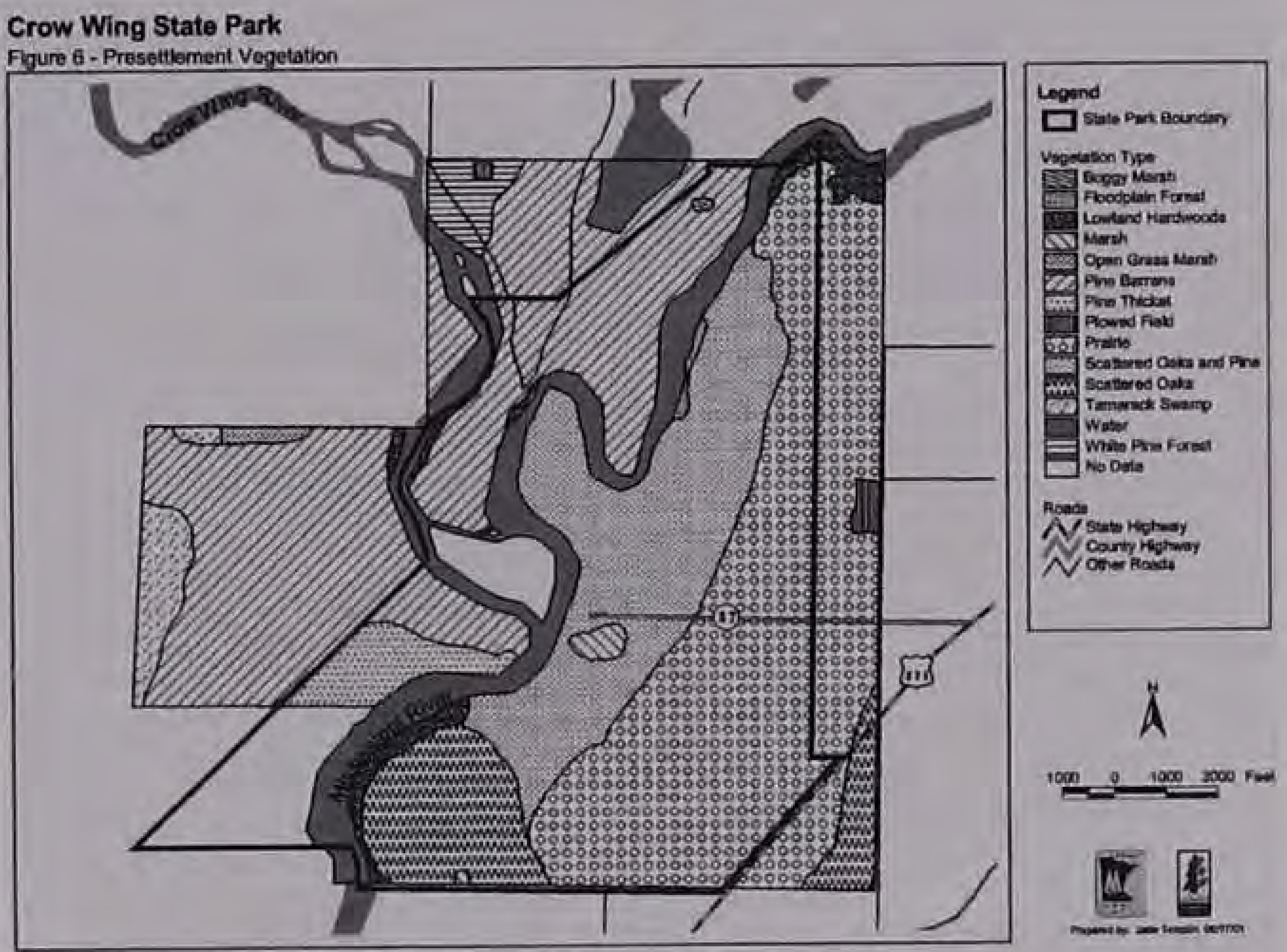

Figure 4. Government Land Office survey, 1870, Crow Wing State Park area.

fulfill a government contract for laying off of township lines ($10 old each) and subdivisions ($6 each). From March 31-November 22, 1849, his crew mapped Fairview Township (T96N, R3W, 5th Principal Meridian, Iowa). He died May 1, 1895. Guy H. Carleton served as Deputy Surveyor; the chainmen were Alexander Calhoun, Edwin D. Williams, David Wright and William Logan; the axemen were Joseph Stolts and James Maher; additional chainmen were Michael Dignan and Mathew Cullen; an additional axeman was Joe S. Anderson. Since these are listed on the same survey page for the township, it is unclear as to whether he had two separate crews or started with one crew and ended with another

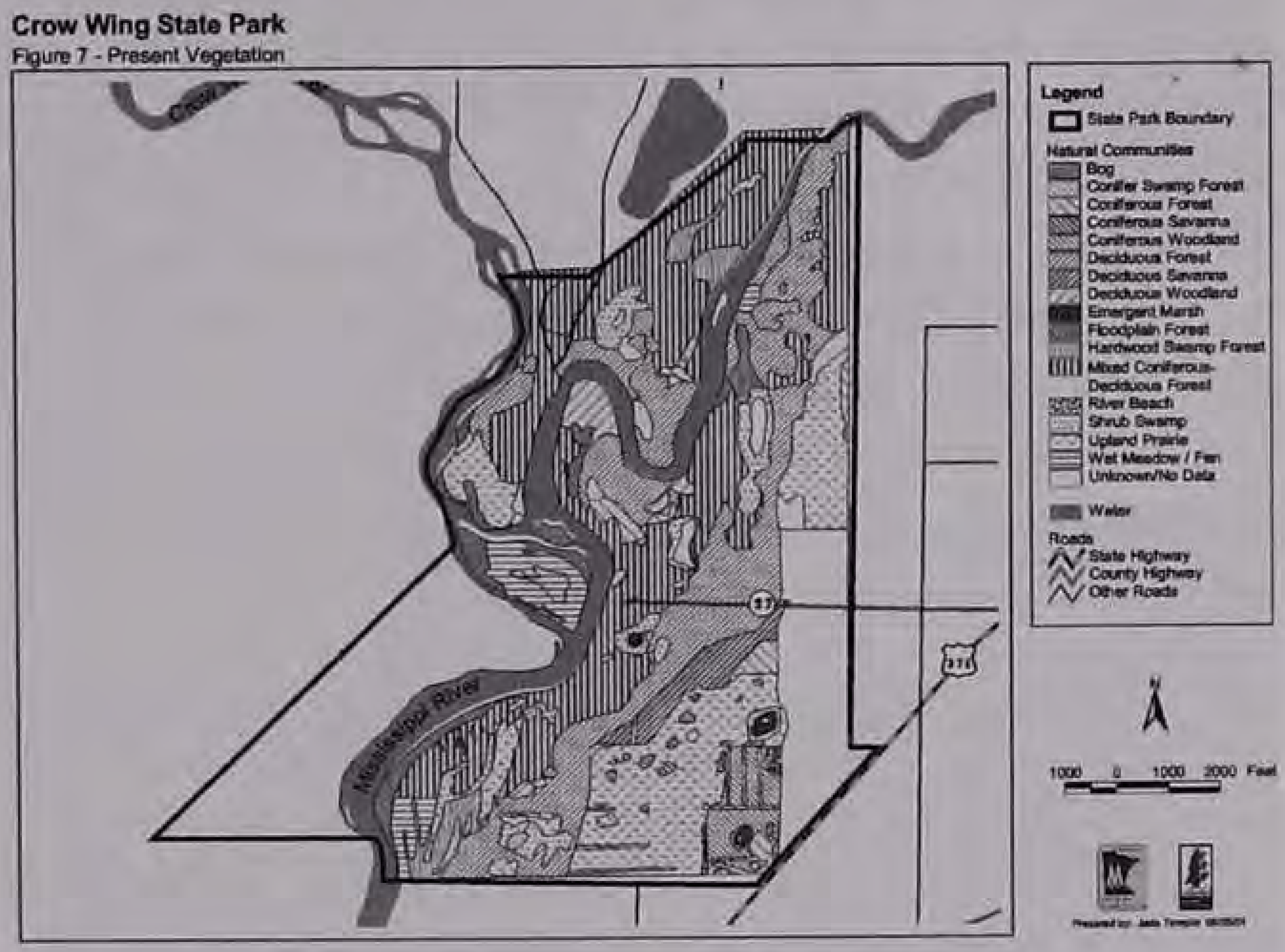

Figure 5. Minnesota DNR Forest Stand Inventory, 1979, Crow Wing State Park area.

Such information is more than merely interesting; knowing where surveyors came from, for example, may clarify terminology. A Minnesotan's "jack pine" was a "pitch pine" in New England, where many of the surveyors originated.

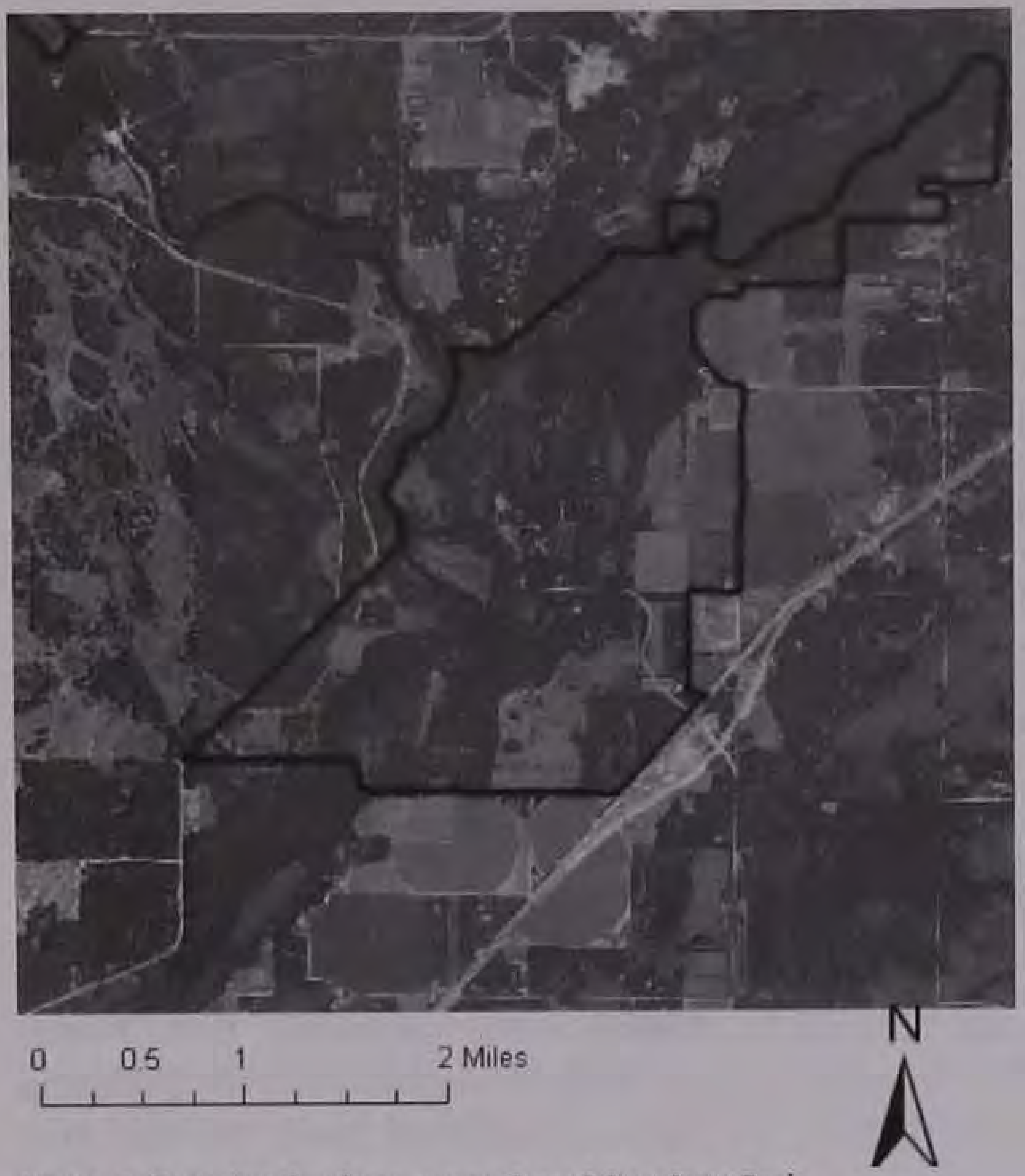

Figure 6. Digital orthophoto, 2000, Crow Wing State Park.

IOWA RESULTS

The plat maps of the Iowa townships have been scanned and rectified and are available in GIS format. The bearing trees (Figure 7) were mapped to include species, dbh, and any special characteristic (e.g., tri-pronged, etc.). The chains and links measurements were converted to feet for ease in mapping. In addition to using the plat maps of the townships, the county digital orthoquads (DOQs) were also used as a base map. All land features on the plat maps were verified with the line notes and were mapped. All items in the line notes were mapped under the following headings: vegetation type (vegtype), trees, undergrowth, and rating. Each line-note segment was assigned an ID number.

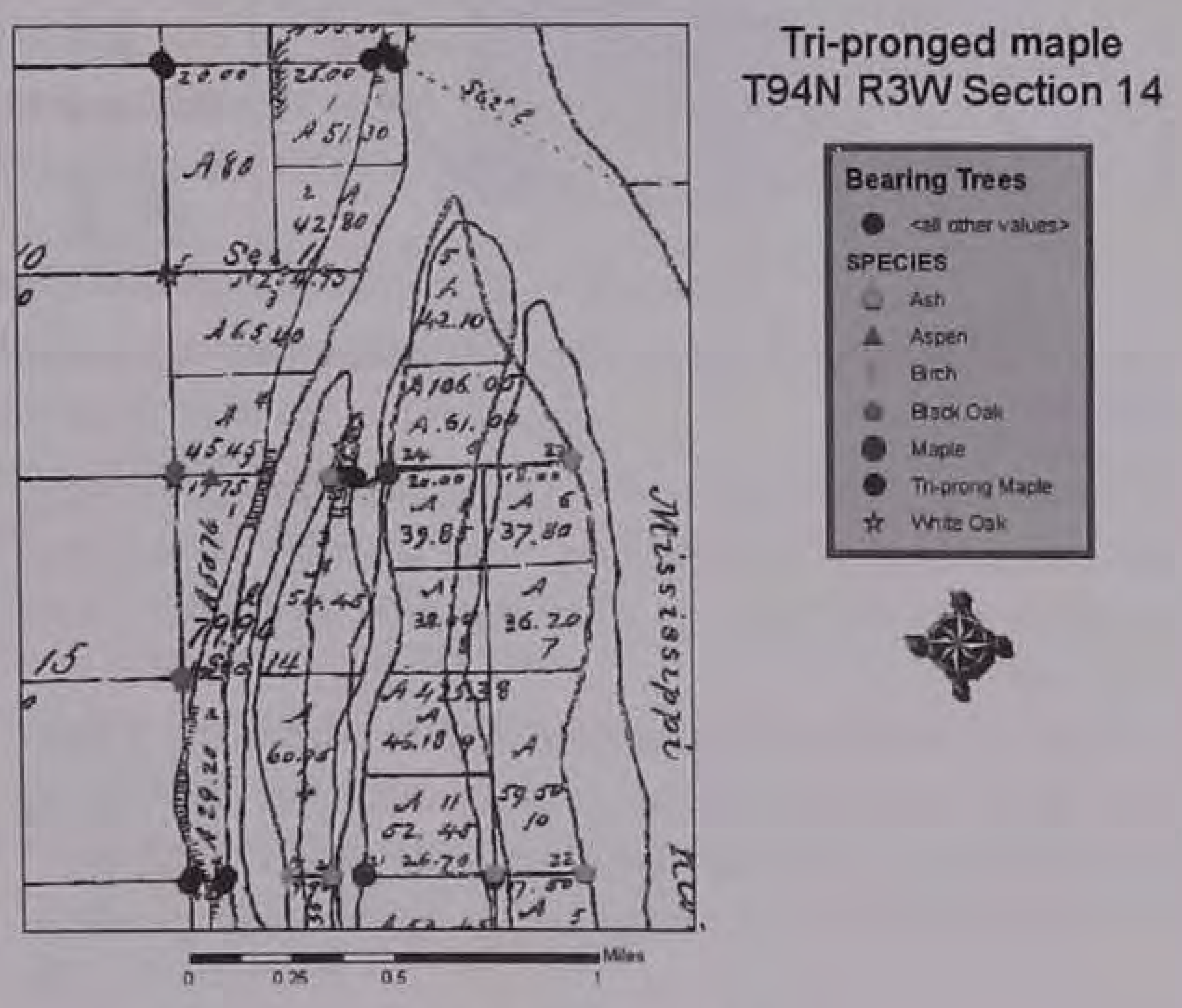

Figure 7. Location of tri-pronged maple.

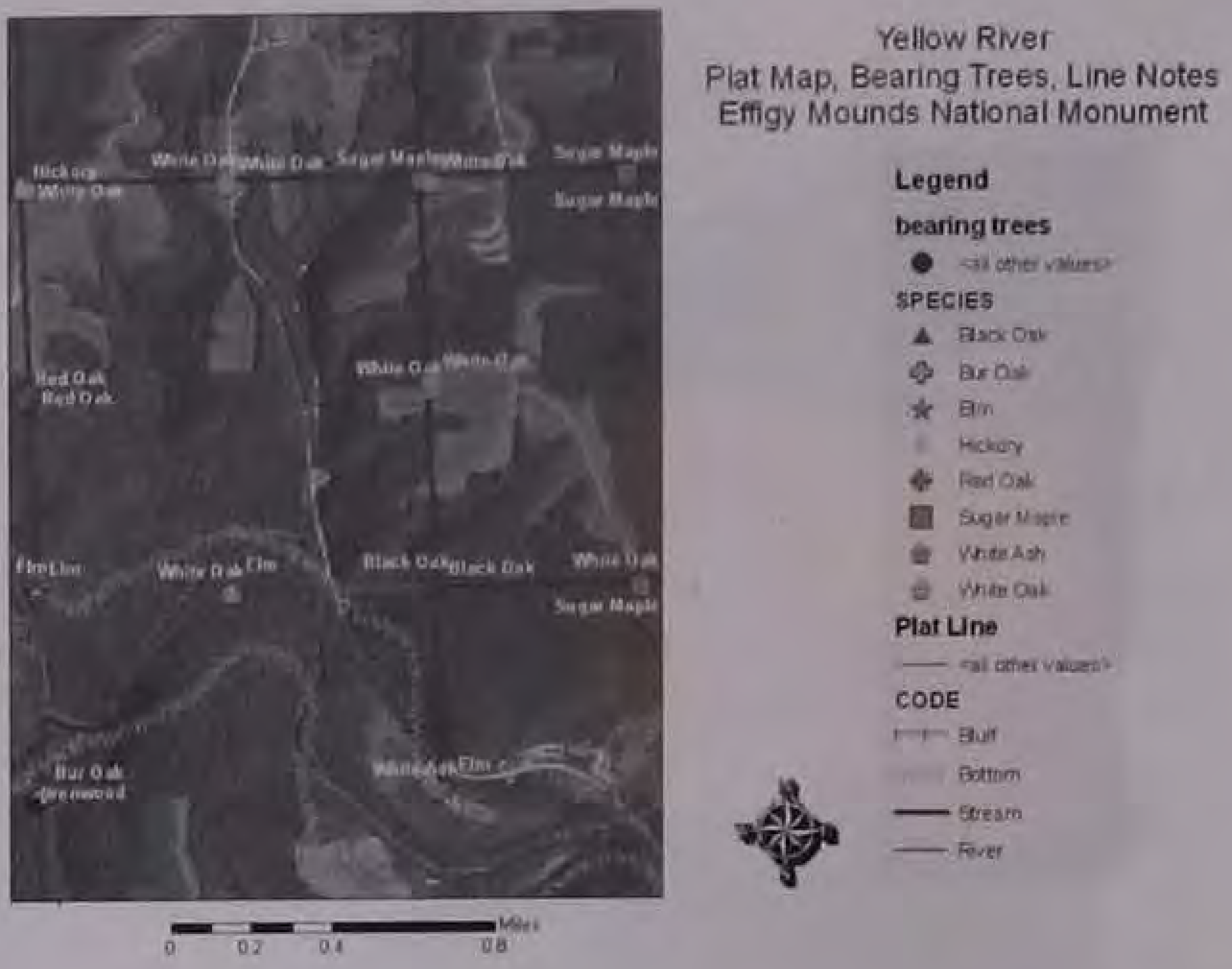

Figure 8. Line note no. 38 east between sections 28 and 33, T96N, R3W (pink east/west line).
Vegtype: Bottom
Trees: White oak and elm
Undergrowth: none
Rating: 2

CONCLUSION

The early U.S. General Land Office surveys, while never intended to be vegetation surveys, are nevertheless highly useful for present-day reconstruction of presettlement landscape vegetation. Modern technology— GIS plat maps and orthophotos in digital format—allows integration of the surveyors' notes and maps to produce a more detaled view of the general landscape of the time. Bearing trees, particularly important markers in the original notes, should be used as part of the picture, not as scientific data. All plat line information and all plat map information should be included and matched to the line notes.

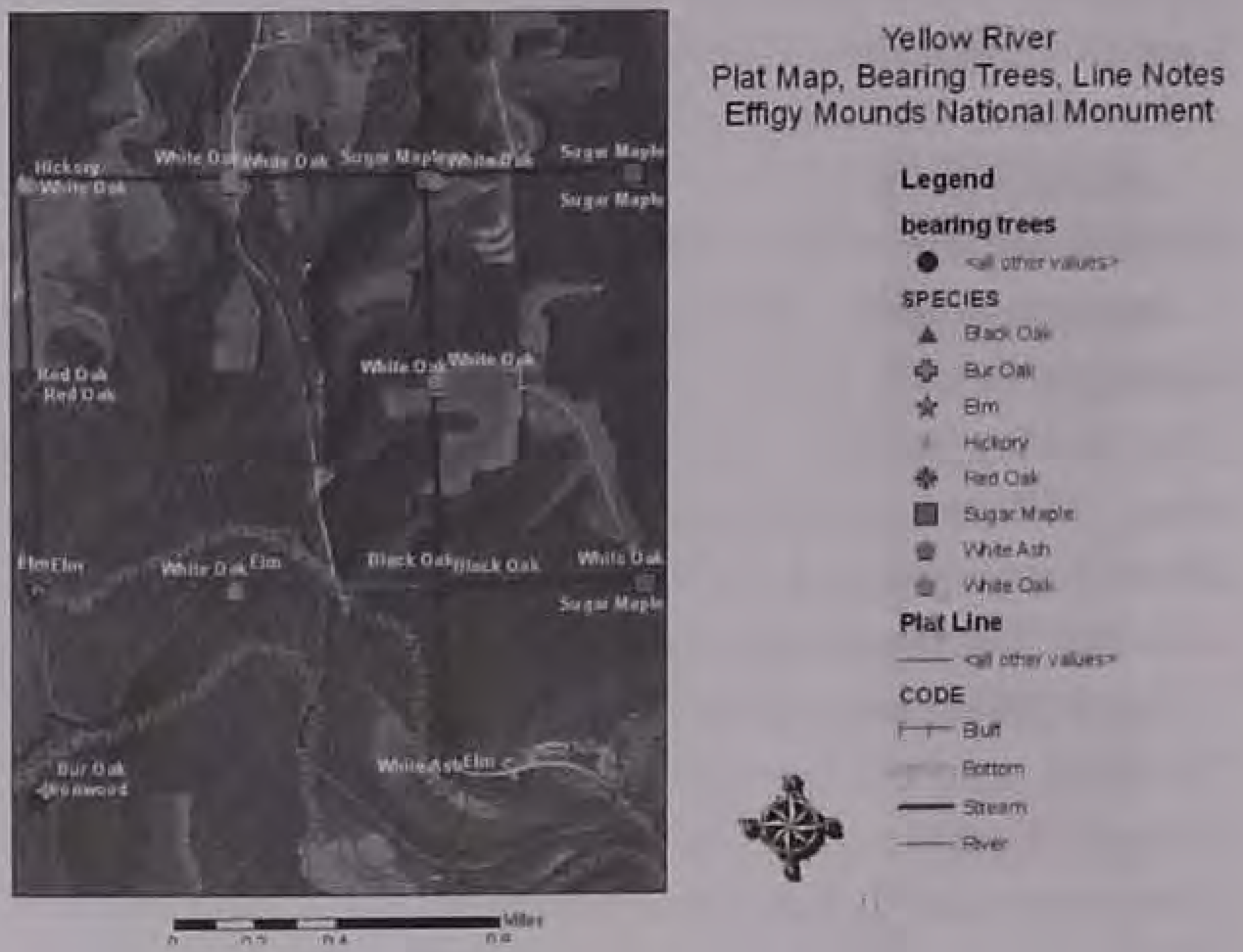

Figure 9. Line note no. 40 east between sections 28 and 33, T96N, R3W (east quarter: light blue line to right of line in Figure 8).
Vegtype: None
Trees: White oak, black oak
Undergrowth: None
Rating: 2
Land: Hilly

The analyst should not only be familiar with vegetation communities in the local area, but should learn who the surveyors were and where they came from, and know contemporaneous terminology.

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LESSONS LEARNED FROM THE GRASSLAND RESTORATION NETWORK, 2003-2010

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Abstract: The Grassland Restoration Network was formed in 2003 by The Nature Conservancy, along with many other partners. The original goals were to share expertise and experiences between large-scale prairie restoration (reconstruction) sites across the country, and to work together on shared challenges. Staff from those project sites, along with many other participants, have worked toward those goals through annual workshops and targeted peer review sessions. Considerable time has been spent defining the role of prairie restoration as a tool for restoring ecological function, and exploring ways to evaluate success in that regard. In addition, many discussions have focused on the kinds of techniques used by the diverse group of participants to establish prairie communities. Those techniques include harvesting, cleaning, storing, and planting seeds, propagating plants in nurseries, and both short- and long-term management of restored prairies. Participants have found that some methods, such as dormant-season broadcast seeding, are universally successful across all or most sites. In contrast, seeding rates that lead to successful prairie community establishment seem to be driven largely by unique local conditions. After eight years of existence, the Grassland Restoration Network has successfully built collaboration, shared experiences and expertise among project sites, and is investing in research projects to further build our shared knowledge about prairie restoration.

Key Words/Search Terms: prairie reconstruction, restoration, lessons learned, grassland restoration network

INTRODUCTION

The Grassland Restoration Network (GRN) is a loose affiliation of projects and project staff engaged in the restoration of diverse native grassland communities. The network was formed in 2003 by The Nature Conservancy and a wide variety of other conservation organizations, government agencies, and private landowners. There are three major objectives of the Grassland Restoration Network: 1. Facilitate communication and cross-site learning among large-scale grassland restoration sites. 2. Identify and close critical knowledge gaps regarding grassland restoration and measures of restoration success. 3. Foster a grassland restoration culture that increases the quantity and quality of grassland restoration.

The network sponsors annual workshops, generally in the summer or early fall, at which participants meet at a restoration site, tour nearby restoration projects, and discuss topics such as seed harvest and planting methods, plant propagation techniques, invasive species challenges, ideas for long-term prairie management, evaluation of restoration success, and research needs and results. In addition to these workshops, the network has facilitated several focused peer reviews of restoration projects through The Nature Conservancy's Conservation Audit Program. In these peer reviews, four to six experts from around the network converge at a single site and spend several days getting an in-depth look at the project's objectives, strategies, challenges, and evaluation strategies. At the end of the process, the review team presents the host site with both a detailed report on what the project is doing well and suggestions for possible adjustments or improvements.

WHY DO PRAIRIE RESTORATION?

While all participants in the GRN come to the group with their own specific reasons for doing prairie restoration, the network itself was organized to facilitate the use of high-diversity prairie restoration as a tool for increasing the ecological viability and function of grassland ecosystems. The most common examples of this occur when relatively small and/or isolated prairies are functionally enlarged or connected to others through the conversion of cropland to high-diversity grassland communities. Success in this case is not measured only by whether or not individual seedings have successful establishment of a diversity of plant species. More important, success is measured by whether or not the seedings increase the viability (long-term sustainability of ecological function) of the remnant prairie(s). To increase viability, those seedings must have a variety of functions beyond simply providing additional habitat for prairie plants and animals, including the facilitation of movement by animals, plants, pollen, seeds, and genetic information.

While high-diversity seedings are the best option when restoration is being used to promote the viability of the entire prairie community, some objectives can be met with lower-diversity seedings as well. For example, if the major goal of a restoration project is to increase the size of grassland patches for grassland breeding birds, the use of high-diversity restoration techniques may not be critical to the success of the project. In this case, the need is simply to make larger patches of grass-dominated vegetation, either to encourage breeding birds to nest in those patches, and/or to increase potential breeding success. Because grassland birds can breed successfully in monoculture fields of alfalfa, smooth brome, and other similar vegetation types, just adding some type of grassland vegetation to areas around and between existing grassland patches can be successful. However, the benefits of those lower-diversity seedings will be limited to grassland birds and a few other species with similar habitat needs.

When high-diversity prairie seedings are used to enlarge or connect prairie remnants for the benefit of the entire prairie community, measuring the success of the restoration project can be difficult. Establishing and maintaining plant diversity is the first key to success, because that diversity is important for building overall ecological resilience, helping to repel invasive species, increasing total grassland productivity, providing season-long resources for such groups as herbivores and pollinators, and meeting the needs of insects and other animals that require the presence of particular plant species for food, larval habitat, or other needs. Measuring the establishment and maintenance of plant diversity can be time consuming, but there are tools and techniques available for that kind of evaluation work. However, extending that evaluation to include measures of prairie function and ecological viability is much more difficult. Many sites have documented increases in the richness and/or abundance of grassland birds, reptiles and amphibians, insects, and other prairie animals, but measuring whether or not restoration efforts have increased the viability of those populations remains difficult. Finding ways to better measure the success of efforts to increase ecological function through prairie restoration continues to be a major point of discussion at annual GRN workshops.

LESSONS LEARNED

Apart from addressing challenges related to measuring prairie restoration success, one of the biggest strengths of the GRN to date has been the consolidation of experience and information regarding the kinds of restoration techniques used to establish diverse prairie plant communities across a wide range of geographic locations, soil types, and moisture conditions. Participants at GRN workshops have come from almost every grassland habitat type across North America, from the prairies of the Pacific Northwest to longleaf pine ecosystems in the southeastern U.S. The following is a synthesis of the kinds of techniques being employed by Network participants for harvesting, cleaning, storing, and planting seeds, as well as for both short- and long-term management of seedings.

SEED HARVEST

Many network participants successfully harvest seeds from over 200 plant species annually, and some harvests can include up to 400 species. Higher species counts generally include some level of nursery production of difficult-to-obtain species, and nursery production will be dealt with more specifically in the next section.

Seed harvest of most plant species is accomplished by hand, rather than through mechanized harvesters. Successful hand-harvesting of large quantities of prairie seed relies more on organization and efficiency than a large workforce. A small group of staff and/or volunteers can easily harvest a diverse mixture of seed sufficient to convert tens to hundreds of hectares of cropland per year if they follow several guidelines: 1. Develop and refine a comprehensive list of seed sources (remnant prairies, established prairie seedings, etc.) for each species to be harvested, along with approximate harvest dates—from late spring through the fall. 2. Harvest entire seed heads or plant tops, instead of individual flowers, and strap buckets or bags to your waist to free up both hands for harvesting. 3. Harvest from multiple seed sources to help ensure genetic diversity, but select sites where the target species is abundant and easy of access.

While hand-harvesting can gather sufficient seed from most prairie species, mechanical harvesters such as seed strippers and combines can be very helpful when harvesting large quantities of dominant grass species and other plant species found in large patches. However, mechanical harvesting is useful only when the area to be harvested is free of invasive species or there is a way to separate out the seeds of those invaders through subsequent seed cleaning. Some project sites employ seed-stripper machines that range in size from small rotating-brush machines pulled by ATVs to the larger Flail-Vac strippers mounted on loader arms of tractors. Besides dominant grasses, other species that can be efficiently harvested with seed strippers include minor grasses, various sedges and rushes, and some forb species that grow in dense patches.

Combines, such as those used in rowcrop harvesting, can be very useful for harvesting large quantities of seed but, in addition to the issues associated with seed strippers, can present challenges related to their mechanical complexity and size. Mounting a stripper head (aka rice head) on the combine, rather than a small grain head with a sickle bar and reel, can reduce many problems associated with passing large quantities of fluffy and/or stemmy material through the inner workings of a combine. When using a stripper head, most operators simply remove most or all of the screens and/or shakers from the rear of the combine and allow anything the stripper head picks up to go directly into the bin of the combine. If a small grain head is used, the material is cut, rather than stripped, and relatively long stems and more material overall must be passed through the machine and separated from the seed. Long stems can cause problems by getting wrapped around augers or other moving parts, and by simply jamming up the flow of material. Combines that use shakers and air to move material tend to work better than those that use internal augers, because stems and fluffy seed tend to "bridge" on top of augers and thus fail to be transported effectively. It can be tricky to adjust the amount of air needed to move material through the combine. Too much air blows much of the seed out the back of the combine, and too little air causes the material to jam inside the combine. Many sites have moved to the more expensive but easier-to-use stripper heads to avoid these issues.

Regardless of the type of head used on a combine, the final challenge is to unload the seed from the bin. Most combines unload with an auger along the bottom of the bin. Even when seed is harvested with a seed stripper, the fluffy seed can be very difficult to transport because it often bridges across the top of the auger. One method used by many operators is to stand a long PVC pipe of 20-30 cm (8-12 in) in diameter vertically in the bin while harvesting so that the seed fills in around the pipe. When it is time to unload the seed, the pipe is pulled from the bin, leaving a large tunnel all the way to the auger at the bottom. The operator can then use a thin wooden pole or PVC pipe to feed the seed gradually down the hole to the auger so that it feeds without bridging. An alternative to this is to simply build a wooden floor over the top of the auger and scoop the seed out of the bin with grain scoops. A third option is to build a conveyor belt system that unloads seed out the side of the bin.

NURSERY PRODUCTION

Many sites augment their wild harvest of seed with some degree of nursery production, often for species that are difficult to wild-harvest in adequate amounts or that rarely produce seed in the wild. For example, many early season grasses and forbs are difficult to find when seed is ripe, they produce small amounts of seed, they drop their seed upon ripening, and/or they occur in widely scattered small populations. Alternatively, many wetland sedge species and other plants rarely produce seeds in the wild at all, relying instead on asexual reproduction through rhizomes. Still other plant species are simply not common enough in wild populations to supply enough seed to plant large areas annually—and/or there are concerns about the impact of regular harvests of large quantities from those populations.

Nursery production normally begins in a greenhouse, where plants are started from seed in flats or smaller containers. There are a wide variety of techniques for starting seed and cultivating plants. For most species, simply planting seeds at shallow depths in soil and keeping the soil moist is a successful way to establish seedlings. Some project sites start seeds in flats and then transplant plants to individual containers (e.g., conetainers in trays) after they germinate and grow several leaves. Other sites start the plants directly in conetainers and thin out any extra plants as needed. Many times it can improve germination rates to start seeds in the late fall and leave them exposed to winter temperatures before bringing them into the greenhouse in the later winter or early spring.

Once plants have established, they can either be planted directly into prairies or put into seed production beds. Some sedges and other species that rarely produce seed and usually reproduce by rhizomes are most efficiently established in prairies/wetlands by direct planting. For the most part, seedlings can be successfully established in young seedings simply by plugging them into the site and watering them one time. In older seedings or remnant prairies, it may be necessary to suppress nearby vegetation (herbicides, mowing, or tillage) and water the plants more than once.

When putting seedlings out in seed-production beds, there are a number of challenges that must be addressed, including watering, weed suppression, herbivory from insects and larger animals, and seed harvesting. It is critically important to decide how each of these challenges will be addressed before production beds are established, because the layout of the beds should be determined by those answers. Kankakee Sands in Indiana uses a center pivot to water its expansive area of production beds, but many other sites use either soaker hoses or a traveling gun (a sprinkler head on skids reeled in slowly, such as those commonly used on athletic fields). Regardless of watering method, the design of the beds should fit that method.

Weed suppression can be a major challenge, and requires both forethought and vigilance. There are two basic methods of suppression used by most sites, herbicide and weed mats/mulch, and both are normally supplemented with hand-weeding. Sometimes herbicides that are selective enough to kill weeds but not the nursery plants (grass-specific or broadleaf-specific herbicides) can be used successfully, but weeds that are resistant to those herbicides typically become more abundant over time. Often, the use of preemergent herbicides can be more effective because they kill plant embryos as they emerge from seeds but do not affect more mature plants. A common method for starting new production beds is to kill any existing vegetation with herbicides and/or tillage prior to plugging in seedlings. Then as soon as the seedlings are in place, the bed can be sprayed with a preemergent herbicide to prevent any new seed germination (granular forms of preemergent, such as the kind of crabgrass preventer used in yards are also effective). Reapplication will be necessary through the season, the frequency depending upon the particular herbicide used. Once a preemergent has been used, any soil disturbance will break the "barrier" on the soil surface created by the herbicide, so hand-weeding or tillage should be minimized until it is time to re-apply the herbicide. During subsequent seasons, preemergent can be used to maintain low weed numbers. Timing of the first application will be determined through experience, and an initial round of weeding and/or tillage may be needed to eliminate any winter annual rosettes present before application.

The other major method of weed suppression is the use of either mulch or black plastic weed mats. When using mulch, seedlings are typically planted in tilled or pre-weeded beds and then surrounded with layers of mulch (wood chips, straw, etc.) 3-6 inches thick. The mulch is supplemented as necessary through subsequent seasons as it degrades. When using plastic weed mats, the matting material is laid out first, and seedlings are planted in small holes created in the material. Those holes can be cut with a knife or created by melting holes with a hot ring of metal attached to the end of a propane torch. Once seedlings are in place, the mat typically prevents weeds except in the holes themselves, so it is important to make holes large enough to allow the seedling plants to grow but small enough to minimize space for weeds. With at least some types of weed mats, the holes in the mats can be so small that the use of sprinklers to water the plants is ineffective; much of the water runs off rather than soaking through. Soaker hoses can be a more effective watering method.

Strategies for preventing herbivory and other damage from insects and larger animals vary by the species causing damage. Sometimes fencing is required to prevent access by deer, rabbits, or ground squirrels. But often those mammals target only a small subset of plant species in production, so only those beds being damaged need to be fenced. When insects are causing damage to flowers or seeds, pesticides may be effective, but other strategies can work as well. For example, damage to some plant species can be mitigated by cutting off the first round of flowers, forcing them to reflower several weeks later. This can sometimes break the cycle of insect damage because the insect is no longer present when the flowers eventually bloom again. In other cases, it may be worthwhile to put mesh bags over flowers of species which don't rebloom when cut, or for which seeds are valuable enough to make the time-consuming task worthwhile. Oftentimes, if the particular pest species can be identified, tips for avoiding or mitigating damage can be found on the internet, but if not, researching the natural history of the species may provide hints at prevention strategies. For example, vole populations can be kept at lower numbers by using prescribed fire to reduce litter and thatch levels in the beds and nearby areas.

Because plants in production beds often grow more robustly and produce more seeds than their wild-growing counterparts, it may take fewer plants in production than expected to provide a desired amount of seed. Often 500 plants can supply sufficient seed for a plant species to be well represented in more than 80 ha (200 ac) of seedings. It may be a good idea to start with small beds and leave room for later expansion if necessary. Some sites are experimenting with raised beds to see whether or not they can reduce weed pressure and make it easier to keep track of small-statured plants and/or plants that drop seeds soon after ripening. As should be clear from the above discussion, a seed production nursery can require a deep skill set and a lot of labor. Species should be selected for seed production only if they are truly hard to come by through other means. Starting small and growing gradually as you hone your techniques is usually the best way to go. In some cases, contracting with a commercial greenhouse/nursery to grow plant materials for you may be the best option. Regardless of whether you manage your own nursery operation or contract it out, it's important to ensure that seed used in the nursery represents multiple wild populations and/or genetic variations for those species. Otherwise, the seed produced can be very limited in the genetic diversity represented because of the artificial genetic bottleneck created by the nursery production process.

SEED CLEANING AND STORAGE

The extent to which seed cleaning is necessary often depends upon the planting method to be used. If the seed is to be passed through a seed drill or similar equipment, the seed needs to be clean enough not to clog that equipment. In contrast, if the seed is to be broadcast through a fertilizer spreader with an aggressive agitator, very little cleaning may be necessary. When weed seeds are a concern, they can be removed by cleaning with screens, fanning mills, or other equipment. In addition, non-viable seeds can be removed by cleaning processes that utilize air or water to separate those lighter seeds from the rest so that seeding rates of those species can be more accurately judged, if that is a concern.

Regardless of cleaning method, it is important to dry the harvested plant material immediately to prevent mold and to facilitate further cleaning. Many project sites simply dry seed by thinly spreading the harvested material on a hard dry floor—or on tarps, screens, bucket lids, etc., until dry. It may be necessary to turn the material periodically if it is not spread thinly enough to dry evenly. When drying large quantities of material, some project sites use commercial grain-drying bins or have designed smaller substitutes that force air through harvested material until it dries. One simple substitute can be made by piling seed on top of perforated pipes, that are hooked up to a blower fan.

The most critical part of the seed-cleaning process is to break up pods and flower heads to separate the seeds from each other. This is often done with some variation of a hammer mill. Most commercial seed hammer mills utilize whirling blades or brushes that break apart plant material until it is small enough to pass through selected sizes of screens below. Alternatives to hammer mills include leaf mulchers and other machines that pass seeds through heavy fan blades and knock seeds from the plants. Once the seed is separated from stems and pods, some sites consider the seed ready to plant, while other sites continue to clean with a combination of hand-screening and/or fanning mills. Again, the amount of cleaning depends largely on the method of planting to be used and any need to remove weeds or get accurate assessments of seed viability. Before and after seed is cleaned, proper storage conditions are important to prevent loss of seed viability. When possible, seeds should be stored in cool, dry conditions. Climate-controlled rooms can be useful, particularly for multiyear storage, but seeds of most prairie species can maintain their viability for a year or two even in uninsulated metal buildings, especially when stored in large piles or in paper sacks that allow them to "breathe" and that provide insulation. The seeds of some early spring-blooming plants can be the most vulnerable to loss of viability, even in climate-controlled conditions. Experimentation has shown that some of these species establish best when they are planted immediately after they are harvested. Finally, protecting seeds from mice and other animals can be an important consideration, both because of seed loss and health concerns due to fecal matter from mice (and the cats that follow them). Most-climate controlled rooms are also well sealed against these kinds of pests, but other storage buildings (e.g., barns) are not. Sectioning off storage areas with hardware-cloth barriers and keeping seed-drying areas swept clean can greatly help to reduce exposure of seeds to mice.

SITE PREPARATION AND PLANTING

Proper preparation of a restoration site is very important. Sites with an abundance of weed seeds in the soil, and small or narrow sites surrounded by invasive species and/or woody plants, can result in failed restoration efforts even when everything else is done well. Taking the time to eliminate, or at least greatly reduce, invasive-species threats prior to seeding will save countless hours of labor later, and perhaps spell the difference between eventual success and failure.

At the 2008 Grassland Restoration Network workshop the participants were able to agree upon a consistently successful seeding technique that works across the mixed-grass and tallgrass prairie locations represented at the workshop. Excepting extraordinary circumstances, the experience of network participants shows that a dormant-season broadcast seeding onto Roundup Ready soybean stubble will always establish a diverse prairie plant community. That said, there are countless ways to achieve success, and establishment varies greatly from year to year and site to site even when identical methods are employed.

While it is significant that participants from across the country agreed that a certain set of factors leads consistently to success, that doesn't necessarily mean that seedings done in other ways will not also succeed. For example, planting into soybean stubble is becoming increasingly popular among network project sites, but is certainly not the only seed bed that works. Many successful seedings have also been established following corn harvest, although those sites are typically disked or burned/harrowed prior to seeding to smooth out the ridges and allow good seed/soil contact. What appears to be more important than the type of crop harvested from the field prior to seeding is the timing of that seeding. Seeding in the winter or early spring following harvest has become the most common timing, but seedings into late spring or even early summer can bring success, although sometimes the establishment can be slower. What does not appear to work well is seeding into a field that has been idled for a season or more. Seeding into old field conditions tends to result in higher than acceptable weed pressure, even if the field is tilled prior to seeding. Finally, the use of cover crops has fallen out of favor among network participants because of poor success. Often, cover crops are found to fail at the two primary purposes for which they are employed—suppression of weed pressure and provision of fuel to carry a fire. Moreover, they can sometimes compete with establishment of desirable plants as much as would the weeds they displace.

Seeding rates vary widely between sites. However, most sites have been moving toward lighter seeding rates of major warm-season grasses to keep those species from either becoming dominant before forb diversity is well established or outcompeting forbs later. Some sites are using total PLS seeding rates of 1-2 kg/ha (2-4 lbs/acre) for the "big three" warm-season grass species combined—big bluestem (*Andropogon gerardii*), Indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*). Other sites are experimenting with eliminating those dominant, strongly rhizomatous species from initial seedings but including native bunchgrasses and other less dominant grasses for species and structural diversity. The other warm-season grass species can then be added after forbs and minor grasses are well established—or they may come in on their own if they are present in adjacent areas. While early results from this kind of experimentation seem to be largely positive, there are few examples of mature seedings from which to draw firm conclusions. In some cases, weedy species, including some goldenrod (*Solidago*) species can become alarmingly abundant in the absence of competition from dominant rhizomatous grasses.

Total seeding rates vary from west to east, and depend largely upon the weed pressure at the site. For example, in central and eastern Nebraska, sites are seeding at rates of about 1-2 kg/ha (2-4 lbs/acre) of the big three tallgrasses and 0.5-1 kg/ha (1-2 lbs/acre) of forbs, minor grasses, and sedges. Such a low seeding rate prolongs the establishment period, giving forbs time to establish before the big three grasses become dominant. Low seeding rates are also popular among organizations attempting to create habitat for wildlife species such as ring-necked pheasants, because annual weeds remain abundant in seedings for up to three or four years.

By contrast, in the eastern portions of the tallgrass prairie, and where rainfall and soil organic matter levels are higher, prairie seeding rates tend to be much higher as well—up to 100 kg or more of bulk seed per hectare (40 lbs/ac), with little or no seed for the big three tall grasses, and with low rates of some weedy native species, such as wild bergamot (*Monarda fistulosa*), gray-headed coneflower (*Ratibida pinnata*), etc. In those areas, the high seeding rates help to establish forbs and bunch grasses quickly to compete with perennial weeds such as birdsfoot trefoil (*Lotus corniculatus*), Canada thistle (*Cirsium arvense*), sweet clover (*Melilotus* spp), and others. One trap into which some have fallen, and now regret, is the use of high seeding rates of rhizomatous tall grasses in order to help battle weeds. While that kind of seeding can help outcompete weeds, it also results in a stand dominated by grasses—and low overall plant diversity. Trying later to increase forb abundance and diversity in a prairie restoration dominated by tall grasses has been found to be extremely difficult.

Overall, seeding rates appear to need determination by experimentation at each site, rather than by generalizations based on soil or moisture conditions. Necessary seeding rates tend to increase with soil productivity, annual rainfall, and perennial weed pressure, but there is great variation within that continuum. Utilizing multiple small experimental seedings to refine seeding rates during the early years of a restoration project pays great dividends. Most important, necessary seeding rates should help determine the size of area seeded annually (based on the amount of seed available) rather than the other way around.

While most of the GRN's efforts have dealt with converting cropland to prairie, some sites are also experimenting with the restoration of severely degraded prairies and/or tame pastures by removing unwanted vegetation and seeding prairie species. In the Midwest and northern Great Plains, the first challenge is usually to suppress or eliminate dominant cool-season exotic grasses. Spraying with glyphosate herbicides in the late fall (after the first hard freeze) and/or in the early spring—when warm-season vegetation is dormant—can often be successful, although it may take repeated effort. It is also necessary to burn, harrow, or disk, or to use some combination of those soil preparation measures, to allow seed soil contact and provide light to new seedlings. Results have been positive from these kinds of restoration efforts, but there is still much to learn about the establishment, and particularly the long-term maintenance, of these seedings.

POSTPLANTING TREATMENT

It is common to mow first-year prairie plantings to keep the agricultural weeds short and provide light to the tiny prairie seedlings emerging late in spring, especially on sites where productive soils and high rainfall amounts promote particularly vigorous weed growth. Some sites have found that they need to mow only when the weed density is so high as to cause discoloration of the prairie seedlings or to prevent light from reaching the soil. Large patches of giant ragweed have been found to be particularly competitive with prairie plants, but many other annual weeds present much less of a problem, particularly in sandy soils and/or drier climates. Regardless of whether or not a site is mowed during the first growing season, dormant-season burning or mowing prior to the second field season can remove tall weed stalks and make access to the site for weed control and/or other purposes much easier—and may also help to further speed up establishment. At least for sites east of the Mississippi, a prairie planting should be weeded aggressively for the first three years. This can involve intensive measures such as carefully patrolling a planting to remove or spray invasive plants like sweet clover and birdsfoot trefoil. Some sites also remove weedy plants that might not be on an invasive list but can increase if not treated, including plants like red clover (*Trifolium pretense*), yarrow (*Achillea millefolium*), and Queen Anne's lace (*Daucus carotus*). In more western sites, perennial weeds tend to compete less well with prairie vegetation and may not require control unless it is required by noxious-weed laws. In other cases, mowing species like Canada thistle can sometimes provide adequate control by suppressing their growth and reproduction until perennial prairie plants are established sufficiently to compete with them.

Just as with seeding rates, postplanting weed management needs appear to vary greatly from site to site. Weed species that are not a problem in one site can be a severe problem in another. Generally speaking, weeds are less of an issue in more western and drier sites, but that is not universally true. Small-scale experimentation with various levels of weed management effort during the early years of a long-term restoration project can tell you much about what will be required as the project grows in size. Along with the limitations of seed availability, required levels of weed management effort should be used to determine the size of the area planted annually. Smaller acreages of good seedings, with adequate plant diversity and manageable weed pressure, will add up to success much more quickly than large acreages of low plant diversity and/or unmanageable weed problems.

LONG-TERM ADAPTIVE MANAGEMENT

Seedings can vary tremendously in the amount of work required to maintain their long-term plant diversity and ecological function. Some of the differences are related to geographic location, such as the higher weed pressure normally found in higher-productivity soils and with relatively high rainfall and/or soil moisture. Others can be related to the seeding density, cropping history, weed and tree pressure, and other factors. Tapping into the knowledge base from nearby restoration efforts can help both to design restoration strategies that will be successful and to avoid pitfalls that others have discovered. Regardless, because of the inherent variability in prairie restoration, it is smart to experiment with small seedings—using a variety of methods—and let them establish for several years to gauge the management needs they will require before starting to plant larger acreages.

Suppression of invasive species is typically the management strategy that requires the most time and energy in restored prairies—just as in remnant prairies. As discussed earlier, intelligent site selection and preparation can be very helpful in preventing severe invasive species issues. Removing trees and invasives from within or around the edges of sites to be restored can make a big difference. Avoiding seedings in old field conditions and working to reduce the seed bank abundance of weedy species prior to seeding can also be helpful. Once a site is seeded, however, the key to suppressing invasive species is quick and consistent action. Getting infestations under control before they get too large is critically important, and well worth the investment of time and resources. It is also necessary to follow up with repeated control efforts and using GPS technology or other mapping techniques to ensure that all infestations are hit until they are eliminated. Again, planning for weed-management resource needs should be incorporated into up-front cost estimates for any restoration project, to ensure that those resources are available when they're needed. Finally, many sites have had to make the difficult decision to start over on seedings where invasive species and/or poor establishment of prairie plants has made continued stewardship costs higher than restarting the restoration process from a clean slate.

Another issue that plagues many established seedings is a gradual increase in the dominance of a relative few species (often grasses) and an associated loss of overall plant diversity. This seems to happen more quickly in productive soils and wet climates, but is not exclusive to those site types. Seeding with lighter seeding rates of dominant species can help delay or even prevent those issues in many cases, as can using a diverse mix of less-dominant grass species to take their place. In other cases, it can be necessary to take action to reduce the dominance of those species. A variety of methods have been tried, including mowing, disking, herbicide application, and grazing. Sometimes a light stocking rate of cattle can produce favorable results, because the cattle tend to select grasses over forbs and can help tip the balance of competition toward forbs. Specifically, patch-burn grazing is being used successfully in Nebraska to reduce grass dominance and maintain plant diversity, but it is just now being tested further to the east. Overall, the requisite long-term management of seeded prairies varies by site, and involves consistent evaluation and adaptation of strategies—just like the management of any other prairie.

Regardless of the variation in challenges faced by sites, the universal keys to success are consistent evaluation and adaptive management. Evaluation strategies should be tied to the original objectives for the restoration project. If the objective is to increase habitat for grassland nesting birds, the abundance and nesting success of grassland birds should be measured. However, if the objective is to increase the size and viability of the larger prairie community, indicators of that success should be identified and measured. For example, populations of species in adjacent remnant and restored prairies could be assessed to determine whether or not the restoration is acting as an extension of the remnant habitat. Whatever the restoration objective, evaluation should be a regular part of long-term management plans. Year-to-year climatic variation makes it difficult to assess progress within a short time window, so site managers should plan to repeat measures in order to establish trends. Management should then be adapted to address whatever trends are seen, whether those trends indicate changes in invasive species abundance, plant or insect diversity, or more complex markers of ecological function.

Finally, because the kind of restoration discussed in this paper is ultimately designed to improve the viability of prairie remnants, it's important to remember the management needs of those remnants. Allowing remnant prairies to degrade in quality because resources and attention have been siphoned off to deal with new restored prairies nearby is a conservation failure. As mentioned multiple times, starting a restoration project with small experimental seedings allows a site manager to gauge the effort needed to achieve success with those seedings—and it can also help determine whether the site has enough staffing capacity to restore new sites while maintaining existing ones. A successful restoration project enlarges and connects remnants by providing diverse native communities around and between them. However, poorly restored prairies, with abundant invasive species, can actually increase problems for the remnant prairies the restoration project was designed to improve.

RESEARCH NEEDS

In addition to sharing lessons learned, Grassland Restoration Network participants have also worked together to identify critical knowledge gaps and research needs related to prairie restoration. In some cases, multisite collaborative research projects have already been developed to begin addressing those questions in ways that would have been impossible before the GRN was established. In other cases, questions remain, but are at least defined sufficiently that they can be addressed when the necessary funding and/or capacity is identified.

Some research questions are related to prairie restoration techniques. The following are examples:

- When is it most effective to plant seeds from spring wildflowers? Initial work indicates that at least some difficult species may establish better when planted immediately after seed harvest.
- Which site conditions determine the seeding rates and ratios (e.g., grass to forb ratio) necessary to establish diverse plant communities?
- Which techniques work best when attempting to increase plant species richness/diversity in low-diversity restored prairies or degraded remnant prairies?

Other questions address broader issues regarding the use of prairie restoration as a conservation tool. Examples of those include the following:

- What is the correlation between the plant diversity of a prairie restoration and the conservation benefits it provides to species and ecological function? How does plant species diversity affect pollinators, invertebrate populations, resistance to invasive species, soil faunal communities, etc.?
- How well do restored prairies around and between fragmented remnant prairies act to enlarge and/or connect those prairies? Which factors influence processes such as species dispersal and pollen/gene flow from the remnant to/through the restored prairie? Do restored prairies positively or negatively affect weed pressure on adjacent remnants?
- What are the ramifications (positive and negative) of using local-ecotype seed versus seed from outside the immediate geographic area? What constitutes "local" for prairie species?
- How large do restored/remnant prairie complexes need to be to preserve the viability of populations and ecological functions within them? What management techniques are needed to prevent the loss of species diversity and/or invasive species encroachment in restored prairies over time?

CONCLUSION

During its first eight years, the GRN has had significant achievements. Chief among them was simply improving communication among restoration sites around North America regarding lessons learned. Participants in the network have gained considerably from the experience of others doing similar work across broad geographic locations, and many of the lessons learned are captured in this report. In addition, participants are working together on research and evaluation projects that test assumptions and address challenges related to the use of prairie restoration as a conservation tool.

The degradation and fragmentation of grasslands across North America has made immediate and efficient conservation action necessary. The Grassland Restoration Network continues to facilitate collaboration and exchange of ideas and experiences between those working to restore and conserve prairie. Harnessing the collective experience and capacity of all of those working on prairie restoration is the best way to achieve lasting conservation success.

NATIVE TALLGRASS PRAIRIE RECONSTRUCTION AND MANAGEMENT AT NEWPORT CHEMICAL DEPOT, INDIANA

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Abstract: Tallgrass prairie reconstruction at the U.S. Army's Newport Chemical Depot began in 1994. Inspiration for this project was garnered from a 1993 Indiana Division of Nature Preserves flora survey and their subsequent 1994 report which stated: "A large acreage of the southwestern portion of NAAP (Newport Army Ammunition Plant) was formerly mesic silt loam prairie. The Vermillion County Soil Survey showed prairie soils for this area, old Indiana county maps showed a large area of prairie, and we observed Big Bluestem and Prairie Dock along the road in the area. A restoration this large (1900 +/- acres) in this part of the Midwest is an exciting opportunity. There are no remnants in Indiana of prairies of the size of this potential restoration. The largest prairies that do remain in the state are sand prairies; thus the significance of this area is even more important because it is silt loam prairie restoration." Therefore, in the spirit of good land stewardship, the depot voluntarily contracted with Peter Schramm Prairie Restorations for ten years between 1994 and 2005 to plant 336 acres of tallgrass prairie. All of these acres were former presettlement tallgrass prairie soils that had been producing corn/soybeans in recent years. Using some of the proceeds from the remaining depot agricultural tracts the Depot spent nearly $128,000 for site preparation, seed, and planting. A total of six species of grasses and fifty-four species of forbs were planted. Management of this reconstructed ecosystem has consisted mainly of prescribed burning in cooperation with the Indiana Division of Forestry.

Key Words / Search Terms: Peter Schramm, Army, prairie reconstruction, Newport

INTRODUCTION

The Newport Chemical Depot (NECD) is a U.S. Army installation that was originally established as the Wabash River Ordnance Works (WROW) in 1941. Previously, the installation has also officially been called the Newport Army Ammunition Plant (NAAP) and Newport Chemical Activity (NECA). NECD is in a unique approximately 7,100-acre area of west-central Indiana. NECD is located in Vermillion County between two natural regions of west-central Indiana climate—the Grand Prairie and the Central Till Plain. The Grand Prairie is characterized by its dark and fertile soils and the Central Till Plain is known for its nearly flat to gently rolling landscape. The Depot lies along the boundary between the Grand Prairie and Entrenched Valley Sections. The Wabash River dominates the area where these regions meet; however, the boundary is not well defined, but rather a mosaic of bluestem prairie and oak-hickory forest. This edge effect has increased the diversity of flora and fauna at NECD (Hedge and Bacone 1994). In 1993, the NECD Operating Contractor, Mason and Hanger Corp., contracted with the Indiana Division of Nature Preserves to complete the report "Inventory of Natural Areas and Rare Plant Species within the Newport Army Ammunition Plant." The report provided inspiration to start reconstruction of the Newport Chemical Depot Tallgrass Prairie, stating, "A large acreage of the southwestern portion of NAAP was formerly mesic silt loam prairie. The Vermillion County Soil Survey showed prairie soils for this area, old Indiana county maps showed a large area of prairie, and we observed Big Bluestem and Prairie Dock along the road in the area. A restoration this large (1900 +/- acres) in this part of the Midwest is an exciting opportunity. There are no remnants in Indiana of prairies of the size of this potential restoration. The largest prairies that do remain in the state are sand prairies; thus the significance of this area is even more important because it is silt loam prairie restoration" (Hedge and Bacone, 1994). In recognition of the Grand Prairie's importance to Indiana's history—and more specifically, the ecology of NECD—the NECD successfully reconstructed a portion of NECD to presettlement prairie conditions through a series of prairie reconstruction projects begun in 1994 and continuing until 2005. The prairie reconstruction projects were halted after NECD was placed on the 2005 Base Realignment and Closure (BRAC) List and subsequently scheduled for closure.

PLANTING THE PRAIRIE

A total of approximately 336 acres was removed from row-crop production and converted to tallgrass prairie. Peter Schramm of Peter Schramm Prairie Restorations was contracted to prepare the seedbed, supply the seed, and plant all of the acreage with a Nesbit drill in the spring of the year. Local farmers prepared the seedbed. Schramm is a retired Knox College (Galesburg, Illinois) professor who pioneered prairie restoration techniques and has over 40 years of experience in the field. The total contracted cost of the 336-acre prairie reconstruction was $127,589; and it was paid entirely from a small portion of the proceeds from NECD's annual agricultural leases (Table 1).

Table 1. Area planted and total contracted cost for Newport Chemical Depot Prairie, 1994-2005.

YEAR	ACRES	$
1994	7.2	7,850
1995	19.0	9,757
1997	22.5	11,000
1999	30.7	12,790
2000	47.2	14,036
2001	43.4	16,500
2002	62.2	20,100
2003	23.2	7,905
2004	35.6	12,139
2005	44.7	15,512
TOTALS (10 YEARS)	**335.7 ACRES**	**$127,589**

Only in 1994 was herbicide (glyphosate) used to assist with the seedbed preparation. In all of the subsequent years, only shallow tillage methods were used. A total of six species of grasses and fifty-four species of forbs were planted (Table 2).

Table 2. Species planted in the Newport Chemical Depot Tallgrass Prairie.

GRASSES	
Andropogon gerardi	big bluestem
Andropogon scoparius	little bluestem
Bouteloua curtipendula	sideoats gramma
Panicum virgatum	switchgrass
Sorghastrum nutans	indian grass
Sporobolus heterolepis	prairie dropseed
FORBS	
Amorpha canescens	lead plant
Asclepias tuberosa	butterfly weed
Aster azureus	sky-blue aster
Aster laevis	smooth blue aster
Aster novae-angliae	New England aster
Baptisia austrailis	blue false indigo
Baptisia leucantha	white false indigo
Ceanothus americanus	New Jersey tea
Coreopsis lanceolata	lanceleaf coreopsis
Coreopsis palmata	prairie coreopsis
Coreopsis tripteris	tall coreopsis
Desmodium canadensis	showy tick trefoil
Desmodium illinoense	Illinois tick trefoil
Echinacea pallida	pale purple coneflower
Echinacea purpurea	purple coneflower
Eryngium yuccilfolium	rattlesnake master
Euphorbia corollata	flowering spurge
Fragaria virginica	wild strawberry
Gentiana flavida	yellow gentian
Heliopsis helianthoides	oxeye false sunflower
Heuchera richardsonii	prairie alum root
Lespedeza capitata	round-headed bush clover
Liatris aspera	rough blazing star
Liatris pycnostacya	prairie blazing star
Lithospermum canescens	hoary puccoon
Monarda fistulosa	wild bergamont
Parthenium integrifolium	wild quinine
Penstemon digitalis	white beardstongue
Petalostemum candium	white prairie clover
Petalostemum purpureum	purple prairie clover
Physostegia virginiana	false dragonhead
Potentilla arguta	prairie cinquefoil
Psoralea onobrychis	French grass
Pycnanthemum virginianum	mountain mint
Ratibida pinnata	yellow coneflower
Rosa carolina	prairie rose
Rudbeckia hirta	black-eyed Susan
Rudbeckia subtomentosa	sweet black-eyed Susan
Ruellia humilis	wild petunia
Salivia azureus	pitcher's sage
Silphium integrifolium	rosinweed
Silphium laciniatum	compassplant
Silphium perfoliatum	cupplant
Silphium terebinthinaceum	prairie dock
Sisyrinchium campestre	blue-eyed grass
Solidago nemoralis	old field goldenrod
Solidago ohiensis	Ohio goldenrod
Solidago rigida	stiff goldenrod
Solidago speciosa	showy goldenrod
Tradescantia ohiensis	common spiderwort
Viola papiloniacea	common blue violet
Vernonia fasciculata	ironweed
Veronicastrum virginicum	Culver's root
Zizia aurea	golden Alexanders

PRAIRIE MAINTENANCE

Prescribed burning in the spring has been the primary means to protect the prairie from weedy invasions. The Indiana Division of Forestry has played a major role in conducting the prescribed burns as the schedule of NECD would allow. All of the areas have been burned at least once, except for the 47.2 acres planted in 2000. In 2007 and 2009 NECD partnered with the Indiana Division of Forestry and Sycamore Trails RC&D to train local firefighters, natural resources professionals and the general public in how to manage prairie with prescribed burning. In addition, Mason and Hanger Corp. has mowed wildlife travel lanes in the larger tracts of prairie that were planted in 2001 and 2002. Also, approximately 91 acres of prairie that was planted in 2003, 2004, and 2005 was leased and cut once per year for prairie hay in 2007 and 2008. The forb-rich areas adjacent to the main roads were not allowed to be cut for prairie hay.

WILDLIFE OF THE PRAIRIE

The main reason for reconstructing the NECD Tallgrass Prairie was to increase the abundance and species diversity of wildlife in west-central Indiana. The significant decline of grassland bird species is well documented in reports such as the multiagency 2010 State of the Birds Report (North American Bird Conservation Initiative). Many grassland bird species (most declining) have been documented at the Newport Chemical Depot. State endangered grassland bird species that have been documented at NECD are as follows: Henslow's sparrow (Ammodramus henslowii), upland sandpiper (*Bartramia longicauda*), northern harrier (*Circus cyaneus*), sedge wren (*Cistothorus platensis*), Virginia rail (*Rallus limicola*), and peregrine falcon (*Falco peregrinus*) (Table 3).

Table 3. Grassland Bird Species of the Newport Chemical Depot.

GRASSLAND BIRD SPECIES OF THE NEWPORT CHEMICAL DEPOT	
Accipiter cooperii	Cooper's hawk
Accipiter striatus (SC)	sharp-shinned hawk
Ammodramus henslowii (SE)	Henslow's sparrow
Ammodramus savannarum	grasshopper sparrow
Bartramia longicauda (SE)	upland sandpiper
Buteo jamaicensis	red-tailed hawk
Buteo lagopus	rough-legged hawk
Carpodacus mexicanus	house finch
Carpodacus purpureus	purple finch
Chondestes grammacus	lark sparrow
Circus cyaneus (SE)	northern harrier
Cistothorus platensis (SE)	sedge wren
Colinus virginianus	northern bobwhite
Falco peregrinus (SE)	peregrine falcon
Grus canadensis (SC)	sandhill crane
Hirundo rustica	barn swallow
Icterus spurius	orchard oriole
Passerculus sandwichensis	savannah sparrow
Passerina cyanea	indigo bunting
Phasianus colchicus	ring-necked pheasant
Pooecetes gramineus	vesper sparrow
Rallus limicola (SE)	Virginia rail
Spinus tristis	goldfinch
Spiza americana	dickcissel
Spizella pusilla	field sparrow
Stelgidopteryx serripennis	northern rough-winged swallow
Sturnella magna	eastern meadowlark
Sturnella neglecta (SC)	western meadowlark

Another interesting prairie area sighting is the state special concern species Bobcat roaming the area.

CONCLUSION

The Newport Chemical Depot was awarded the U.S. Army Environmental Security Award for Natural Resources Conservation in 1996 (for 1994, 1995, and 1996) and 2003 (for 2001, 2002, and 2003). Reconstruction of the prairie was a major focus of winning these awards. Now the Newport Chemical Depot is home to the largest contiguous black soil prairie in the State of Indiana (by far), with room to potentially expand to over 2,000 total acres if the prairie reconstruction is continued on presettlement prairie soils and beyond. As stated in the Indiana Department of Natural Resources March 20, 2009 letter to the Newport Chemical Depot Reuse Authority,

> At one time 14% of the State of Indiana was covered with prairie grasses. Today less than one-tenth of 1% exists. Today, in the 21st century, prairies and the grassland animals dependent upon them are considered globally threatened. The importance of prairie to endangered grassland species, floodwater retention, groundwater recharge, watershed protection, return of carbon to soils, erosion control, and aesthetics, among many other realized benefits, has caused prairie conservation to become an increasingly important issue. A restoration of prairie on this scale would be of national significance. (Davis 2009)

However, the Newport Chemical Depot is scheduled for closure in accordance with Base Realignment and Closure Law. The Newport Chemical Depot Reuse Authority's Reuse Plan (Matrix Design Group 2009) makes no provisions for absolutely protecting the prairie. In fact, approximately 90% of the Newport Chemical Depot Tallgrass Prairie is in areas that could be plowed up or paved over, with little regard for the prairie and its wildlife. Furthermore, the Department of Defense has not indicated that they have a problem with the probable new owner (Newport Chemical Depot Reuse Authority) implementing the Newport Chemical Depot Reuse Plan's indifference concerning the conservation of the Newport Chemical Depot Tallgrass Prairie. Therefore, the Newport Chemical Depot Prairie and all of the life that it holds are in jeopardy—reminiscent of its fate in the nineteenth century.

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Compass Plant (*Silphium laciniatum*) pointing the way to the largest contiguous black soil prairie in Indiana at the U.S. Army Newport Chemical Depot, Vermillion County."

Dense Blazing Star (*Liatris spicata*) blooming in the largest contiguous black soil tallgrass prairie in Indiana at the U.S. Army Newport Chemical Depot, Vermillion County.

NO EVIDENCE OF LOCAL ADAPTATION IN SEEDLING RECRUITMENT OF NARROW-LEAVED PURPLE CONEFLOWER

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Abstract: Restoration guidelines often call for locally sourced seed, to reduce the likelihood of introducing maladapted genotypes and to avoid outcrossing depression. However, few empirical studies focus on the degree of local adaptation of native plants. To evaluate local adaptation in narrow-leaved purple coneflower (*Echinacea angustifolia*), a native perennial commonly used in prairie restorations, seeds were collected at three sites along a 500-km transect from northwestern South Dakota to west-central Minnesota. Collecting from locations at the same latitude allowed sampling of populations along a moisture gradient, while controlling for daylength and mean annual temperature. The seeds were reciprocally sown into prairie restorations located near each source population. Seedling emergence was lowest in the western South Dakota plot and highest in the Minnesota plot for all three source populations, corresponding to expectations given the moisture gradient. Moreover, seedling emergence was higher for the western South Dakota and Minnesota seeds than for the central South Dakota seeds in all three experimental plots. Survival to the end of the first growing season ranged from <50% at the central South Dakota plot to >80% in western South Dakota. Survival was highest for western South Dakota seedlings and lowest for Minnesota seedlings at all three experimental sites. There is no evidence for local adaptation at this early stage in the life cycle of *E. angustifolia*, though local adaptation may become evident at later life stages.

Key Words / Search Terms: *Echinacea angustifolia*, local adaptation, reciprocal transplant

INTRODUCTION

Human activities (e.g., logging of forests, conversion of land to agricultural use, urbanization) have resulted in widespread conversion of terrestrial ecosystems. Temperate grasslands and savannas have been identified as a critically endangered biome due to extensive habitat conversion accompanied by limited habitat protection (Hoekstra et al. 2005). Less than 5% of the original habitat remains in the northern and central tallgrass prairies of North America (Samson et al. 2004). In response to this extensive habitat conversion, there is a need to conserve extant prairie, and there is increasing interest in restoring native habitat (Anderson 2009).

Restoration guidelines often recommend using local germplasm. For example, entities using funding from Minnesota's Outdoor Heritage Fund for restoration projects are directed to "plant vegetation or sow seed only of ecotypes native to Minnesota, and preferably of the local ecotype, using a high diversity of species originating from as close to the restoration site as possible" (Laws of Minnesota 2009). The rationale for using local seed is to introduce plants that are well adapted to local environmental conditions, and to avoid genetic contamination of extant native populations. Reciprocal transplant studies have shown that plant populations are often adapted to local environmental conditions (Rehfeldt et al. 1999, Etterson 2004, Hufford et al. 2008), suggesting that establishment or persistence of individuals translocated outside their ranges may be reduced, and populations that do become established may have reduced fitness compared to populations growing in regions to which they are adapted. In a recent review of reciprocal transplant studies, Hereford (2009) found an overall frequency for local adaptation (defined as equal or higher fitness of the local population compared to foreign populations) of 0.71. Similarly, Leimu and Fischer (2008) found that plants have a home-site advantage in 71% of cases analyzed. However, in only about 45% of population pairs did performance of the local population exceed that of the foreign one at both sites in a reciprocal transplant (Leimu and Fischer 2008). Additionally, nearby populations in fragmented habitat may not contain sufficient genetic diversity for successful reintroductions or to allow for evolutionary response to changing climatic conditions (Broadhurst et al. 2008).

Here, we report on a reciprocal transplant experiment designed to test whether *Echinacea angustifolia* (narrow-leaved purple coneflower) exhibits local adaptation. *E. angustifolia* is a perennial native to the tall- and mixed-grass prairies of North America. *E. angustifolia* is representative of many native prairie species in that it is long-lived, reproduces by seed, has limited seed dispersal, is pollinated by generalist insects, and is self-incompatible (Wagenius 2000, 2004; Ault 2006). It is commonly used in restorations.

Genetic differentiation both within and among populations of *E. angustifolia* has been demonstrated using isozyme markers (Baskauf et al. 1994), random amplified polymorphic DNA (RAPD) markers (Kapteyn et al. 2002), and amplified fragment length polymorphism (AFLP) markers (Kim et al. 2004, Still et al. 2005). In the latter study, Still et al. (2005) detected increased genetic divergence with geographic distance for *E. angustifolia* populations located along a 1,500-km north-south transect from North Dakota to Oklahoma. In spite of this molecular evidence of genetic differentiation, it is not known whether adaptive differentiation has occurred, since differentiation in neutral markers is not well correlated with variation in selectively important traits (Reed and Frankham 2001, McKay and Latta 2002).

METHODS

EXPERIMENTAL DESIGN

Seeds were collected from native *E. angustifolia* populations in three locations along an east-west line. Source populations were located in the Grand River National Grassland in western South Dakota (latitude 45° 40' N, longitude 102° 10' W), and The Nature Conservancy's Samuel H. Ordway Prairie in central South Dakota (45° 43' N, 99° 5' W) and Staffanson Prairie in western Minnesota (45° 45' N, 95° 44' W). The aim was to sample along an aridity gradient, while minimizing temperature and daylength variation that would accompany a north-south transect. Average monthly temperature maxima and minima are similar for the three collection sites. Average annual precipitation is approximately 40 cm for the western South Dakota site, 50 cm for the central South Dakota site, and 62 cm for the Minnesota site, based on data from the High Plains Regional Climate Center, University of Nebraska, Lincoln (http://www.hprcc.unl.edu/).

Following the protocols suggested by the Seeds of Success program of the Chicago Botanic Garden (http://www.cbg-seedbank.org/), we collected capitula (seed heads) from 50 randomly selected individuals of each population. Mature capitula were collected near the end of the growing season (September 2008), and processed in the lab. Achenes from each individual were counted and divided into six lots. *E. angustifolia* capitula contain from zero to more than 200 achenes (personal observation); achene lots consisted of zero to more than 30 achenes. A total of 11,855 achenes were sown. One achene lot from each capitulum was sown into each of two blocks in experimental plots located near each collection site. In each block, achene lots were sown into random locations within a grid. Thus, each of the three experimental plots consisted of two blocks, each with 150 sowing positions (six rows, each with 25 positions at 0.5 m intervals). Achenes were scattered in a line directly west of each sowing position. Small brooms were used to brush away litter so that achenes could be placed directly on the soil. Experimental plots were sown in November 2008; sowing in the fall allowed for cold stratification in situ, and simulated natural conditions for seed germination (Ault 2006).

The experimental plots are located in the Grand River National Grassland (45° 40' N, 102° 10' W, managed by the U.S. Forest Service), the Perch Lake Waterfowl Production Area (45° 38' N, 99° 12' W, managed by the South Dakota Department of Environment and Natural Resources), and the Hegg Lake Wildlife Management Area (45° 47' N, 95° 40' W, managed by the Minnesota Department of Natural Resources). All three plots are on previously tilled land that has been restored to grassland. The predominant grass at Hegg Lake and Perch Lake is smooth brome (*Bromis inermis*), while crested wheat grass (*Agropyron cristatum*) dominates at the Grand River site. The experimental plots at Grand River and Hegg Lake had been burned in spring 2008; these two plots had experienced one season of regrowth prior to our fall 2008 sowing. The plot at Perch Lake had taller grass and more litter than the other two plots.

In May 2009, all seedlings that emerged were counted and their positions marked with toothpicks. In August 2009, a second census was performed, to assess seedling survival. A third search was performed in May 2010 to census the survivors and identify new germinants. In each survival census, we counted all plants present, including those with wilted or dead leaves.

The central South Dakota experimental plot was inadvertently sprayed with a combination of Tordon® and Telar® herbicides on August 4, 2009. We performed the second census on this plot on August 9-10, 2009. Although some of the *E. angustifolia* plants were beginning to wilt, the second census likely captured seedling survival up until the point of herbicide application.

DATA ANALYSIS

We analyzed the proportion of achenes sown that emerged as seedlings, using a binomial family generalized linear model (GLM), implemented in R (R Development Core Team 2009), with seed source and sowing site as predictors. The residual deviance of the full model exceeded the residual degrees of freedom, indicating overdispersion; therefore, the quasibinomial family was used for the analysis (Crawley 2007). Emergence as a proportion of achenes sown was estimated from the simplest model found to adequately fit the data. As an alternative approach, we performed a mixed effects GLM, using the R function glmer in package lme4, to account for the spatial structure in the experimental design. Block was designated as a random effect, while seed source and sow site were fixed effects. The results of the mixed effects models were qualitatively similar to the GLM models, and we here report only the results from the GLM using the quasibinomial.

Seedling survival was analyzed with a binomial family GLM, using seed source and sowing site as predictors. As in the emergence analysis, the residuals indicated overdispersion; accordingly, the quasibinomial family was used for further analysis. Estimates of the proportion of emerged seedlings that survived to late in the first growing season, and to the second season, were estimated from the simplest models that adequately fit the data. A mixed effects GLM was also performed, with qualitatively similar results to the GLM models. We here report only the results from the GLM using the quasibinomial.

RESULTS

Seedling emergence was low (less than 15%) at all three sowing sites for all three source populations (Figure 1). There was no significant seed source × sow site interaction, but the main effects of seed source and sow site were significant (Table 1). Emergence was lower for the central South Dakota seeds than for the western South Dakota or Minnesota seeds at all three experimental plots (Figure 1). Seeds from all three sources had the highest emergence at the Minnesota site and the lowest emergence at the western South Dakota site (Figure 1). A small number of new seedlings (six at the Minnesota plot and nine at the central South Dakota plot) had emerged in spring 2010. The total number of new seedlings represented only 0.12% of the total number of achenes sown.

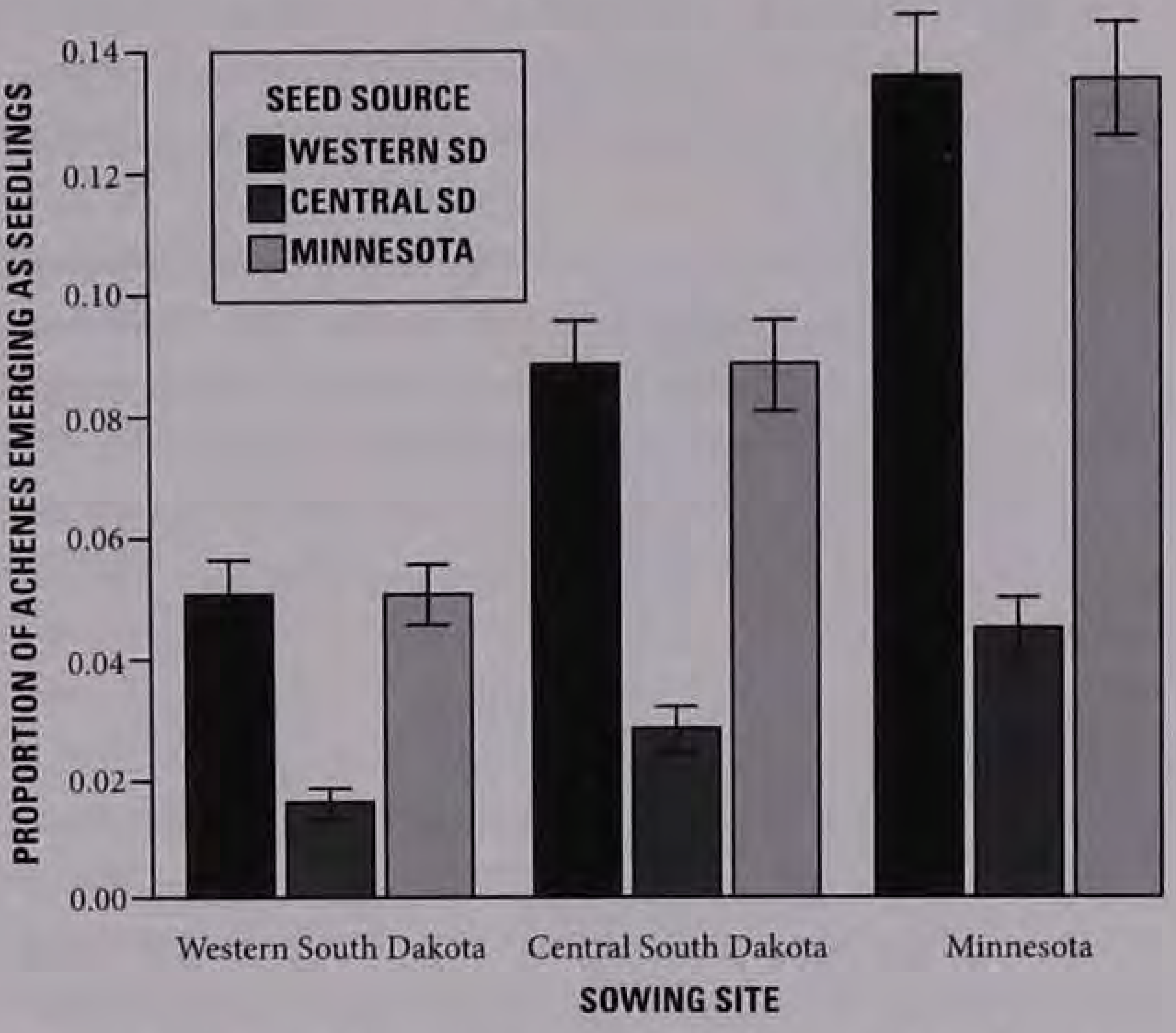

Figure 1. Mean (+/- 1 SE) proportion of Echinacea angustifolia achenes sown in fall 2008 that emerged as seedlings in spring 2009. Means were predicted from a binomial family generalized linear model that included main effects of seed source and sowing site. Seeds (achenes) were collected from the Grand River National Grassland in western South Dakota ("Western SD"), Samuel H. Ordway Prairie in central South Dakota ("Central SD"), and Staffanson Prairie in western Minnesota ("Minnesota"). Sowing sites were located in the Grand River National Grassland ("Western South Dakota"), Perch Lake Waterfowl Production Area ("Central South Dakota"), and Hegg Lake Wildlife Management Area ("Minnesota").

Table 1. Likelihood ratio tests for stepwise model simplification using backward elimination for the seedling emergence analysis. Deviance is the likelihood ratio test statistic. P values are for the F-test of the null hypothesis that a model simplified by removing the listed interaction or factor is not significantly different from the model including the listed term. The full model included main effects of seed source and sow site as well as the seed source × sow site interaction. After model simplification, the minimal adequate model included main effects of seed source and sow site.

INTERACTION OR FACTOR	DF	DEVIANCE	P
seed source × sow site	4	4.93	0.5388
seed source	2	9.99	0.04391
sow site	2	132.98	<0.0001

Survival over the first growing season (through August 2009) ranged from approximately 50% at the central South Dakota plot to approximately 80% at the western South Dakota site. There was no significant seed source × sow site interaction, but main effects of seed source and sow site were significant (Table 2). The significant source effect reflects a higher survival rate for western South Dakota-sourced seedlings and a lower survival for the Minnesota-sourced seedlings at all three sowing sites (Figure 2). The significant sow-site effect reflected a much lower survival in the central South Dakota site compared to the western South Dakota and Minnesota sites.

Table 2. Likelihood ratio tests for stepwise model simplification using backward elimination for the analysis of seedling survival to the end of the first growing season. Deviance is the likelihood ratio test statistic. P values are for the F-test of the null hypothesis that a model simplified by removing the listed interaction or factor is not significantly different from the model including the listed term. The full model included main effects of seed source and sow site as well as the seed source × sow site interaction. After model simplification, the minimal adequate model included main effects of seed source and sow site.

INTERACTION OR FACTOR	DF	DEVIANCE	P
seed source × sow site	4	5.88	0.7263
seed source	2	314.4	<0.0001
sow site	2	256.84	<0.0001

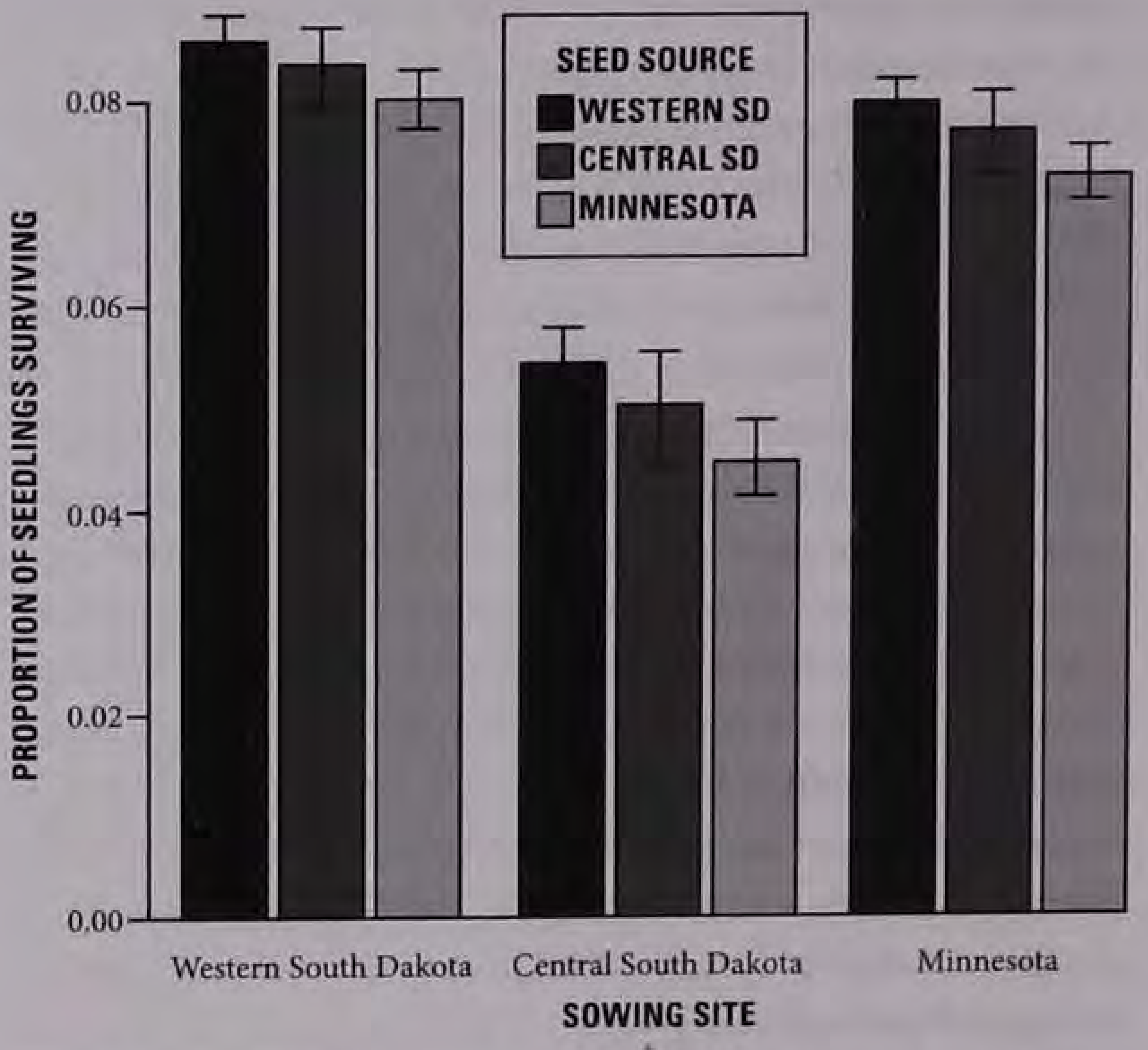

Figure 2. Mean (+/- 1 SE) proportion of seedlings that emerged in 2009 surviving until late summer 2009. Means were predicted from a generalized linear model that included main effects of seed source and sow site. Seeds were collected from the Grand River National Grassland in northwestern South Dakota ("Western SD"), Samuel H. Ordway Prairie in central South Dakota ("Central SD"), and Staffanson Prairie in western Minnesota ("Minnesota"). Sow sites were located in the Grand River National Grassland ("Western South Dakota"), Perch Lake Waterfowl Production Area ("Central South Dakota"), and Hegg Lake Wildlife Management Area ("Minnesota").

For the analysis of survival to the second growing season (through May 2010), there was no significant seed source × sow site interaction. The main effect of seed source was also ns, but the main effect of sow site was highly significant ($P < 0.0001$), reflecting the fact that there were no surviving seedlings at the central South Dakota experimental plot, while survival at the Minnesota and western South Dakota plots was relatively high. When the central South Dakota plot was excluded from the analysis, the main effects of seed source and sow site were both ns, as was the two-way interaction term. Excluding the central South Dakota plot, the mean proportion of 2009 seedlings that survived to May 2010 was 0.75.

DISCUSSION

Our results do not indicate that the *Echinacea angustifolia* populations we sampled are locally adapted, as measured by seedling emergence or early seedling survival. Evidence of local adaptation may emerge as longer-term survival and fecundity are considered. The magnitude of local adaptation tends to be smaller when fitness is measured as viability than when fitness is measured as fecundity or a composite of viability and fecundity (Hereford 2009).

The significant effect of seed source observed in both emergence and survival could reflect genetic differences among the populations sampled. Alternatively, maternal environmental effects may have influenced the emergence or early survival of the *Echinacea angustifolia* seedlings (Roach and Wulff 1987). In particular, the low seedling emergence for the central South Dakota population relative to the other two populations may be at least partially attributable to collection of immature seeds of the central South Dakota population. We noted during collections that at least three of the central South Dakota seed heads were collected from green stems. The *E. angustifolia* seed heads matured earlier in western South Dakota than in central South Dakota or Minnesota in 2008 (personal observation). Collections were made on September 6 at the western South Dakota site, September 7 at the central South Dakota site, and September 20 at the Minnesota site. Since maternal effects often decrease at later life stages (Roach and Wulff 1987, Zeiter and Stampfli 2008), a signal of local adaptation may become evident when long-term fitness is assessed.

For all three source populations, seedling emergence was lowest in western South Dakota and highest in Minnesota (Figure 1). The increase in emergence from west to east did not seem to be directly attributable to the precipitation gradient; total precipitation from the time of seed sowing until the first seedling census (December 2008-May 2009) was approximately equal at the western South Dakota (20 cm) and central South Dakota (21 cm) sites, and lower at the Minnesota site (12 cm). Seeds were exposed to a suite of biotic and abiotic environmental conditions at each site (e.g., diversity and abundance of herbivores and competitors, soil characteristics, snow cover, relative humidity, etc.). The sowing sites were selected as representative of potential restoration sites as close to the seed source populations as possible, with the aim of testing for adaptation to the entire suite of local conditions.

Survival over the first growing season was lower in the central South Dakota site than in western South Dakota or Minnesota (Figure 2). This reduced survival in central South Dakota may have been due to higher competition for light or other resources at that site relative to the western South Dakota and Minnesota sites, which had been burned in the spring prior to the fall sowing, reducing grass cover compared to the central South Dakota site (personal observation). Our results agree well with a ten-year recruitment experiment (Wagenius et al. *2011*) showing that survival of *E. angustifolia* seedlings was enhanced by burn treatments.

The western South Dakota population had the highest survival and the Minnesota population had the lowest survival at all three sites over the first growing season (Figure 2). However, this trend was not observed in survival to the second growing season, as all three populations had an equally high proportion surviving (mean 0.75).

The initial analysis of survival to the second growing season indicated a significant main effect of sowing site, driven entirely by the mortality of all *E. angustifolia* seedlings at the central South Dakota site, which was sprayed with herbicide in August 2009. When the central South Dakota site was removed from the analysis, the main effect of seed source was no longer significant, indicating that there were no significant differences in survival among the three source populations or between the two remaining sites.

Although this study did not provide evidence of local adaptation in *E. angustifolia*, we do not suggest that seed provenance is an unimportant consideration for restoration. As mentioned above, nonlocal populations may have reduced fecundity or long-term viability compared to local plants. Additionally, crosses between local and nonlocal genotypes may result in outcrossing depression (Hufford and Mazer 2003, Crémieux et al. 2010). Wagenius et al. (2010) performed between-population and within-population crosses for remnant *E. angustifolia* populations in Douglas County, MN. They report a nonsignificant decrease in fitness for the between-population crosses compared to the within-population crosses. Between-population crosses for more distant populations may exhibit more severe outcrossing depression.

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PRELIMINARY FINDINGS OF INTERSEEDING PRAIRIE FORBS INTO NATIVE GRASS STANDS

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***Abstract*:** Interseeding native forbs into reconstructed grasslands could restore plant species diversity and improve wildlife habitat. However, many managers report having limited experience with interseeding and poor success with a few early attempts. Survival of forbs interseeded directly into existing vegetation may be enhanced by management treatments that reduce competition from established grasses. In 2009 a study was initiated to investigate the effects of two mowing and two herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota. In January of 2009 a pilot site was interseeded and two mowing treatments (once or twice per season) and two grass-selective herbicide treatments (high- and low-rate) were applied during the 2009 growing season. Vegetation from the pilot site was monitored during the summer of 2010. One month after treatments were applied, mean visual obstruction readings were less in treated plots than in control plots, indicating the prescribed treatments were effective in suppressing growth of grass. Black-eyed Susan (*Rudbeckia hirta*) was the most common seeded species in all treatments and was more frequent in the herbicide-low, mow, and herbicide-high treatments, respectively, than in the control. Results of the pilot study will be used to guide a larger study on 16 sites in southern Minnesota.

Keywords / Search Terms: prairie management, interseeding, prairie reconstruction, wildflower establishment, seedling establishment, species diversity, mowing, grass selective herbicide

INTRODUCTION

In a survey on grassland information needs (Tranel 2007), 82% of Minnesota Department of Natural Resources (MNDNR) wildlife managers indicated a need for information on maintaining plant species diversity in restored grasslands. In particular, managers wanted more information on establishing and maintaining an abundance and diversity of forbs in grasslands. A diversity of forbs in grasslands provides the heterogeneous vegetation structure needed by some bird species for nesting and brood rearing (Volkert 1992, Sample and Mossman 1997). Forbs also provide habitat for invertebrates, an essential food for grassland birds and their broods (Buchanan et al. 2006). Insect abundance in chick diets has been positively correlated with growth rates and survival in gallinaceous birds, such as grouse (*Lagopus lagopus scoticus*, Park et al. 2001; Huwer et al. 2008), gray partridge (*Perdix perdix*, Sotherton and Robertson 1990), and pheasants (*Phasianus colchicus*, Hill 1985). Broods of gallinaceous birds, such as prairie chickens (*Tympanuchus cupido*), move directly from nests to brood habitat (Svedarsky 1979), and habitats with high forb abundance were preferred (Jones 1963, Drobney and Sparrow 1977).

The forb component on many restored grasslands has been lost or greatly reduced. Managers interested in increasing the diversity and quality of forb-deficient grasslands are faced with the costly option of completely eliminating the existing vegetation and planting into bare ground, or attempting to interseed forbs directly into existing vegetation. Management techniques that reduce competition from established grasses may provide an opportunity for forbs to become established in existing grasslands (Collins et al. 1998). Temporarily suppressing dominant grasses may increase light, moisture, and nutrient availability to seedling forbs, ultimately increasing forb abundance and diversity (Schmitt-McCain 2008). Williams et al. (2007) found that frequent mowing of grasslands in the first growing season after interseeding increased forb emergence and reduced forb mortality. Snyder et al. (unpublished data) reported that patch tilling and interseeding with forbs can increase species diversity in grass-dominated stands. Additionally, Hitchmough and Paraskevopoulou (2008) found that forb density, biomass, and richness were greater in meadows where a grass herbicide was used.

In this study, the effects of two mowing and two herbicide treatments on diversity and abundance of forbs interseeded into established grasslands in southern Minnesota were investigated.

METHODS

One pilot site was selected in 2008 and 17 sites were selected in 2009 for the full study. Study sites were distributed throughout the southern portion of Minnesota's prairie/farmland region (Figure 1), including 17 sites on state-owned Wildlife Management Areas (WMA) and one site on a federally owned Waterfowl Production Area (WPA).

Each site was ≥4 ha and characterized by relatively uniform soils, hydrology, and vegetative composition. All sites were dominated by relatively uniform stands of native grasses with few forbs, most of which were nonnative. Pre-study observations revealed that the pilot site also contained some forbs prior to interseeding, most of which were nonnative, such as sweet clover (*Melitotus alba, M. officinalis*).

The pilot site was interseeded during January 2009 following a fall 2008 burn. An unusually wet October did not allow for burning at nine of the 17 sites for the full study. As a result, eight sites were burned in October-November 2009 and frost interseeded during December 2009 and March 2010, eight sites were burned and interseeded during April and May 2010, and one site in Faribault county that was not burned was dropped from the study (Figure 1). The same 30-species mix of seed (Table 1) was broadcast-seeded at all sites using a tractor- or ATV-mounted Vicon spreader at a rate of 239 pure live seeds/square meter. The seed mix was designed to include species from a variety of functional guilds including perennial legume, biennial forb, perennial forb, annual legume, and perennial sedge. Seed used on spring-burned sites was cold-moist stratified for 3-5 weeks in wet sand to stimulate germination during spring 2010. Seed used on fall-burned sites was not cold-moist stratified.

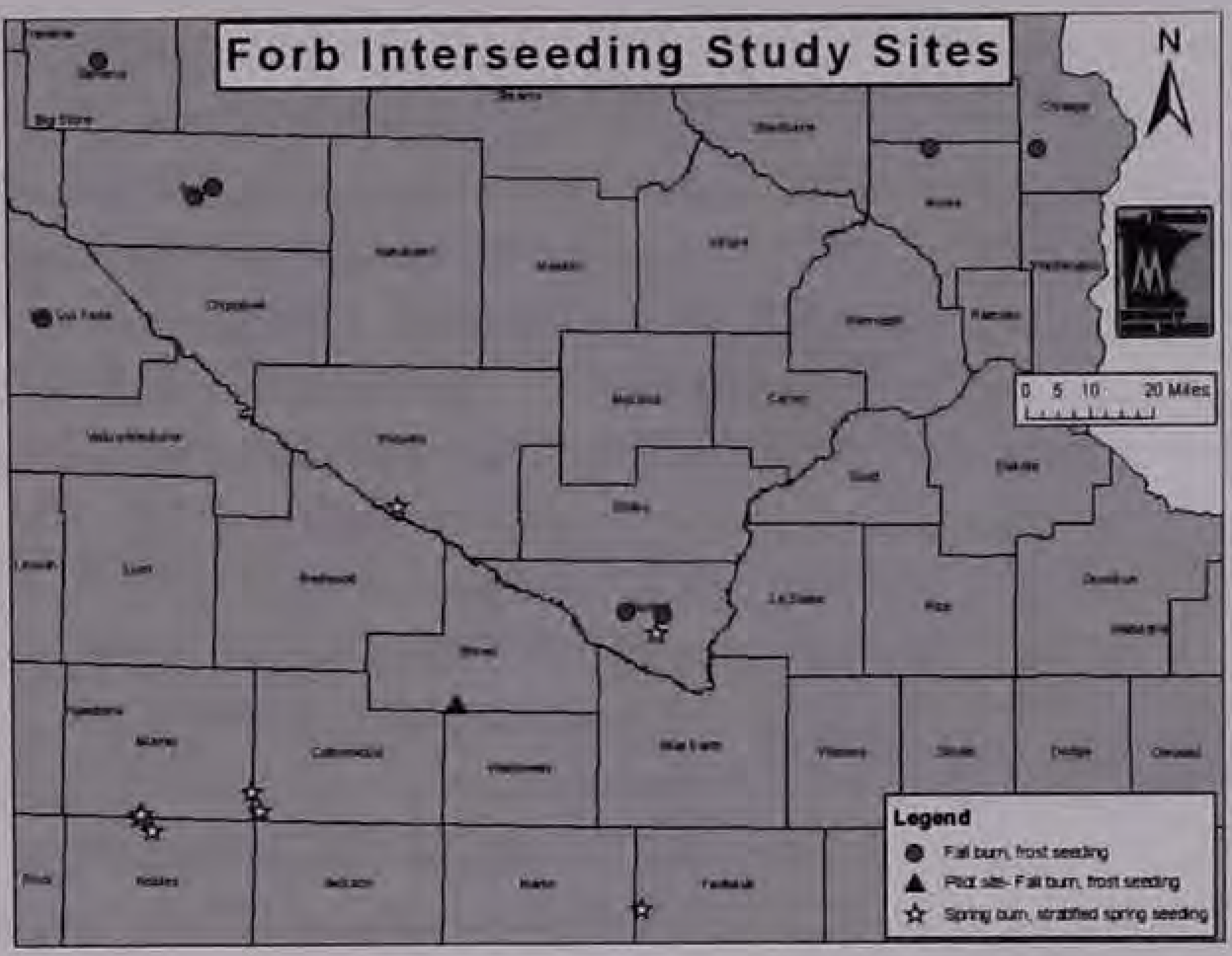

Figure 1. Locations of study sites for forb interseeding study, categorized by season of interseeding, southern Minnesota, 2009-2010.

TREATMENTS

After each site was prepared and seeded, it was divided into 10 plots of approximately equal size , each of which was randomly assigned one of the four treatments or the control. Each site received all treatments to account for variability among sites, and each treatment was replicated twice at each site. The following treatments, which were designed to suppress grass competition, were applied during the first growing season after interseeding (2009 for the pilot study, 2010 for the full study) while the forbs were becoming established:

1. Mowed once to a height of 10-15 cm when vegetation reached 25-35 cm in height
2. Mowed twice to a height of 10-15 cm when vegetation reached 25-35 cm in height
3. Applied grass herbicide Clethodim (Select Max®) at 108 mL/ha (9 oz/A) when vegetation reached 10-15 cm
4. Applied grass herbicide Clethodim (Select Max®) at 215 mL/ha (18 oz/A) when vegetation reached 10-15 cm

SAMPLING METHODS

Prior to burning and interseeding, all sites, except the pilot, were surveyed by a botanist in summer 2009 to determine species already present and general condition of each site. This also allowed for field testing of the vegetation survey protocol. Four transects 50 meters in length were randomly located within each study plot and recorded using a Global Positioning System unit. Percent cover of live vegetation (Daubenmire 1959) was estimated in 76 x 31cm² quadrats spaced every 5 meters and litter depth was estimated every 10 meters. Visual obstruction readings (VOR; Robel et al. 1970) were recorded in the four cardinal directions at the beginning and the end of each transect. Species richness was estimated by counting the number of sampling quadrats for which each species was present (frequency).

Table 1. Seed mix consisting of 29 forb and 1 sedge species interseeded into established grass stands at all study sites. Mix was interseeded at a rate of approximately 239.39 pure live seeds per square meter.

SCIENTIFIC NAME	COMMON NAME	SEEDS/ METER²	SEEDS/ FOOT²
Allium stellatum	Prairie Onion	1.90	0.18
Amorpha canescens	Leadplant	1.98	0.18
Asclepias syriaca	Common Milkweed	0.99	0.09
Aster oolentangiensis	Sky Blue Aster	16.80	1.56
Aster ericoides	Heath Aster	7.41	0.69
Aster novae-angliae	Nw England Aster	10.60	0.98
Astragalus canadensis	Canada Milk Vetch	7.35	0.68
Carex vulpinoidea	Brown Fox sedge	6.18	0.57
Chamaechrista fasciculata	Partridge Pea	1.33	0.12
Coreopsis palmata	Prairie Coreopsis	0.62	0.06
Dalea candida	White Prairie Clover	7.04	0.65
Dalea purpurea	Purple Prairie Clover	5.56	0.52
Desmodium canadense	Showy Tick Trefoil	1.16	0.11
Echinacea angustifolia	Narrow Leafed Purple Coneflower	1.47	0.14
Gentiana andrewsii	Closed Bottle Gentain	5.54	0.51
Helianthus maximilianii	Maximilian Sunflower	1.61	0.15
Heliopsis helianthoides	False Sunflower	1.95	0.18
Heuchera richardsonii	Alumroot	8.65	0.80
Liatris aspera	Rough Blazingstar	0.59	0.06
Liatris pycnostachya	Prairie Blazingstar	0.54	0.05
Monarda fistulosa	Wild Bergamot	12.11	1.12
Potentilla arguta	Prairie Cinquefoil	48.31	4.49
Ratibida pinnata	Yellow Coneflower	10.01	0.93
Rudbeckia hirta	Black-eyed Susan	27.28	2.53
Solidago rigidum	Stiff Goldenrod	7.60	0.71
Verbena hastata	Blue Vervain	17.24	1.60
Verbena stricta	Hoary Vervain	4.50	0.42
Veronicastrum virginicum	Culver's Root	19.77	1.84
Zizia aptera	Heart Leaf Golden Alexander	0.59	0.06
Zizia aurea	Golden Alexanders	2.72	0.25
TOTAL INTERSEEDING RATE		**239.39**	**22.24**

RESULTS AND DISCUSSION

PILOT SITE

Because of staffing limitations, the second mowing treatment was not applied to the pilot site, and herbicide treatments were applied when the grass was taller (31 cm) than prescribed (10-15 cm). One month after treatments were applied, mean VORs were less in treated plots than in control plots (Figure 2), indicating that the prescribed treatments were effective in suppressing growth of grass.

Big bluestem (*Andropogon gerardi*) was the most frequently occurring species in the control (90%) but appeared less frequently in the mow-once treatment (53.3%), herbicide-low treatment (72.5%), and the herbicide-high treatment (70.0%) (Figure 3). Quackgrass (*Agropyron repens*) was the most frequently occurring species in the mow-once treatment (56.7%) and was abundant in the control (50.0%), herbicide-low treatment (35.0%), and the herbicide-high treatment (45.0%) (Figure 3).

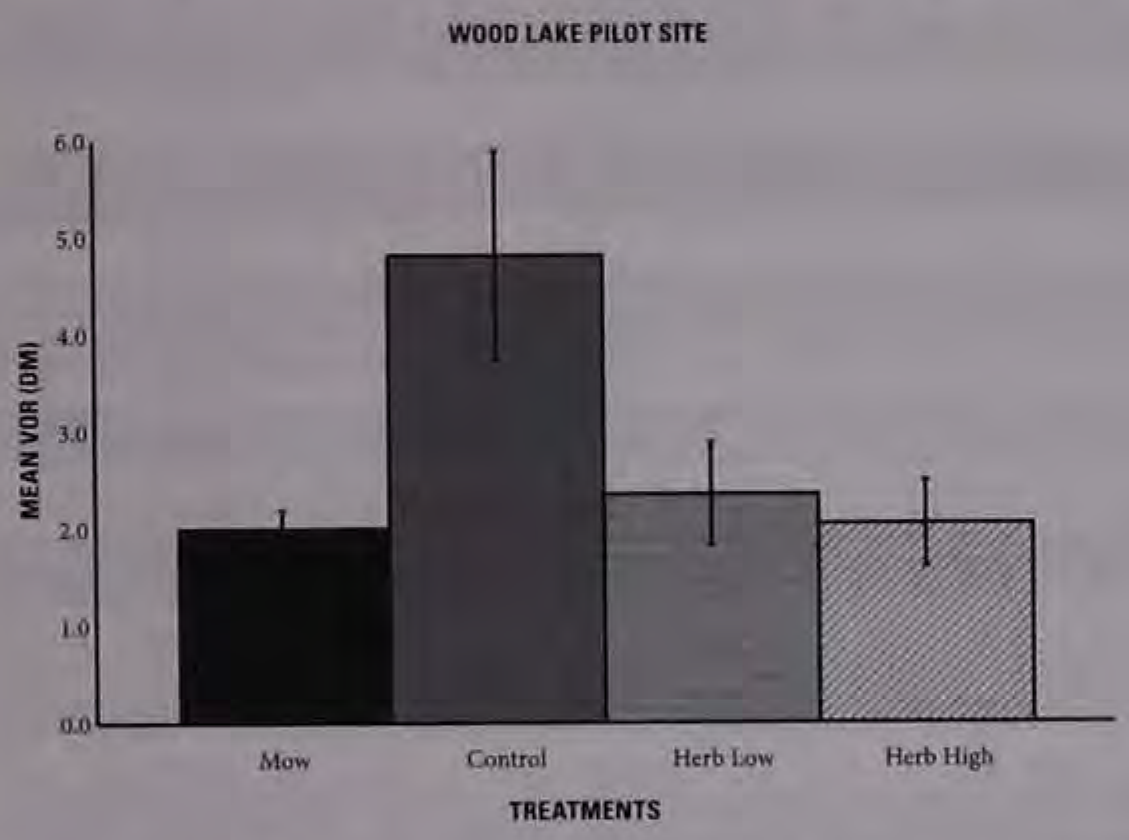

Figure 2. Mean Visual Obstruction Readings by treatment type (mow = mow once; herb low = herbicide applied at low rate; herb high = herbicide applied at high rate) observed during treatment year at the pilot study site, Wood Lake, Minnesota, 2009. Error bars represent 95% confidence intervals.

Dandelion (*Taraxacum officinale*) was more frequent in the mow-once treatment (46.7%) and herbicide-high treatment (60.0%) than in the control (30.0%) (Figure 4). Creeping woodsorrel (*Oxalis corniculata*) was more frequent in the herbicide-low treatment (57.5%), mow once treatment (46.7%), and herbicide-high treatment (45.0%) than in the control (36.7%) (Figure 4). Sweet clover was common regardless of treatment (Table 2 and Figure 4).

One year following treatments, seeded native forbs were present in all treatments (Table 2). Black-eyed Susan (*Rudbeckia hirta*) was the most common seeded forb species in all treatments, and was least common in the control (13.3%) and most common in the herbicide low treatment (50%; Table 2). The mow once treatment had the greatest diversity of seeded native species and the herbicide high treatment had the lowest (Table 2).

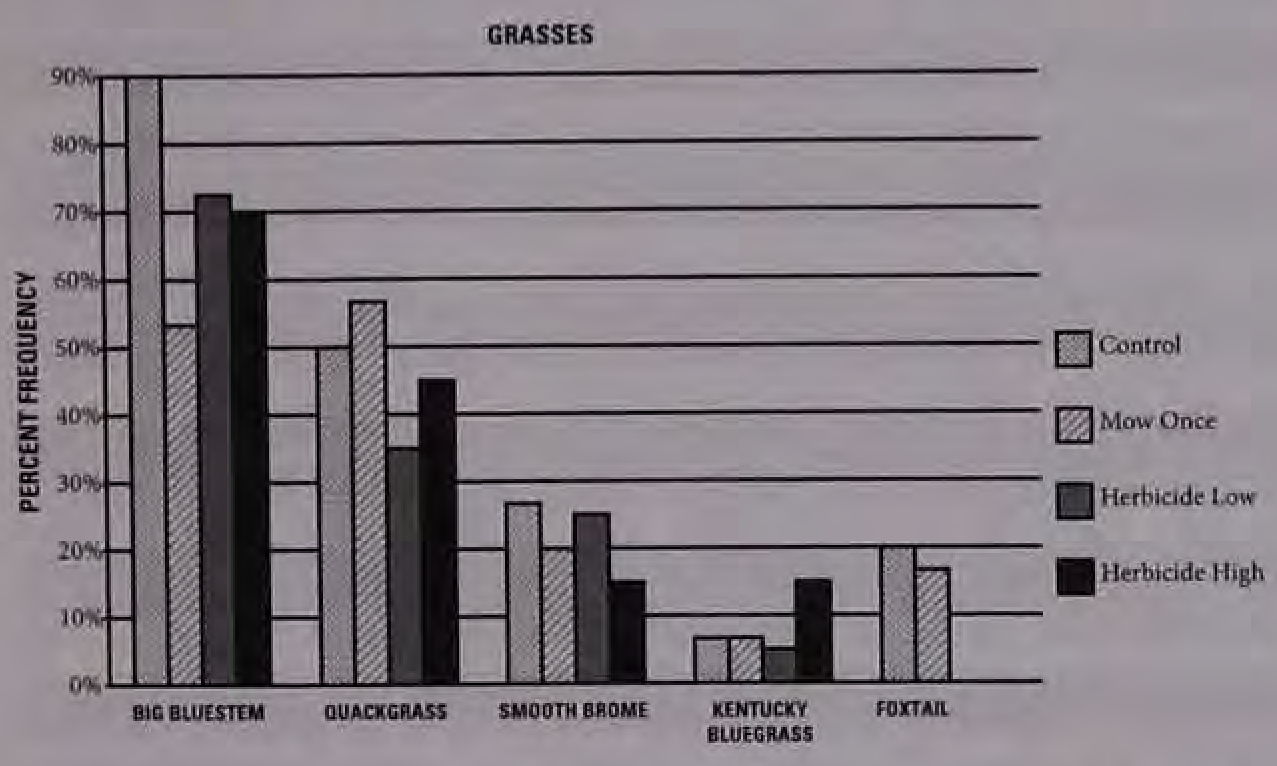

Figure 3. Percent frequency of selected grass species observed during treatment year at the pilot study site, Wood Lake, MN, 2009 (n=30 for mow once and control, n=40 for herbicide low, and n=20 for herbicide high).

Our mowing and herbicide treatments were effective at suppressing growth of grasses, as demonstrated by reduced mean VORs (Figure 2) and reduced frequency of occurrence of both native and exotic grasses in the first year of the pilot study (Figure 3). The increased frequency of weedy, disturbance-loving species in the herbicide treatments (Figure 4) suggests these treatments were successful at suppressing dominant grasses. Hitchmough and Paraskevopoulou (2008) found that in treatments where grass was suppressed with a graminoid herbicide, sown forb density was higher in the second and third years after treatment and forb richness greater three years after treatment. Additionally, Willliams et al. (2007) found that in the fourth year sown forbs were twice as abundant in treatments where grass was suppressed by mowing than in untreated controls. A complete vegetation survey will be conducted on all sites in summers 2011-2013 to determine the extent of forb survival, species diversity, and weed persistence in my study.

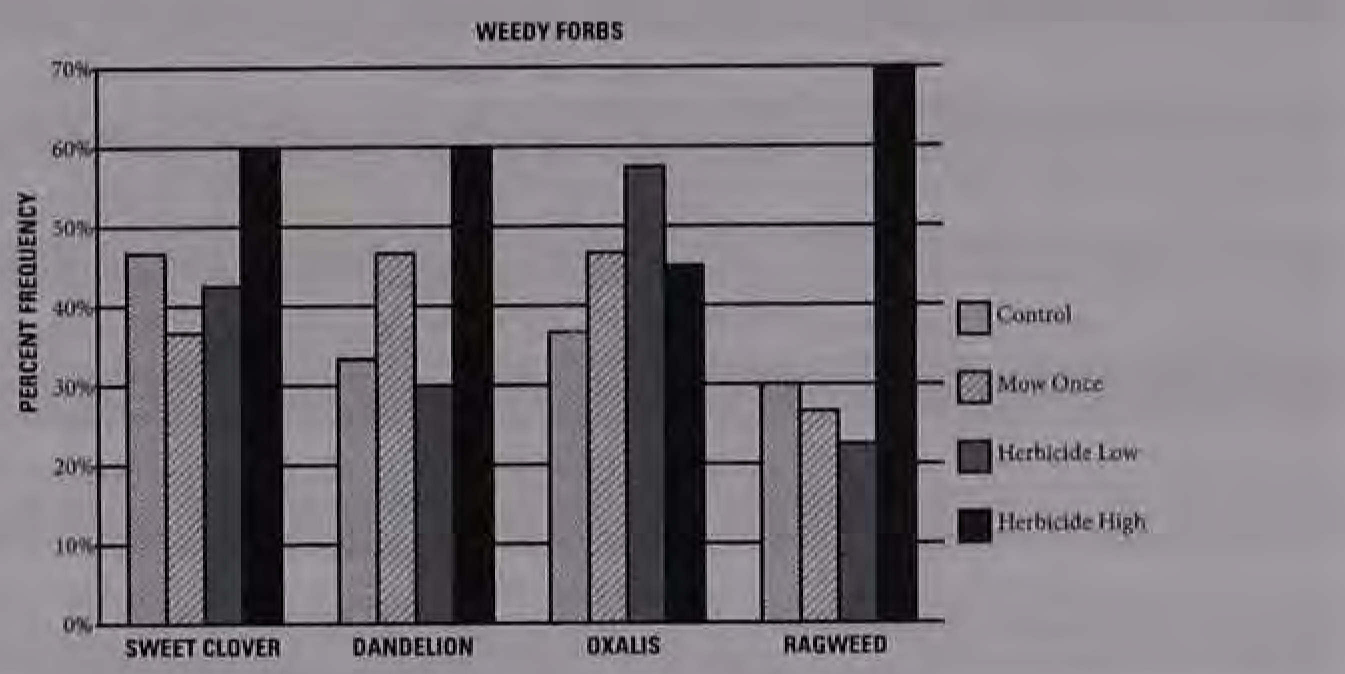

Figure 4. Percent frequency of selected weedy forb species observed during treatment year at the pilot study site, Wood Lake, MN, 2009 (n=30 for mow once and control, n=40 for herbicide low, and n=20 for herbicide high).

FULL STUDY SITES

Prior to the study, vegetation sampling at each of the study sites revealed that invasive grasses and forbs were present at most sites (Table 3). Percent cover of native grass at the sites ranged from 12.8% to 59.8% while exotic grass ranged from 0.3% to 36.9%. Percent cover of native forbs varied from zero to 19.4% and exotic forbs ranged from 1.6% to 28.8% cover. Total percent cover could be greater than 100% because of canopy overlap and bare ground or litter underneath a canopy.

After the interseeding and study treatments, forbs that were not present prior to the study were observed at all study sites, indicating that germination of sown forbs has occurred. Weedy or opportunistic species such as sweet clover and Canada thistle (*Cirsium arvense*) appeared to exploit the distur-

Table 2. Frequency of selected forb species by treatment type at Wood Lake, MN pilot site one year following treatments. Sampling size (n) is number of quadrats placed in groups of ten along transects within each treatment type.

SCIENTIFIC NAME	COMMON NAME	PLANT STATUS	PRESENCE	n	FREQUENCY
MOW ONCE TREATMENT					
Rudbeckia hirta	Black-eyed Susan	Native, seeded	8	30	26.7%
Solidago canadensis	Canada goldenrod	Native, volunteer	3	30	10.0%
Monarda fistulosa	Wild bergamot	Native, seeded	3	30	10.0%
Aster spp.	Unidentified asters	Native, seeded	1	30	3.3%
Zizia aurea	Golden Alexanders	Native, seeded	1	30	3.3%
Dalea candida	White Prairie Clover	Native, seeded	1	30	3.3%
Melitotus alba & M. officinalis	White & Yellow sweet clover	Non-native	11	30	36.7%
CONTROL - NO TREATMENT					
Rudbeckia hirta	Black-eyed Susan	Native, seeded	4	30	13.3%
Zizia aurea	Golden Alexanders	Native, seeded	1	30	3.3%
Monarda fistulosa	Wild bergamot	Native, seeded	1	30	3.3%
Amorpha canescens	Leadplant	Native, seeded	1	30	3.3%
Melitotus alba & M. officinalis	White & Yellow sweet clover	Non-native	14	30	46.7%
HERBICIDE LOW TREATMENT					
Rudbeckia hirta	Black-eyed Susan	Native, seeded	20	40	50.0%
Ratibida pinnata	Yellow coneflower	Native, seeded	3	40	7.5%
Solidago canadensis	Canada goldenrod	Native, seeded	2	40	5.0%
Zizia aurea	Golden Alexanders	Native, seeded	1	40	2.5%
Monarda fistulosa	Wild bergamot	Native, seeded	1	40	2.5%
Melitotus alba & M. officinalis	White & Yellow sweet clover	Non-native	17	40	42.5%
HERBICIDE HIGH TREATMENT					
Rudbeckia hirta	Black-eyed Susan	Native, seeded	4	20	20.0%
Solidago canadensis	Canada goldenrod	Native, volunteer	3	20	15.0%
Asclepias syriaca	Common milkweed	Native, seeded	1	20	5.0%
Melitotus alba & M. officinalis	White & Yellow sweet clover	Non-native	12	20	60.0%

bances associated with interseeding. Efforts aimed at weed control prior to interseeding could potentially maximize the ability of native forbs to compete and to reduce the need to treat noxious weeds after interseeding, when native forb seedlings are vulnerable to harm.

MANAGEMENT IMPLICATIONS

The use of the pre-emergent grass-selective herbicide Clethodim (Select Max) at 108 mL/ha (9 oz/A) and 215 mL/ha (18 oz/A) was effective at suppressing well-established native and exotic grasses at the pilot site. Growth of grass

Table 3. Percent cover types for each study site prior to the study. Total percent cover can equal greater than 100 percent because of canopy overlap and bare ground or litter underneath a canopy.

STUDY SITE	NATIVE GRASS	NATIVE FORB	EXOTIC GRASS	EXOTIC FORB	BARE GROUND	DEAD LITTER
Duck Lake	56.7	0.0	9.1	2.3	0.0	85.0
Middle Lake	59.8	0.4	6.1	5.9	4.9	61.1
Peterson Lake	59.1	1.8	7.1	14.2	0.3	85.0
Lake Guckeen	45.9	1.8	1.8	5.0	2.0	82.2
Beaver Falls	12.8	18.6	27.3	28.8	4.7	85.0
Cleanwater	53.0	1.9	27.4	1.8	3.3	82.1
Fenmont	50.4	11.2	15.8	8.6	3.3	85.0
Swessinger	36.2	6.4	36.9	10.5	11.4	47.1
Talcot Lucky	41.5	2.1	31.8	5.3	5.3	77.6
Talcot North	42.3	9.4	32.2	4.1	4.4	76.4
25th Anniversary	52.0	9.1	2.8	15.2	0.7	82.8
Schultz WPA	58.6	0.4	3.0	1.6	13.1	61.3
Claire Rollings E.	47.0	1.3	8.6	1.6	8.4	68.3
Claire Rollings W.	28.6	19.4	17.8	19.8	4.8	77.1
Bethel	37.6	4.5	0.3	9.9	11.8	58.1
Sunrise	16.0	3.4	0.6	1.3	8.4	68.3
AVERAGE	**43.8%**	**6.5%**	**13.8%**	**9.1%**	**5.9%**	**71.7%**
RANGE	**12.8-59.8%**	**0-19.4%**	**0.3-36.9%**	**1.6-28.8%**	**0-15.2%**	**28.8-85%**

was stunted but grass mortality was not observed even at the high application rate at any of the study sites. Because this herbicide is fairly inexpensive and requires only one application in a growing season, it could prove to be a cost-effective alternative to repeated mowing in areas where grass suppression is desired. The use of grass-selective herbicides to achieve other management objectives such as the control of reed canarygrass (*Phalaris arundinacea*) through herbicide application followed by interseeding native sedges (*Cyperaceae*) warrants further investigation.

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THE ROLE OF FORB SEEDING RATE IN ENHANCING FLORISTIC DIVERSITY IN PRAIRIE RECONSTRUCTIONS

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Abstract: Although creating floristic and habitat diversity are central goals of prairie reconstruction, numerous studies show that most reconstructions fail to match the diversity of remnants. Equipping restorationists with a better understanding of basic planting variables, such as seed mix composition, seeding rates, and planting times, would enhance the ability to create more authentic and diverse plantings, particularly given the widespread impact of certain planting practices, such as those promoted through the Conservation Reserve Program (CRP). This study examined the effects of seeding rate, grass to forb ratio, and planting time on floristic diversity in 144 reconstruction plots planted in full factorial design. Despite planting of equal numbers of species in all plots, diversity increased with absolute forb seeding rate and with increased proportions of forb seeds relative to grass seeds. Diversity positively correlated with total and forb seeding rates but not with grass seeding rate, indicating that forb seeding rate is a key driver of diversity, explaining 32% of the variation in diversity values. Native plant density and richness showed similar trends, which also linked increased forb rates to suppression of weed growth. In this study, diversity, native plant density, and weed suppression were all enhanced by planting at a rate of at least 430 seeds/m^2 with forbs comprising more than 50% of the seeds (by number). These results contradict the common practice of planting high rates of native grass seed with minor portions of forb seed when high native diversity, density, and weed suppression are goals.

Key Words / Search Terms: prairie restoration, grassland restoration, Midwest, tallgrass prairie, weed suppression, planting time.

INTRODUCTION

Numerous studies have shown that prairie reconstructions seldom succeed in closely reproducing the plant communities of remnant prairies, especially with regard to floristic diversity and conservative or climax species (Howe 1994, Kindscher and Tieszen 1998, Weber 1999, Allison 2002, Sluis 2002, Kleijn 2003, Averett et al. 2004, Baer et al. 2005, Martin et al. 2005, McLachlan and Knispel 2005, Polley et al. 2005). A variety of causes have been suggested for the lack of diversity in reconstructions including excessive planting of pioneering, aggressive, or dominant species, particularly dominant grasses (Weber 1999, Polley et al. 2005) early successional establishment problems for conservative species (Howell and Kline 1994, Kleijn 2003); lack of soil/nutrient heterogeneity (Baer et al. 2005, Lane and BassiriRad 2005); excess available soil nitrogen (Averett et al. 2004); improper management of disturbances such as fire and grazing (Howe 1994, 1999; Helzer and Steuter 2005; Martin and Wilsey 2006); and underdevelopment of local processes such as resource partitioning and dispersal (Polley et al. 2005).

Many of these hypothesized causes are directly linked to basic planting methodologies, and much debate remains about questions such as what to plant (seed mixes), how much to plant (seeding rates), when to plant, and how to plant it. Development and adoption of improved planting practices is, thus, an important step in advancing prairie reconstruction, especially given the widespread impact of certain planting practices, such as those promoted through the Conservation Reserve Program (CRP).

Existing studies of planting methodology tend to focus on native grass plantings (Dale and Smith 1986, Huddle et al. 2001, Bakker et al. 2003, Wilson et al. 2004). Many studies of more diverse plantings report diversity levels, but only a few systematically evaluate the contribution of particular reconstruction practices to diversity. These include Martin et al. (2004) for the basic planting variables of seeding rate, percentage of forb seed (grass to forb ratio), and planting time; Piper and Pimm (2002) and Piper et al. (2007) for seed-mix richness and multiphase plantings, Baer et al. (2005) for experimentally induced soil heterogeneity; and Averett et al. (2004) for soil carbon amendment.

Dickson and Busby (2009) report what they consider the first replicated restoration experiment examining the impact of varied grass-seeding density on forbs, finding that dominant grasses do compete strongly with forbs and likely limit the resulting diversity of reconstructions. This study expands on their work while including other methodological variables.

This study varies seeding rate, relative percentage of forb seed (grass to forb ratio), and planting time, but not seed-mix richness, to examine their contributions to floristic diversity. We hypothesized that given appropriate site preparation, highly diverse plantings could be achieved with low seeding rates provided that, a species-rich seed mix is used. We expected even low-seeding-rate treatments to develop high diversity since, although initial establishment might be sparse, the same rich complement of species was available to generate diversity (a density-independent measure) and could fill in sparse stands over time. If true, low seeding rates of diverse mixes could save limited resources in landscape-scale restorations. We also hypothesized that the high planting densities of warm-season grass seed that are commonly used in many reconstructions are unnecessary or even detrimental to the floristic diversity of the plantings. We expected higher-forb-percentage treatments to result in higher diversity due to reduced competition from aggressive grasses and increased availability of forb propagules.

MATERIALS AND METHODS

STUDY SITE AND SEEDS

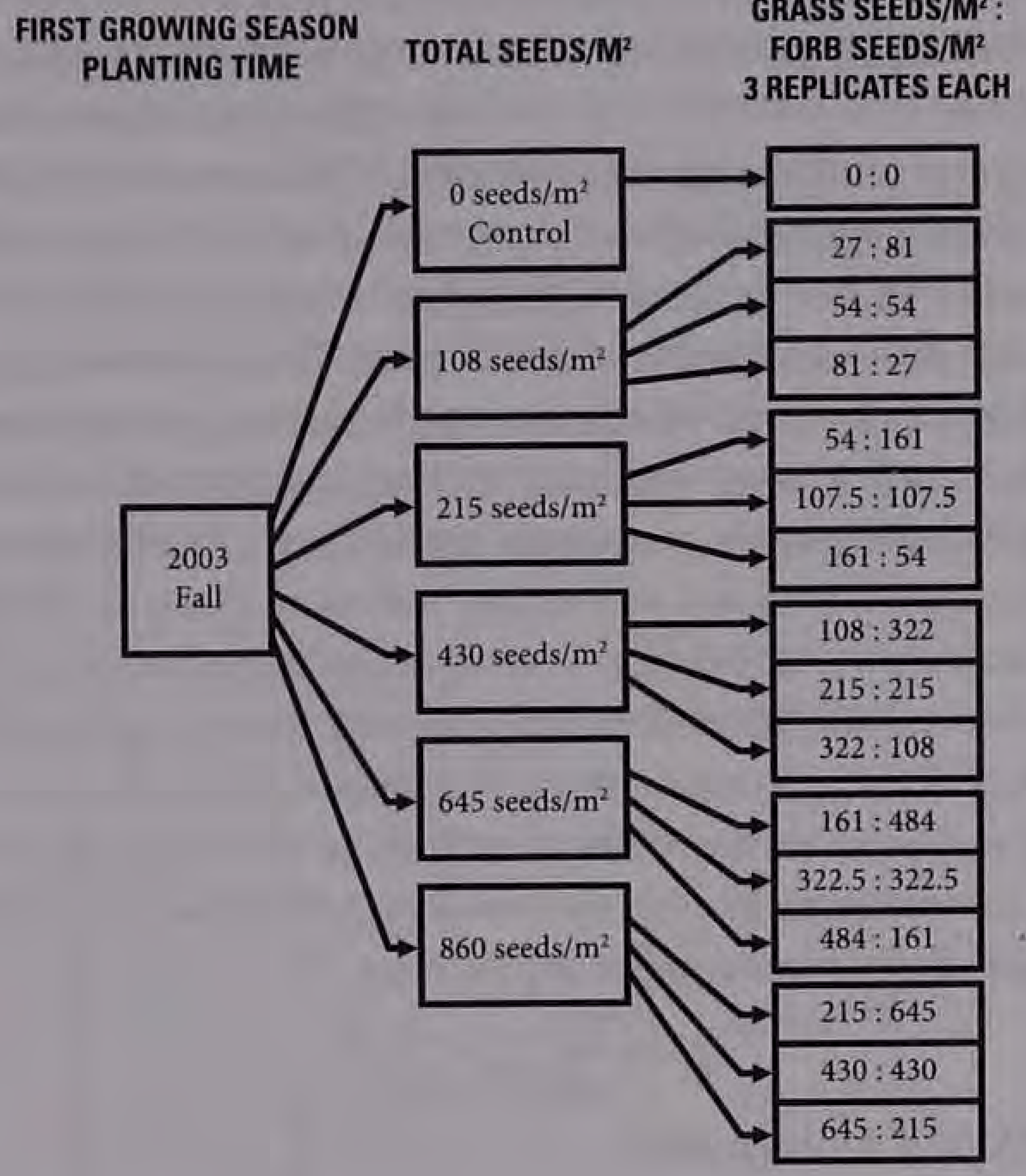
The study site is located at the St. Croix Watershed Research Station (SCWRS) in east-central Minnesota. Previously in row-crop agriculture, the site was fallow for ten years prior to initiation of this study and was colonized by exotic cool-season grasses dominated by *Bromus* and *Poa* species. Soils on the site are Chetek sandy loams (0%-2% slopes, coarse-loamy, mixed, eutric glossoboralfs) (Soil Survey Staff 2008). Site preparation was conducted for one full year prior to planting. All old-field vegetation was treated with glyphosate, burned, and tilled in the spring-summer of 2001. The bare soil was retreated with glyphosate to control weeds until the time of planting. The study site was burned early each spring as a default fire regime, except during the first growing seasons of plantings.

All seed used in this project was of local St. Croix ecotype origin, having sources within a roughly 25-mile radius of the SCWRS. Seed mixes contained 69 species in 2003 and 67 species in 2004 (see appendix A for lists). For all seed-related calculations we refer to numbers of seeds rather than to weights of seeds. Seed numbers reflect a biological reality, the maximum potential establishment, whereas more commonly used seed weights have little meaning for a mixture of species. Pure hard seed per weight of bulk (seed/chaff) was determined by counting at least 400 seeds from a randomly selected bulk sample of each species. Pure hard seed was visually determined using 10-100x magnification and slicing of seeds to determine complete, intact embryos.

EXPERIMENTAL DESIGN AND SAMPLING

One hundred forty-four contiguous 10 m by 10 m plots were randomly assigned a full factorial compliment, with controls, incorporating five seeding rates, three grass to forb ratios (GFR), and two planting times (spring/fall) (Figure 1). Three replicates of each treatment combination were planted in fall 2002, spring 2003, fall 2003, and spring 2004, with the exception of omitting two seeding rates in the latter two plantings and two GFRs in the spring 2003 plantings due to space constraints. We planted all plots by hand-broadcasting seed mixed with sand/vermiculite (to achieve a consistent seeding rate by increasing the bulk volume of seed mixture).

We tested seeding rates of 108, 215, 430, 645, and 860 seeds/m^2 (10, 20, 40, 60, and 80 seeds/ft^2, respectively) to encompass the range of common seed mixes for CRP plantings. We examined issues of seed-mix composition with three ratios of grass to forb seed (by seed number): 25:75, 50:50, and 75:25. For this ratio "grass" includes four species that comprise the bulk of many commonly planted restorations: Andropogon gerardii (big bluestem), Elymus canadensis (Canada wild rye), Schizachyrium scoparium (little bluestem), and Sorghastrum nutans (Indiangrass). These grasses are typically aggressive species that can dominate plantings in our region and are common in CRP plantings.



FIRST GROWING SEASON	PLANTING TIME	TOTAL SEEDING RATE (SEEDS/M^2)	GRASS : FORB RATIOS (BY NO. OF SEEDS)
2003	fall	108, 215, 430, 645, 860, 0 - control	25:75, 50:50, 75:25
2003	spring	108, 215, 430, 645, 860, 0 - control	50:50
2004	fall	215, 430, 860, 0 - control	25:75, 50:50, 75:25
2004	spring	215, 430, 860, 0 - control	25:75, 50:50, 75:25

Figure 1. Experimental treatments. Diagram (top) shows full factorial expansion of line 1 of the table. In total, 144 plots, 3 replicates of 38 treatment combinations (2 first growing seasons, 2 planting times, 5 (or 3) seeding rates, and 3 grass : forb ratios, plus unseeded controls). Note: To quickly convert seeds/m2 to seeds/ft^2 divide by ~10.

To achieve unbiased seed mixes, all species were represented by equal numbers of seeds within their appropriate grass or forb fraction. For example, an 860 seeds/m^2, 25:75 GFR mix had 215 grass seeds/m^2 divided equally among the four grass species. Thus each grass species was planted at 54 seeds/m^2. Similarly for forbs, in this case there were 645 forb seeds/m² split equally among the 63 forb species, resulting in a rate of 10.2 seeds of each forb species/m². This equal-seeds-per-species concept is in contrast to common restoration practices where seed mixes contain unequal species contributions, typically weighted toward inexpensive species, but is useful in this study to equalize the chances of establishment among species. It is important to note that we seeded the same number of species (69 for 2003, 67 for 2004) in all plots (except controls) in a given establishment year. With our approach a higher seeding rate increases the number of seeds, not the number of species.

Plots were sampled during their second and third growing seasons (2004-05 and 2005-06). Analysis presented here lumps both sampling years for increased statistical power, but separate analysis showed trends to be consistent from year to year. Sampling was conducted using a modified Whittaker method (Barbour et al. 1987) similar to a belt transect. Each plot was sampled with twenty 0.05 m² quadrats for a total sample area of 1 m² (1% of each plot). Quadrats were distributed in two randomly placed and nonoverlapping linear transects of ten contiguous quadrats each. In each quadrat the rooted presence or absence of each species, including weeds (nonplanted species), was recorded. Rooted abundance of each species was then estimated as the number of quadrats (out of 20 per plot) in which a species was present. This allowed calculation of Simpson's diversity index as 1/D, where S=number of species present, n_i= rooted abundance of a given species (i), and N= the rooted abundance of all species. Any species present in the plot but not found in quadrats were recorded separately to calculate plot-scale richness.

$$D = \sum_{i=1}^{s} (n_i / N)^2$$

STATISTICAL ANALYSIS

All statistical analysis was performed using JMP software. For this analysis we combined timing treatments (spring/fall and 2003/2004 planting times) since separate analysis of timing treatments revealed no major differences in trends, and combining treatments increased statistical power (larger N). Within each seeding rate we used Kruskal-Wallis tests to analyze differences in diversity indices and richness between GFRs. To analyze trends in diversity, richness, and plant density across seeding rates we used ANOVA with data grouped by forb, grass, or total seeding rates as appropriate. We used MANOVA to compare diversity indices of spring-versus fall-planted plots.

RESULTS

In general, plots established successfully, yielding gradients in richness, diversity, and plant density that were often visually apparent between treatments. Sixty-three forbs and all four major grass species appeared at least once among the 144 plots, including conservative species such as *Lithospermum* sp. (puccoon). Mean plot-richness values per seeding treatment ranged from 10.5 (SD±4.9) in the lowest seeding rate and forb percentage plots, to 37.8 (SD±12.9) in the highest seeding rate and forb percentage plots. Mean Simpson diversity values per seeding treatment ranged from 2.7 (SD±1.4) in the lowest seeding rate and forb percentage plots, to 15.8 (SD±7.3) in the highest seeding rate and forb percentage plots. Mean native plant density (plants/m²) values per seeding treatment ranged from 12.7 (SD±9.0) in the lowest seeding rate and forb percentage plots, to 85.8 (SD±30.8) in the highest seeding rate and forb percentage plots. Despite favorable establishment of planted species, the overall appearance of many plots was weedy, with nearly continuous cover of *Trifolium repens* (white clover). Proposed reasons for this are discussed later. Typical annual weeds such as *Setaria* spp. (foxtail) and Conyza canadensis (mare's tail) were abundant initially in spring and fall plantings, respectively, and declined in subsequent growing seasons.

EFFECTS OF SEEDING RATES ON DIVERSITY, DENSITY, AND RICHNESS

Floristic diversity (Simpson's) and native plant density were significantly positively correlated with total seeding rate when plots were grouped by forb percentage treatment (Figure 2). This occurred despite equal diversity potential

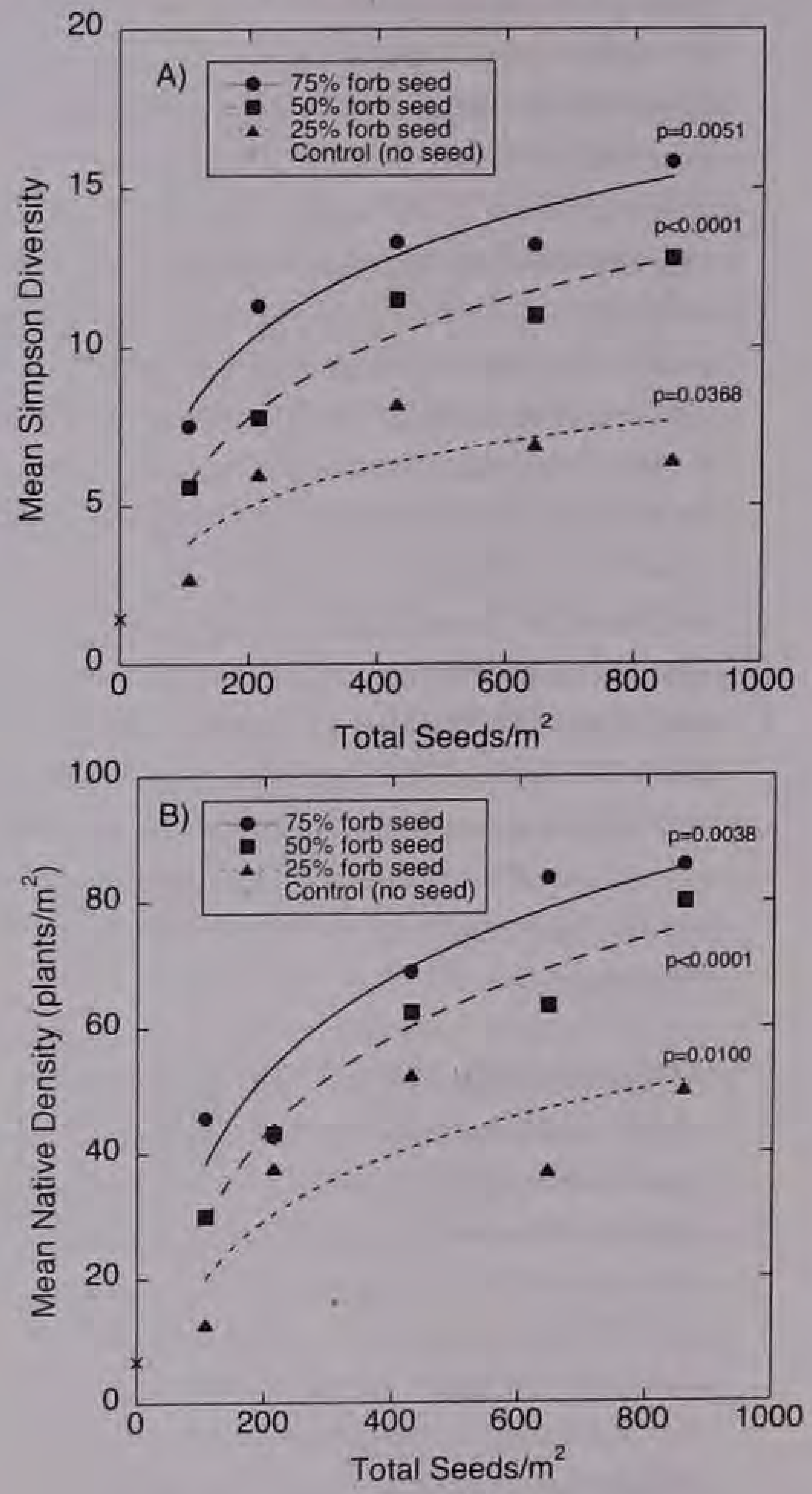


Figure 2. Effects of total seeding rate on (A) mean Simpson diversity, and (B) mean native plant density. Log regressions of all data points (means presented for clarity) show significant increases with seeding rate when plots were grouped by forb-percentage treatment. Error bars omitted for clarity; SD about the mean ranged from ±1.35-7.27 for diversity and ±8.96-53.73 for density.

from planting equal numbers of species in each treatment. This indicates that, at any given seeding rate, increasing the forb percentage results in elevated mean diversity and plant density values. The log fit of these data suggests that while diversity and density continue to increase up to at least 860 seeds/m^2, the greatest gains occur when the seeding rate is increased from low to moderate levels, such as 430 seeds/m^2.

The trends in Simpson diversity and native plant density also held true for richness at quadrat, transect (not shown), and plot scales. Figure 3 highlights these relationships between forb percentage treatments for the lowest and highest seeding rates (means presented for clarity). At each total seeding rate, mean values for diversity and richness at all scales increased with increasing forb percentage. Within each seeding rate, differences in parameter means were not always significant, but the trend was consistent across all seeding rates, and differences became increasingly significant at higher seeding rates. At the highest rate (860seeds/m^2), means for diversity indices and richness at all scales were significantly different between each forb percentage treatment (Figure 3, B).

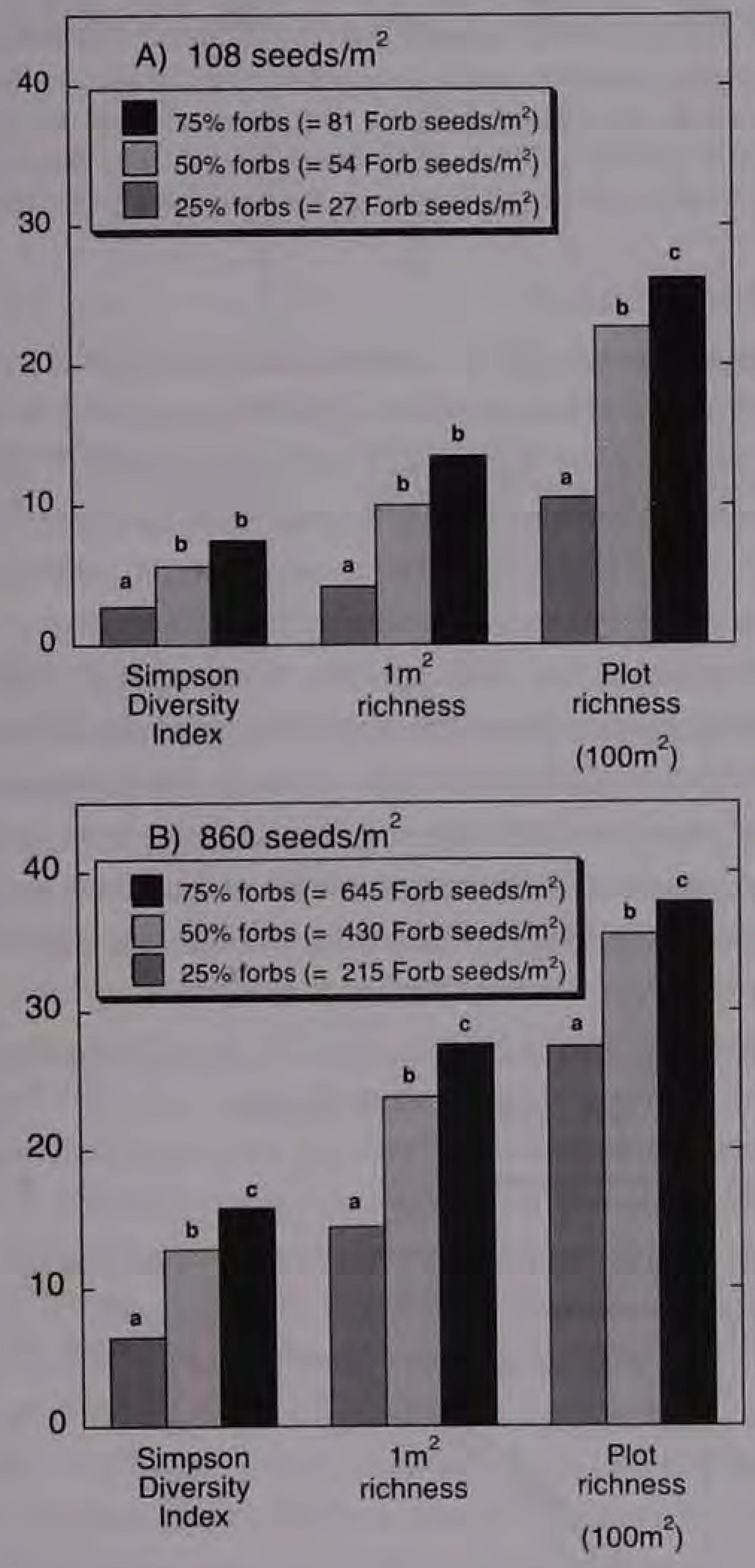


Figure 3. Examples of mean diversity and richness (in no. of species) at (A) 108 seeds/m2 and (B) 860 seeds/m2 with three grass : forb ratios (% forbs). Different letters above bars indicate significant differences between treatments; same letters indicate non-significant differences (Kruskal-Wallis, $p<0.05$). Only highest and lowest total seeding rates are shown, but trends are consistent for all total seeding rates and show increasing significance at higher rates. Grass : forb ratios convert to forb seeding rates by multiplying % forbs by total seeding rate. Error bars omitted for clarity; SD about the mean ranged from ±1.35-7.27 for diversity, ±3.43-10.38 for 1m2 richness, and ±4.89-12.95 for plot richness.

The elevation of diversity, native density, and richness values in higher-forb-percentage treatments indicates that forb seeding rate has a greater effect on diversity, density, and richness than does grass seeding rate. This conclusion was supported by log regressions of forb and grass seeding rates to diversity (Figure. 4). Native plant density shows significant positive correlations to both forb and grass seeding rates ($p \le 0.0002$), but comparison of establishment rates again shows the greater influence of forbs (not shown).

While the variability in our data was high (e.g., Figure 4), regressions had statistical significance due to the large number of plots and data points (n=227). Large variability in this data is to be expected due to the range of environmental variables that affect restorations. The degree of variability in our data matches the real-world experience that no single factor is a guarantor of restoration "success." It does, however, indicate forb seeding

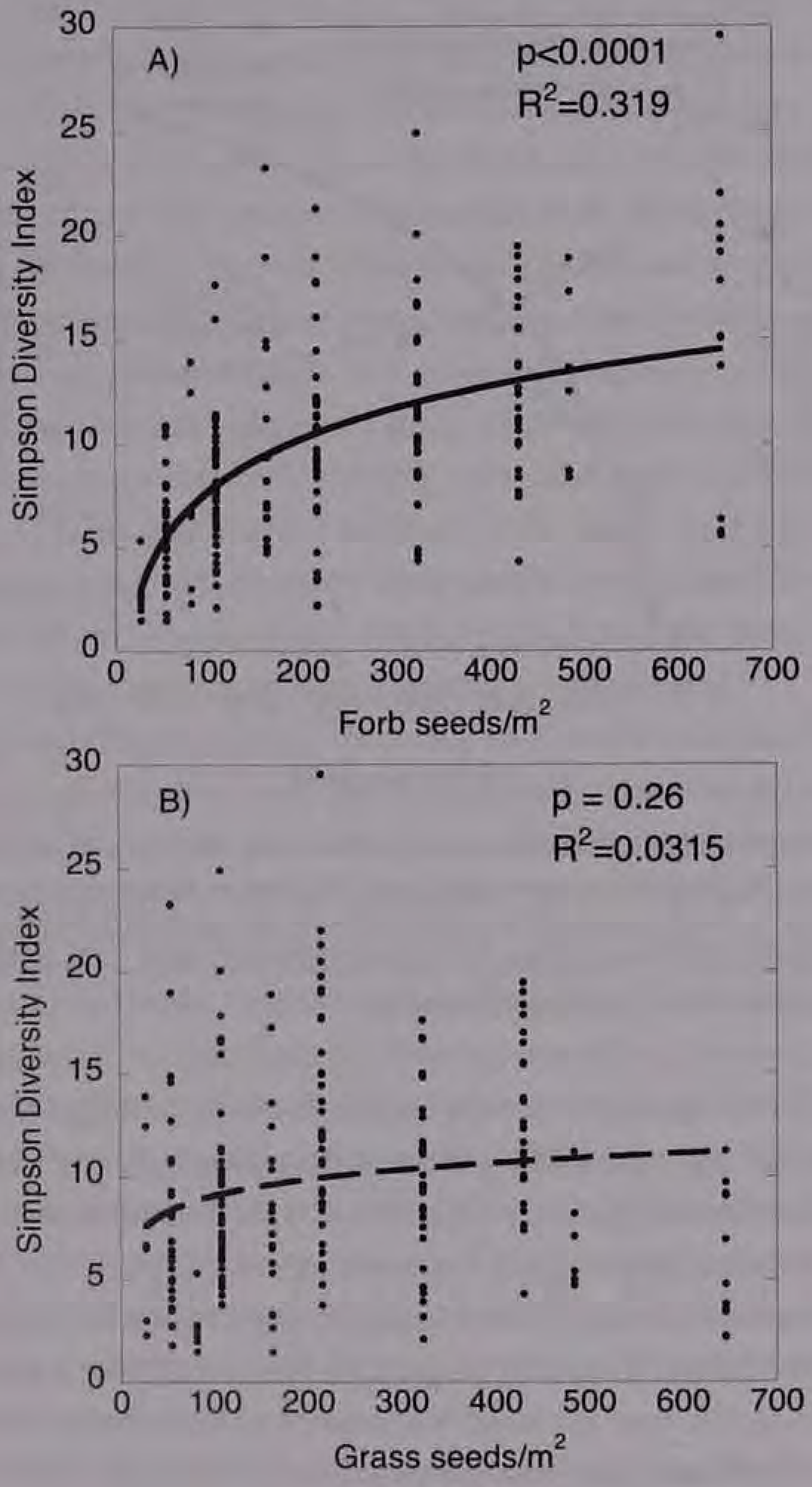


Figure 4. Simpson diversity as a function of (A) forb seeding rate and (B) grass seeding rate. Diversity increases significantly only with forb seeding rate.

rate, explaining about 32% of the variability in diversity, as a major controlling factor that needs to be considered in reconstructing diverse habitats.

EFFECTS OF FORBS AND DIVERSITY ON WEEDS

To better understand the relationship between forb seeding rate and diversity we examined their relationships to forb density and weed density. Forb density showed a significant positive correlation with forb seeding rate in a log regression (Figure 5). Over much of the seeding rate range, forb density increased by 30%-50% as forb rate doubled. Conversely, weed density showed a significant negative correlation to forb seeding-rate (Fig. 5) with weed density decreasing by 9%-10% as forb rate doubled. Weed potential (i.e. seed bank) was assumed to be equal across all plots, and the mean baseline weed density was quantified at 49.9 (SD±12.6) weeds/m² in the 15 unseeded control plots. Changes in weed density from that of the control plots may be attributable to competitive effects of the applied forb-seeding treatments, thus the negative correlation of weed density to forb seeding rate may be due to the increase in forb density with forb seeding rate. In brief, forbs appear to be responsible for suppressing weeds in our plantings, and adding more forbs yields fewer weeds; as a rough estimate, quadrupling the forb rate decreases weeds by 15%-20% (see appendix B for common weed species).

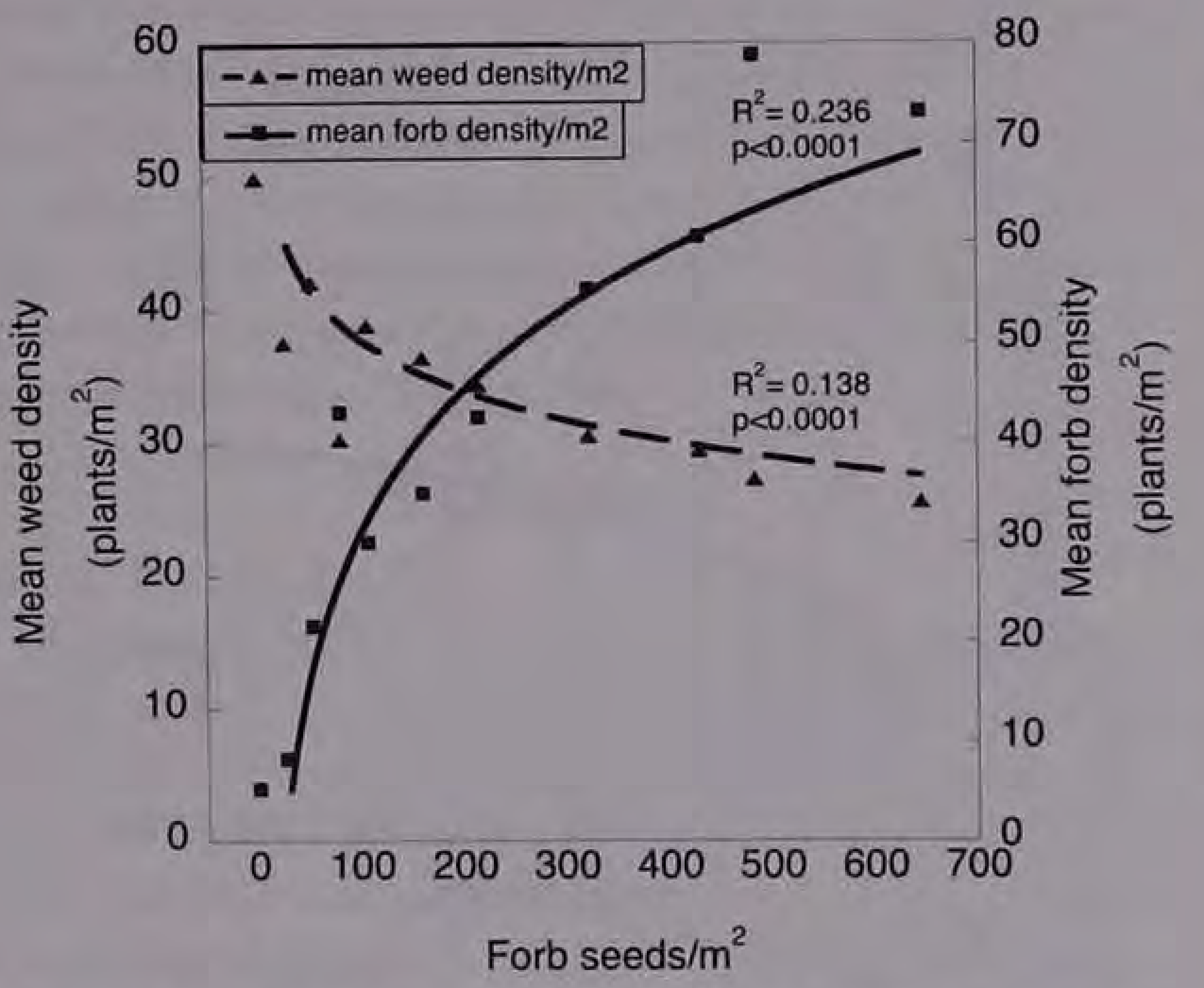


Figure 5. Effects of forb seeding rate on mean forb density and mean weed density. Log regressions show significant increase in forb density and decrease in weed density with increasing forb seeding rate. Error bars omitted for clarity; SD about the mean ranged from ±9.01-51.85 for forb density and ±5.48-13.01 for weed density.

We also see a significant negative correlation between diversity and weed density (Figure 6). Since neither parameter was directly manipulated we cannot conclusively claim a causal relationship, but there may be additional weed suppression occurring that is attributable directly to diversity effects rather than to forb seeding rate. We explore this interaction in the discussion.

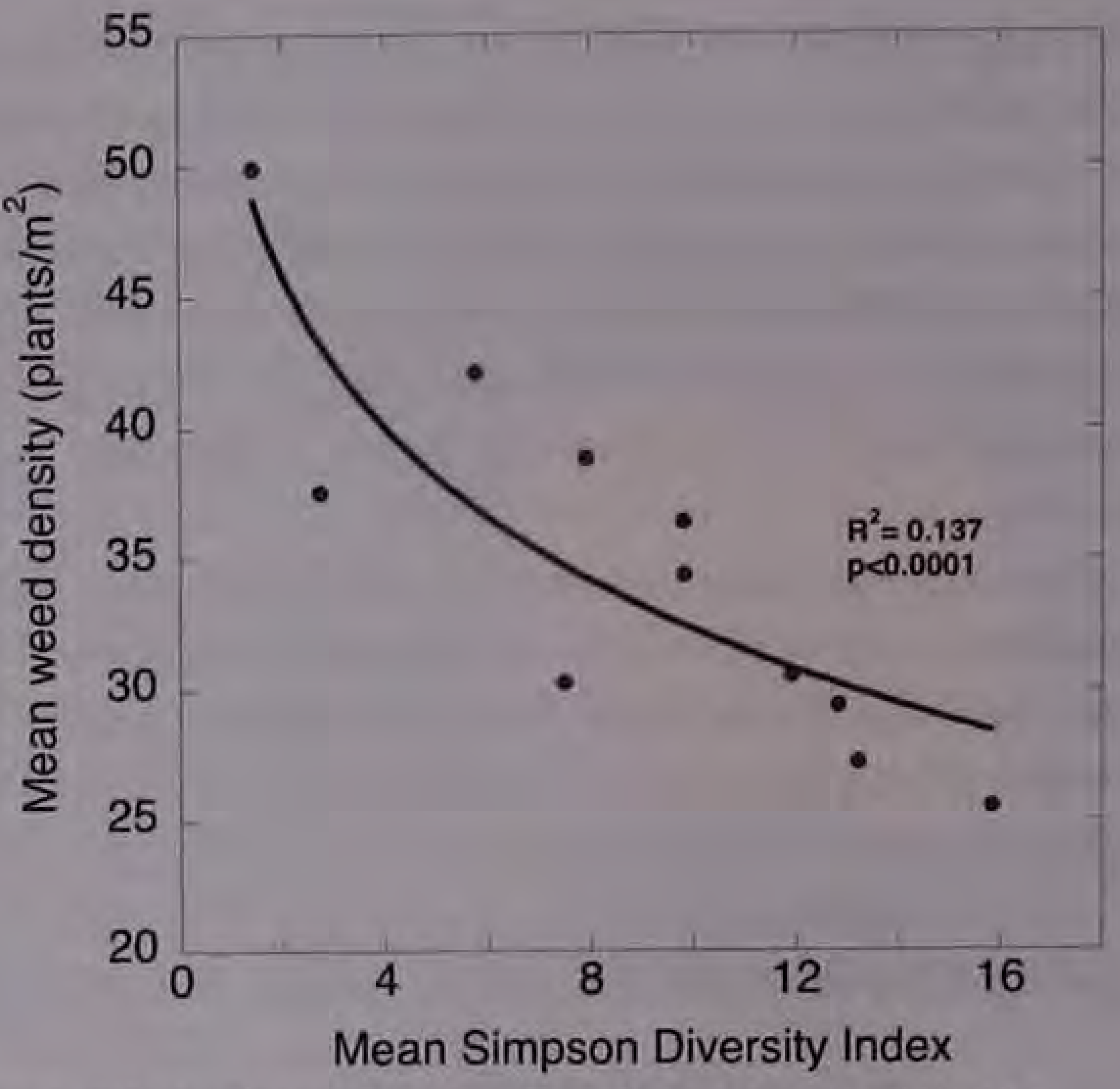


Figure 6. Effects of mean Simpson diversity on mean weed density. Log regression of all data points (means presented for clarity) shows a significant decrease in weed density with increased diversity. Error bars omitted for clarity; SD about the mean ranged from ±1.35-7.27 for diversity (X-dimension) and ±5.48-13.01 for weed density (Y-dimension).

PLANTING SEASON

Comparison of spring- versus fall-planted plots revealed no significant effect on diversity (Figure 7). Although diversity values were equivalent for spring and fall plantings there were differences for some individual species (Figure 8). We quantified these responses via estimated establishment rates based on measured abundances and known numbers of seeds planted for each species. Some species, including *Lespedeza capitata* (bush clover), *Heliopsis helianthoides* (ox-eye sunflower), and *Dalea* spp. (prairie clovers) showed significantly higher establishment for spring planting (Figure 8). Other species, including *Aster laevis* (smooth aster) and *Solidago rigida* (stiff goldenrod), established significantly better when fall-planted (Figure 8).

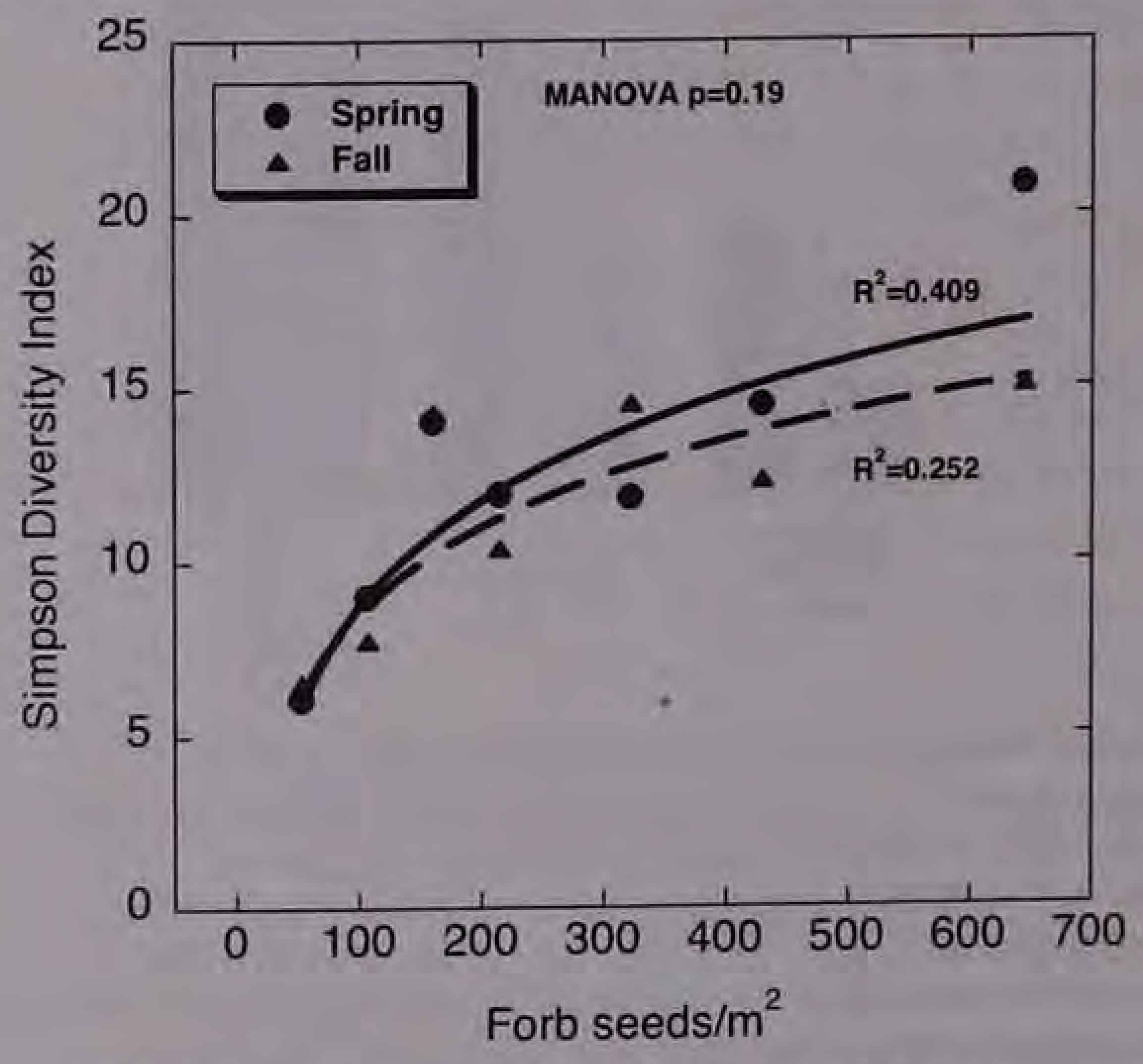


Figure 7. Simpson diversity in spring- and fall-planted plots (means presented for clarity, curve fit through all data). There is no significant difference in diversity between planting times.

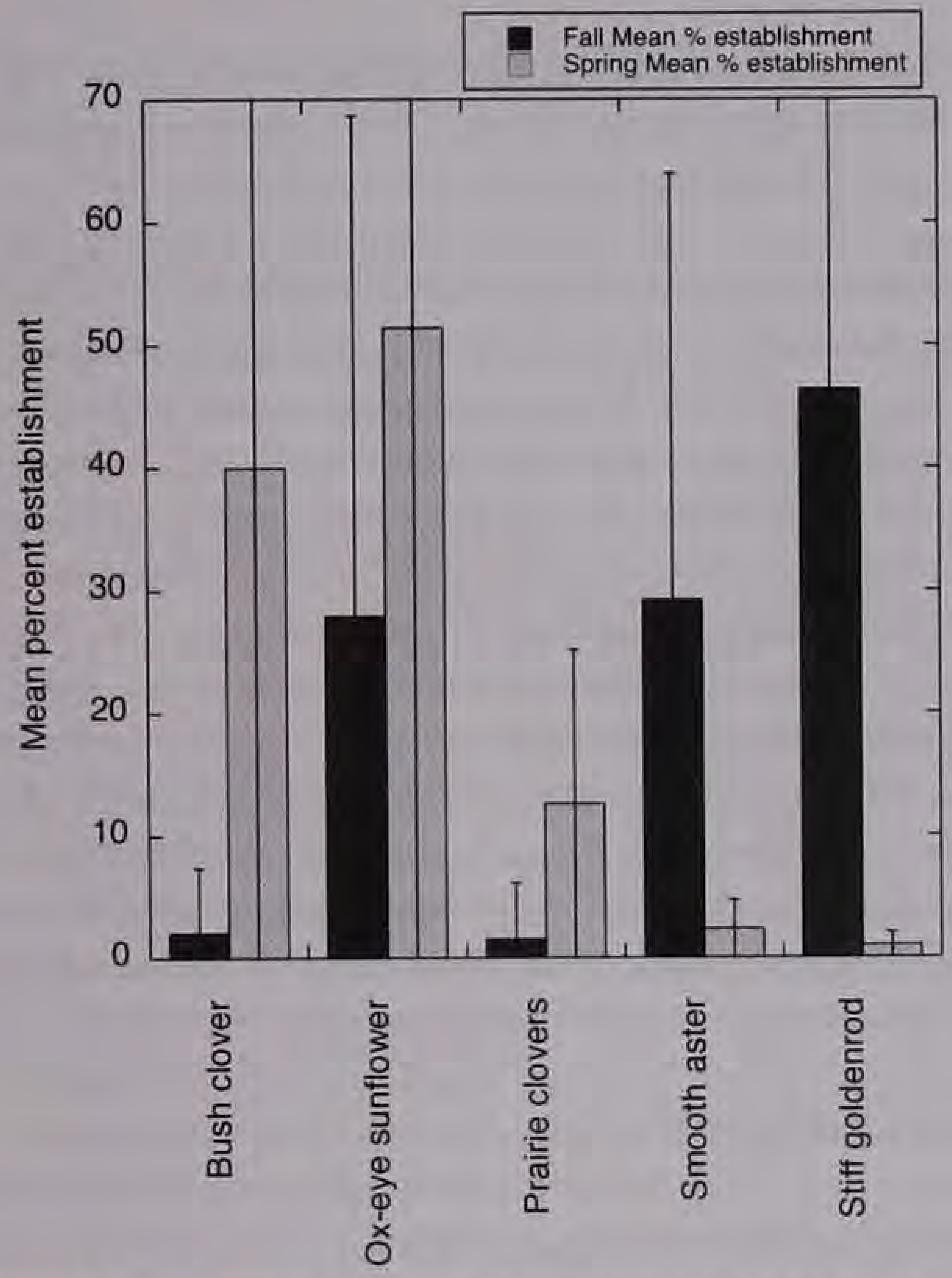


Figure 8. Comparison of mean establishment rates following spring or fall planting for selected species. Data are from the second growing season of plots planted in fall 2002 and spring 2003; other planting years confirmed these findings. All apparent differences are significant (Kruskal-Wallis, p =0.06 to <0.0001) despite high variability; error bars represent ±1SD.

DISCUSSION

This study shows that, within the context of tallgrass prairie reconstruction, floristic diversity, native plant density, and weed suppression can all be improved by planting reconstructions with high absolute and proportional seeding rates of forbs. This is contrary to our hypothesis that low seeding rates would be sufficient to attain diverse restorations, but in agreement with our hypothesis that higher proportions of forb seed and lower proportions of grass seed would result in higher diversity. Low-density plantings were expected to be able to fill in over time from diverse, though sparsely established, plant assemblages. However, low seeding rates and low forb seeding rates, in particular, yielded not only lower plant density, but significantly lower diversity and richness than high seeding rates. This occurred despite equal richness potential in all plots (same number of species planted). Reduced diversity and richness at low seeding rates, especially low forb-seeding rates, indicates that many species failed to establish and may be unavailable to contribute to the future diversity of the mature planting. This is comparable to Martin et al. (Martin et al. 2004, 2005) who also found reduced richness and plant density at low seeding rates at the 1 m^2 scale. This suggests that rapid site capture through higher-density planting of diverse seed mixes is more likely to promote long-term diversity, reduce invasion by weed species, and result in successful restorations.

In tallgrass prairie reconstructions it is common practice to plant many more forb species than grass species, yet grasses are often planted at a much higher seeding rate. Dickson and Busby (2009) substantiated concerns about the negative impacts of high grass seeding rates on forb establishment by demonstrating that high seeding rates of dominant grasses could impair forb cover, biomass, and richness. This study confirms and extends those impacts to include diversity measures. In this study diversity levels are seen to be lowest when grasses are 75% of the seed mix (by number of seeds, not weight) and conversely are greatest when forbs are 75% of the seed mix. Forb seeding rate explained 32% of the variation in diversity values, and is, thus, a major variable that should be carefully attended to in all prairie and grassland reconstructions. Typical restorations achieve high seeding rates by planting large amounts of grass seed of few species accompanied by a much smaller number of forb seeds of many species. This study shows that practice to be counterproductive for achieving diverse restorations.

It is not surprising that in this study forbs exert a greater influence on diversity than grasses because more forb species were planted than grasses. This approach was used to approximate relative grass and forb richness of typical seeding mixes while demonstrating that forb seeding rates are the driver of floristic diversity. While this conclusion may seem self-evident, widespread pursuit of the goal of high diversity has not always led to a corresponding shift in seeding mixes to reflect the driving role of forb seeding rate or the potential consequences of excessive grass seeding rates. It is also worth reinforcing that although each treatment had an equal number of species and, thus, an equal diversity potential, treatments with greater forb seeding rates had more seedlings of more species establish than plots with lower seeding rates. Simply stated: when more forb seeds were planted, more plants of more species established; doubling the forb seeding rate produced a 30%-50% increase in diversity.

Though highly useful, diversity indices alone do not fully encompass the qualities that are desired in a successful restoration. High native plant density is also important to maximize native plant populations and to prevent establishment of weeds that may be ecologically as well as aesthetically detrimental to the success of the planting. Native plant density was greatest at high forb seeding rates and high percentages of forb seed. Total native plant density increased with forb rate when forbs were at least 50% of the seed mix. Thus, per seed, the diverse-forb mix created more plant density than did the mix of four major grasses. This is an important point, given the consideration that many forb seeds are more expensive than grass seeds; on a seed-for-seed basis they may also be more likely to yield a plant.

Forbs were also found to reduce weed establishment. Weed density decreased below that of control plots as forb seeding rate and forb density increased. Higher forb seeding rates increase forb density, which in turn decreases weed density. This indicates that regardless of effects on diversity, increased forb seeding rates benefit restorations through weed suppression and increased establishment of native plants. Piper et al. (2007) buttress this conclusion with their findings that planting more-species-rich seed mixes (including more forbs) decreases weed cover while yielding higher diversity.

Weed density was negatively correlated with diversity in this study. Neither variable was directly manipulated, so we cannot conclusively claim a causal relationship. However, as previously discussed, weed growth declined consistently from that of unseeded control plots as forb seeding rate (an independent variable) and forb density increased; weed declines and forb increases were correlated with increased diversity. It is reasonable to conclude that diversity may be contributing to weed suppression. This supports work by Fargione et al. (2003), Tilman (1997), Biondini (2007), and others that diversity limits invasibility and agrees with the finding of Piper et al. (2007) that more-diverse plantings have lower percentages of weed ("resident species") cover. However, field observations indicate that a limited set of forb species constitute the bulk of the measured forb density and therefore are most likely to be responsible for controlling weed growth. If this is true, then it is not truly diversity that suppresses weed growth and reduces invasibility, but rather it is the competitive nature of a particular suite of early establishing species. In our experience and area these species include *Achillea millefolium* (yarrow), *Monarda fistulosa* (bergamot), *Ratibida pinnata* (yellow coneflower), Rudbeckia hirta (black-eyed Susan), and *Solidago rigida* (stiff goldenrod), which are commonly thought of as pioneer species in restorations. These pioneering forbs may act as a nurse crop to suppress weeds and foster so-called conservative species that are otherwise hindered by excessive weed growth. However, the role of pioneer species is not entirely clear, as Weber (1999) has suggested that excessive planting of pioneer species may limit establishment of conservative species.

Easily established pioneer species often constitute the bulk of the forb seeds planted in a restoration, so understanding their role is important. In this study, equal numbers of each forb species were planted for a given seeding treatment, so the number of seeds of these pioneer or possibly weed-controlling species was reduced, relative to common restoration practices. We hypothesize that there is an optimum proportion of weed-controlling pioneer species relative to the total seeding rate that will result in improved floristic diversity. Further research is needed on the concept of weed-controlling species, on optimum seeding rates of specific forbs, and on the role of forbs in restorations generally. Without this research it remains unclear whether and how pioneering forbs may act as a nurse crop for conservative species or contribute to a causal link between diversity and weed density. For management purposes, however, it is currently important simply to note the triple benefits of using high rates of diverse forb seed mixes: diversity and density of natives are increased and weed growth is suppressed.

CONSIDERATIONS FOR RECONSTRUCTIONS

Increasing forb-seeding rates enhances native floristic diversity, native plant density, and weed suppression. In this study, doubling the forb seeding rate produced a 20%-50% increase in diversity and density. At any given seeding rate, planting more forbs and less grass yields more diversity. Diversity does not increase with grass seeding rate. Weed density declines as native plant diversity increases. Achieving more-diverse restorations, by planting more forbs, can reduce weed problems. In this study, as the forb seeding rate increased by about four times, weed density decreased by 15%-20%. Spring and fall plantings result in equal diversity, but differ in species composition, thus incorporating both spring and fall seeding into a reconstruction site is beneficial.

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Appendix A. List of species planted ("2002" applies to plantings in fall 2002 and spring 2003, "2003" applies to plantings in fall 2003 and spring 2004).

SCIENTIFIC NAME	COMMON NAME	2002	2003
Achillea millefolium	Yarrow	x	x
Agastache foeniculum	Anise Hyssop	x	x
Agastache nepatoides	Yellow Hyssop	x	x
Allium stellatum	Prairie Onion	x	x
Amorpha canescens	Lead Plant	x	x
Andropogon gerardii	Big Bluestem	x	x
Anemone cylindrica	Thimbleweed		x
Asclepias incarnata	Swamp Milkweed	x	x
Asclepias tuberosa	Butterflyweed	x	x
Aster laevis	Smooth Blue Aster	x	x
Aster novae-angliae	New England Aster	x	x
Aster oolentangiensis	Sky Blue Aster	x	x
Aster sagittifolius	Arrow Aster	x	
Aster sericeus	Silky Aster	x	
Astragalus canadensis	Canada Milkvetch	x	x
Baptisia alba	White Wild Indigo	x	x
Baptisia bracteata	Cream Wild Indigo	x	x
Bouteloua hirsuta	Hairy Grama		x
Campanula rotundifolia	Harebell	x	x
Cassia fasciculata	Partridge Pea	x	x
Ceanothus americanus	New Jersery Tea	x	x
Cirsium discolor	Field Thistle	x	x
Coreopsis palmata	Prairie Coreopsis	x	x
Dalea candida	White Prairie Clover	x	x
Dalea purpurea	Purple Prairie Clover	x	x
Desmodium canadense	Tick Trefoil	x	x
Desmodium illinoense	Illinois Tick Trefoil	x	x
Elymus canadensis	Canada Wild Rye	x	x
Elymus hystrix	Bottle Brush Grass	x	x
Eupatorium perfoliatum	Boneset	x	x
Gentiana alba	Cream Gentian	x	x
Geum aleppicum	Avens	x	x
Geum triflorum	Prairie Smoke	x	x
Helianthus giganteus	Giant Sunflower	x	
Helianthus hirsutus/laetiflorus	Rough Sunflower	x	x
Helianthus occidentalis	Naked Sunflower	x	x
Heliopsis helianthoides	Ox-eye Sunflower	x	x

Appendix A. continued.

SCIENTIFIC NAME	COMMON NAME	2002	2003
Heuchera richardsonii	Alumroot	x	x
Hypericum pyramidatum	Great St. Johns Wort	x	x
Koeleria cristata	June Grass	x	x
Lespedeza capitata	Bush Clover	x	x
Liatris aspera	Rough Blazing Star	x	x
Liatris ligulistylis	Meadow Blazing Star	x	
Liatris punctata	Dotted Blazing Star		x
Lithospermum sp.	Puccoon		x
Lobelia spicata	Pale Spike Lobelia	x	x
Lupinus perennis	Lupine	x	x
Monarda fistulosa	Bergamot	x	x
Monarda punctata	Dotted Mint	x	x
Oenothera rhombipetala	Sand Primrose	x	x
Penstemon digitalis	White Beardtongue	x	x
Penstemon gracilis	Slender Beardtongue	x	x
Penstemon grandiflorus	Large Beardtongue		x
Phlox pilosa	Prairie Phlox	x	x
Potentilla arguta	Cinquefoil	x	x
Pycnanthemum virginianum	Mountain Mint	x	x
Ratibida pinnata	Yellow Coneflower	x	x
Rosa arkansana	Wild Rose	x	
Rudbeckia hirta	Black-eyed Susan	x	x
Schizachyrium scoparium	Little Bluestem	x	x
Scrophularia lanceolata	Figwort	x	x
Sisyrinchium campestre	Bue-eyed Grass	x	x
Solidago nemoralis	Gray Goldenrod	x	x
Solidago rigida	Stiff Goldenrod	x	x
Solidago speciosa	Showy Goldenrod	x	
Sorghastrum nutans	Indian Grass	x	x
Sporobolus heterolepis	Prairie Dropseed	x	
Stipa spartea	Needle Grass	x	x
Teucrium canadense	Germander		x
Tradescantia ohiensis	Spiderwort	x	x
Verbena hastata	Blue Vervain	x	x
Verbena stricta	Hoary Vervain	x	x
Veronicastrum virginicum	Culvers Root	x	
Zizia aptera	Heart-leaf Golden Alexander	x	x
Zizia aurea	Golden Alexander	x	x
TOTAL SPECIES PLANTED		**69**	**67**

Appendix B. Common unplanted species ("weeds") occurring in plots (Note: More than 40 unplanted species were found among all plots; those below were by far the most common/widespread). We considered Canada goldenrod a weed due to its extremely aggressive growth in our area.

SCIENTIFIC NAME	COMMON NAME	PHYSIOGNOMY
Ambrosia artemisiifolia	Common Ragweed	annual
Conyza canadensis	Horseweed	annual
Poa sp.	Bluegrass	perennial
Potentilla norvegica	Cinquefoil	annual
Setaria sp.	Foxtail	annual
Solidago candensis	Canada Goldenrod	perennial
Trifolium repens	White Dutch Clover	perennial

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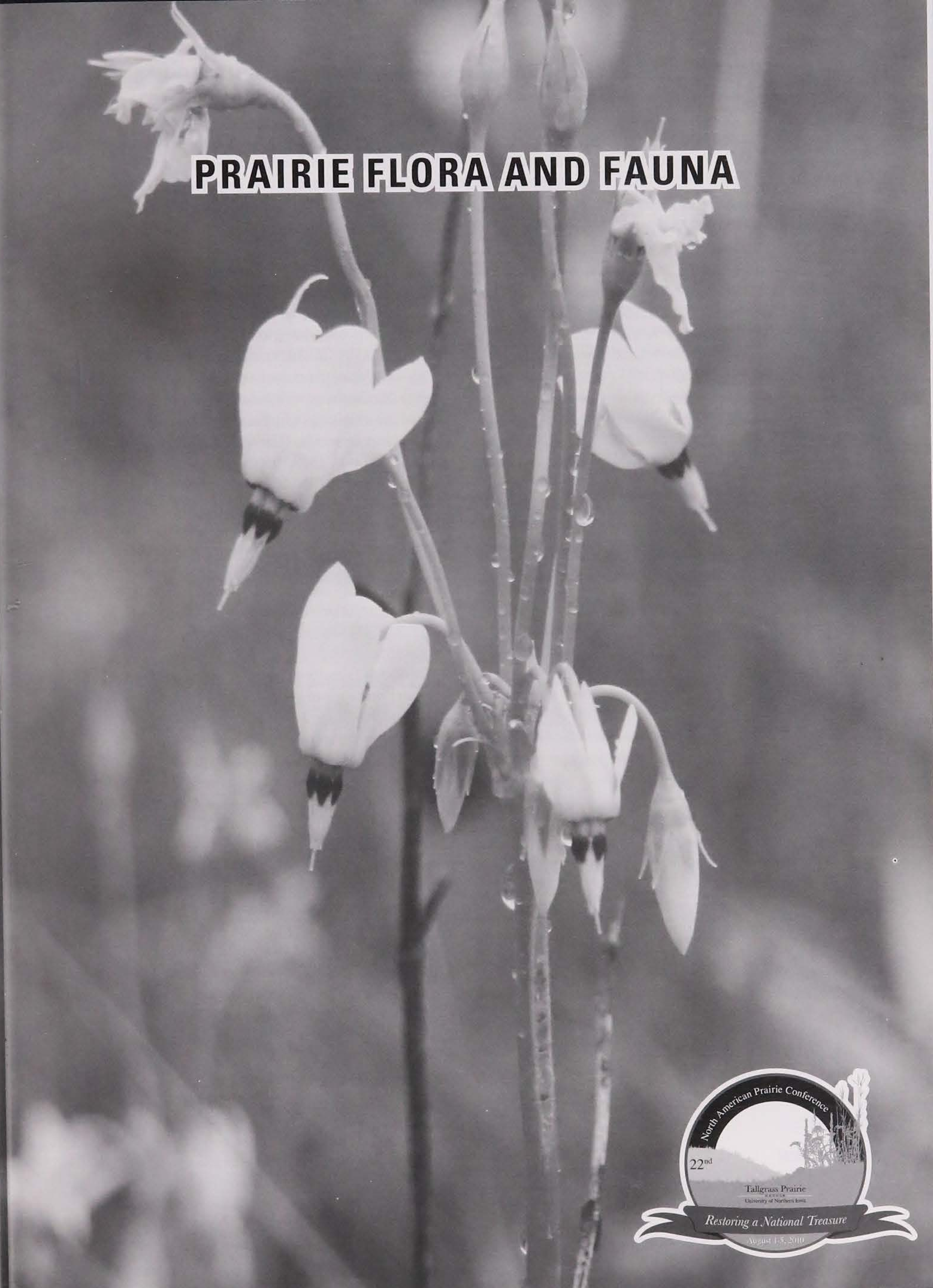

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BIRD COMMUNITIES OF RESTORED PRAIRIES AND OLD-FIELD HABITATS IN AGRICULTURAL AND URBAN AREAS OF WINONA COUNTY, MINNESOTA

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Abstract: Native prairie and savanna habitats in southeastern Minnesota have been mostly eliminated by agriculture and urbanization. Resource management agencies and private landowners have idled former agricultural land and planted prairie vegetation or grassland cover on some sites to restore native plant and animal communities. The goal of this study was to survey bird communities in small (<10 ha) restored prairies and idled old-field habitats in agricultural and urban areas of Winona County, Minnesota. Six sites were surveyed five times each during one or two breeding seasons (May-August, 2008 and 2009). Perched or flushed birds along transects were identified and tallied to determine bird community structure and density. Over 900 birds representing 30 species were recorded during surveys, averaging 11 species/site (range = 7-19 species/site). Six generalist or woody-dependent species (American goldfinch, *Carduelis tristis*; red-winged blackbird, *Agelaius phoeniceus*; song sparrow, *Melospiza melodia*; indigo bunting, *Passerina cyanea*; chipping sparrow, *Spizella passerine*; house sparrow, *Passer domesticus*) were present at four or more sites, and represented >71% of all birds tallied. Species that strongly associate with grassland habitats (bobolink, *Dolichonyx oryzivorous*; dickcissel, *Spiza Americana*; field sparrow, *Spizella pusilla*; grasshopper sparrow, *Ammodramus savannarum*; ring-necked pheasant, *Phasianus colchicus*) represented only 8% of birds sighted and were found at only one or two sites each. Bird densities at most sites ranged from 13-17 birds/ha. Bird community diversity indices were similar at all sites, but bird community structure differed significantly (all Bray-Curtis similarity values <0.6) among all sites. Summer bird communities in small, restored grasslands and old fields of southeastern Minnesota appear to be highly variable and dominated by generalist and woody-dependent species, with grassland specialists present only in the larger (>3 ha) sites surveyed. Small grassland-habitat patch size and proximity of woody vegetation limited the attractiveness of grasslands in this region to native grassland birds.

Key Words/Search Terms: bird diversity, community similarity, community structure, bird density, grassland birds

INTRODUCTION

The majority (nearly 80%) of grasslands in North America have been lost during the past century, mostly as a result of conversion to agriculture or other human activities (Samson and Knopf 1994, Knopf and Sampson 1997). Tallgrass prairies are widely considered to be the single most endangered ecosystem in the United States (Steinauer and Collins 1996). Large-scale losses of prairie have continued to occur in some regions during the past decade (Higgins et al. 2002, Bakker et al. 2006), resulting in further fragmentation and isolation of grassland remnants (Herkert et al. 2003, Cunningham 2005, Askins et al. 2007, Peitz 2007).

Loss and degradation of grassland habitats has resulted in widespread and precipitous declines in abundances of many species of grassland birds in North America (Herkert 1994, Peterjohn and Sauer 1999). Endemic and obligate grassland birds often occur on both federal and state lists of endangered and threatened species, or lists of species of special concern. Loss of breeding habitat for these birds is the most likely cause of their declining numbers (Vickery et al. 1999, Herkert et al. 2003).

Grasslands in many parts of the upper midwestern United States are being restored or managed to enhance habitat for grassland birds (Knopf 1994). Several recent efforts have been initiated to increase the amount of perennial grassland cover available to grassland birds on the landscape, including the conversion of marginal farmland to prairie, the conversion of cropland to perennial grassland cover under the U.S. Department of Agriculture's Conservation Reserve Program (CRP), grassland habitat restoration projects supported by Ducks Unlimited and other conservation organizations, roadside vegetation projects, and biomass fuel plantings (e.g., Knopf 1994, Fritcher et al. 2004, Shochat et al. 2005, Bakker et al. 2006, Borsari and Onwueme 2008, Rahmig et al. 2009).

Grassland bird species differ in their response to changes in their grassland habitats (Knopf 1994, Rahmig et al. 2009). Many grassland bird species are sensitive to the amount of grassland in the landscape (Helzer and Jelinski 1999, Johnson 2001, Winter et al. 2006), and many habitat and landscape features can influence bird use of grasslands, including time since establishment or disturbance, degree of fragmentation/isolation, habitat patch shape, vegetation structure, and management regime (Helzer and Jelinski 1999, Herkert et al. 2003, Fritcher et al. 2004, Shochat et al. 2005, Bakker et al. 2006, Winter et al. 2006). Grassland restoration projects, therefore, can have widely ranging effects on grassland birds (Samson 1980, Knopf 1994).

Many grassland habitats have been restored or established in southern Minnesota in recent years for a variety of purposes (Camill et al. 2004, Cunningham 2005, Borsari and Onwueme 2008, Faber 2010). Restorations, old-field habitats, and CRP lands are scattered widely across this region (Cunningham 2005), creating potential habitat for grassland birds in both rural and urban landscapes. These grassland habitats are mostly small (<10 ha), but often occur in clusters, with other small grasslands nearby (N. Mundahl, personal observation). Management of these grasslands is highly variable, ranging from idle conditions (old fields) to ongoing restorations and annual harvests for biomass fuel production (Borsari and Onwueme 2008). Limited grassland bird research has been conducted on these lands (e.g., Driscoll 2004, Cunningham 2005, Faber 2010).

The objective of the present study was to examine bird communities in a variety of small (<10 ha) grassland habitats in southeastern Minnesota. Specifically, we were interested in comparing bird communities in rural versus urban habitats as well as restored versus old-field grasslands that differed greatly in management.

MATERIALS AND METHODS

STUDY SITES

Birds were surveyed in six small grassland sites in Winona County, Minnesota (Table 1). Two sites (Kriedermacher Farm, Kramer Ridge) were surveyed in 2008 and 2009, whereas the remaining four sites were surveyed only in 2009. Two sites (Kriedermacher Farm, Kramer Ridge) were located in a rural, agricultural setting, and the other four sites were located within the City of Winona, adjacent to residences, schools, and commercial buildings. Three of the sites had been "restored" by seeding with native prairie grasses and forbs, whereas three sites contained plant communities typical of old-field habitats within the region. Table 1 contains a qualitative listing of the common grasses and forbs at each

Table 1. Characteristics of grassland study sites surveyed for bird communities in Winona County, Minnesota, during summer, 2008 and 2009. Plants are listed alphabetically; scientific names are in appendix 1.

	GRASSLAND SITES					
CHARACTERISTIC	Kriedermacher	Kramer R.	SE Tech	Burns V. Ck.	Valley Oaks	Riverbend
SIZE (HA)	4.86	3.00	0.38	1.80	0.81	8.70
TYPE	Restored farm land	Restored farm land	Restored urban land	Old-field urban land	Old-field urban land	Old-field urban land
MANAGEMENT	Annual harvest	None	Annual fall mowing	None	None	Annual spring mowing
COMMON PLANTS	Big bluestem	Big bluestem	Big bluestem	Burr dock	Common milkweed	Black mustard
	Black-eyed susan	Compass plant	Canada thistle	Canada thistle	Goldenrod spp.	Canada thistle
	Canada wild rye	Golden alexanders	Canada wild rye	Common mullein	Queen Anne's lace	Lambs quarter
	Little bluestem	Goldenrod spp.	Compass plant	Goldenrod spp.	Red clover	Reed canarygrass
	Oxeye sunflower	Gray-headed coneflower	Goldenrod spp.	Queen Anne's lace	Reed canarygrass	Virginia wild rye
	Prairie coneflower	Indian grass	Gray-headed coneflower	Reed canarygrass	Rough bedstraw	White sweet clover
	Red clover	Little bluestem	Purple coneflower	White sweet clover	Smooth brome	Yellow sweet clover
	Side oats grama	Partridge pea	Purple prairie clover	Wild parsnip		
	Switch grass	Prairie spiderwort	Side oats grama			
	White sweet clover	Sedge spp.	Wild bergamot			
	Wild bergamot	White wild indigo				
		Wild bergamot				

site. Woody vegetation was not present within any of the study plots, but most sites were adjacent to wooded fence lines, residential plantings, or forest (see individual site descriptions below).

The survey site on the Kriedermacher Farm was a portion of a larger area (7.9 ha) used for biofuel (dry biomass) production (Borsari and Onwueme 2008). Irregularly shaped fields were planted either with mixed native grasses or with mixed grasses and forbs in 2007, and were harvested annually and pelletized for heating fuel for a commercial greenhouse business on site. Bird survey sites were located within three interconnected plots bordered by corn, hay, and soybean fields. Shrubs and trees were located nearby along fence lines, but none were within the biofuel fields. Bluebird nest boxes were scattered along the fence lines.

The Kramer Ridge site is a portion of the Whitewater Wildlife Management Area (managed by the Minnesota Department of Natural Resources), adjacent to the Kriedermacher Farm. Mixed native grasses or mixed grasses and forbs had been planted on 37.4 ha within 13 plots scattered across 5 township sections. Bird surveys were conducted on a single plot bordered by cornfields and forest. A bordering fenceline also contained shrubs and trees, but no woody vegetation grew within the restored plot.

The Southeast Tech site is located on the campus of Minnesota State College Southeast Technical in Winona, Minnesota. Mixed native grasses and forbs had been planted on a portion of a mowed grass lawn (soccer fields) adjacent to a truck drivers' training course. The site is bordered by mowed grass fields and a residential neighborhood and is mowed annually in the fall. Small trees were scattered along one side of the restored prairie.

The Burns Valley Creek site is a riparian grassland habitat bordering a coldwater trout stream. The area is confined within flood control levees that prevent Mississippi River waters from backing up into the creek and flooding the City of Winona. The area is mowed irregularly (every few years) to suppress the growth of woody vegetation. The site is bordered by hotels, small businesses (including a landscaping center and nursery with shrubs and small trees), and the Southeast Tech campus.

The Valley Oaks site is located in a city park, on a hillside with a 20°-30° slope and a westerly aspect. It was a former horse pasture and was mowed regularly for ~10 years after becoming a park. It has not been mowed or managed in any way during the past ten years and is used as a sledding hill during winter. Various bluebird nest boxes have been present on the site for nearly 20 years. The site is bordered by mowed parkland with scattered trees and shrubs, residential lawns, and forest, and is within 400 meters of a native dry bluffside prairie.

The Riverbend survey site is an undeveloped portion of an industrial park created on fill generated by a lake-dredging project in 2002. It was seeded with annual rye to protect the soil from erosion and has been mowed annually to suppress noxious weeds. The site is bordered by city streets, light industry, and two stormwater runoff retention basins. Trees and shrubs were absent in the study plot, but were present along the retention basins.

BIRD SURVEYS

Birds were surveyed along three transects at each study site during each year (May-July 2008, June-August 2009). A fourth transect was added to the Riverbend site after the initial survey date. Transects were 100 m in length except at Valley Oaks, where site size and shape limited the length of transects to 50 m.

Transects were walked slowly with frequent stops by a single observer on each of five survey dates each year. The same transects were used on each date. Perched or flushed birds were identified and tallied, and distance to each bird from the observer was measured with a rangefinder. Birds that flew over a site during surveys, or that could be heard but not located, were not tallied.

Bird data gathered from transects were used to estimate density, diversity, and community similarity. Bird densities were calculated for individual transects with the Hayne-King method for line transects (Buckland et al. 1993, Brower et al. 1998). The five density estimates calculated for a summer (2008 or 2009) for a specific transect were averaged before comparisons to avoid pseudoreplication issues (Krebs 1989). Density comparisons among all six sites were made using only 2009 data with single-factor analysis of variance (ANOVA). Overall bird community diversity at each site was estimated by calculating single Simpson diversity index values (Brower et al. 1998) for the combined data from all transects and dates within the same year for that site. Bird communities in 2009 were compared among all six sites by calculating Bray-Curtis community similarity index values for all possible site-pair combinations. A Bray-Curtis index value >0.6 was interpreted as similar communities between sites being compared, whereas an index value <0.6 was interpreted as significantly different communities. Density, diversity, and community similarity were compared between 2008 and 2009 surveys for the two sites studied in both years (Kriedermacher Farm, Kramer Ridge).

VEGETATION AND LITTER SURVEYS

Vegetation structure (i.e., cover for ground-nesting birds) and litter depth were assessed in late June 2010 at each of the six bird survey sites. Vegetation structure (a combination of vegetation height and density) was assessed by using the visual obstruction method of Robel et al. (1970) at 10 stations spaced 10-15 m apart along a single transect at each bird survey site. Structure was recorded as the highest point above ground where a graduated pole (2.5-cm increments) was completely obscured by vegetation when viewed from a sighting height of 1 m and a distance of 4 m (Robel et al.

1970). Two measurements were taken in opposite directions at each station and averaged. Two measurements of litter depth (nearest mm) also were made 8 m apart at each of the 10 vegetation structure stations by inserting a ruler into the litter until it made contact with the soil. Vegetation structure and litter depth each were compared among bird survey sites with single-factor ANOVA.

RESULTS

During 2008 and 2009 surveys, 918 birds comprising 30 species were observed (Table 2). Bird numbers varied greatly among the sites. The American goldfinch (*Carduelis tristis*) was the only species present at all sites, and it was the most common bird observed during surveys. Eight species (house wren, *Troglodytes aedon*; gray catbird, *Dumetella carolinensis*; red-winged blackbird, *Agelaius phoeniceus*; house sparrow, *Passer domesticus*; indigo bunting, *Passerina cyanea*; chipping sparrow, *Spizella passerine*; and song sparrow, *Melospiza melodia*) were found at four or more sites, and together comprised 67% of all birds observed. Five grassland species (bobolink *Dolichonyx oryzivorous*, dickcissel *Spiza Americana*, field sparrow *Spizella pusilla*, grasshopper sparrow *Ammodramus savannarum*, ring-necked pheasant *Phasianus colchicus*) were found at one or two sites each, with only three sites (Kreidermacher, Kramer Ridge, Riverbend) having any grassland species. These five species represented <9% of all birds observed. In addition, western meadowlarks (*Sturnella neglecta*) were observed at Riverbend, but not during transect surveys.

During 2009 when all sites were surveyed, most sites had similar bird species richness (~9 species/site) and diversity (~0.800) (Table 3). However, the Kreidermacher site had much higher species richness, with 19 species observed. Four species not observed at this site in 2009 were recorded in 2008, for a site total of 23 species for the two years combined. Kramer Ridge also had eight species present in 2008 that were not found in 2009, for a total of 16 species for the two seasons.

Table 2. Numbers of birds observed on transects at six grassland sites in Winona County, Minnesota, summers 2008 and 2009. Additional species observed at sites, but not found on transects, included red-tailed hawk, turkey vulture, bald eagle, American crow, barn swallow, tree swallow, bank swallow, and western meadowlark. Scientific names are in appendix 1.

SPECIES	**GRASSLAND SITES**							
	KREIDERMACHER	**KRAMER R.**	**SE TECH**	**BURNS V. CK.**	**VALLEY OAKS**	**RIVERBEND**	**TOTALS**	**PERCENT**
Sandhill Crane						3	3	0.3
Ring-necked Pheasant	3	1					4	0.4
Mourning Dove	2					13	15	1.6
Ruby-throated Hummingbird	2	2					4	0.4
Red-bellied Woodpecker	4						4	0.4
Downy Woodpecker			1	4			5	0.5
Blue Jay		1					1	0.1
Black-capped Chickadee	4	1	1				6	0.7
House Wren	2	1	6		3		12	1.3
Gray Catbird	7	10	1	1			19	2.1
Eastern Bluebird	11		1		6		18	2.0
American Robin	10				3		13	1.4
Cedar Waxwing	1	2					3	0.3
Yellow Warbler			1			2	3	0.3
Common Yellowthroat	22	33				4	59	6.4
Red-winged Blackbird	68	24	2	21		50	165	18.0
Brown-headed Cowbird	2	2			1		5	0.5
Common Grackle	1	1					2	0.2
Bobolink	8						8	0.9
European Starling		1					1	0.1
House Sparrow			1	17	9	4	31	3.4
Dickcissel	24					6	30	3.3
House Finch				10		77	87	9.5
American Goldfinch	63	20	13	58	6	39	199	21.7
Indigo Bunting	3	2		3		5	13	1.4
Rose-breasted Grosbeak	3						3	0.3
Chipping Sparrow	3		27	5	9		44	4.8
Field Sparrow	5	24					29	3.2
Grasshopper Sparrow	1						1	0.1
Song Sparrow	34	22		44		31	131	14.3
TOTALS	**283**	**147**	**54**	**163**	**37**	**234**	**918**	**100.0**

Bird densities at most sites averaged from 13 to 17 birds/ha (Table 3). Statistical testing indicated significant differences in densities (ANOVA $F_{5,13} = 3.78$, $P = 0.02$) among the six sites, but large density confidence intervals at many sites suggest that these differences are not truly meaningful (Johnson 1999). Densities of grassland bird species were low at all sites, ranging from zero to 3.96 birds/ha.

Table 3. Bird and vegetation characteristics for six grassland study sites in Winona County, Minnesota, summer 2009. Standard deviations are in parentheses.

CHARACTERISTIC	GRASSLAND SITES					
	Kreidermacher	Kramer R.	SE Tech	Burns V. Ck.	Valley Oaks	Riverbend
Total bird species richness	19	8	10	9	7	11
Grassland bird species richness	5	2	0	0	0	2
Simpson diversity	0.896	0.772	0.689	0.772	0.838	0.799
Density (birds/ha)	15.5 (6.8)	13.0 (4.3)	37.5 (16.0)	15.1 (1.8)	17.0 (8.1)	14.8 (6.2)
Visual obstruction (cm)	52 (19)	60 (10)	76 (19)	88 (9)	40 (6)	60 (27)
Litter depth (mm)	12 (11)	59 (28)	4 (5)	49 (29)	44 (15)	18 (24)

Bird communities were significantly different at all sites, based on all Bray-Curtis community similarity values being <0.6 (Table 4). Communities were most similar at Riverbend and Burns Valley Creek (red-winged blackbird, house finch [*Carpodacus mexicanus*], American goldfinch, song sparrow), and at the Kreidermacher and Kramer Ridge sites (common yellowthroat [*Geothlypis trichas*], red-winged blackbird, American goldfinch, field sparrow, song sparrow). In general, nearby sites had more-similar bird communities than did more distant sites.

Table 4. Bray-Curtis bird community similarity matrix for six grassland study sites surveyed in Winona County, Minnesota, summer 2009.

GRASSLAND SITES	GRASSLAND SITES				
	Kramer R.	SE Tech	Valley Oaks	Riverbend	Burns V. Ck.
Kreidermacher	0.477	0.240	0.241	0.366	0.415
Kramer R.	---	0.190	0.110	0.255	0.305
SE Tech	---	---	0.440	0.118	0.212
Valley Oaks	---	---	---	0.074	0.200
Riverbend	---	---	---	---	0.544

For the two sites surveyed in both 2008 and 2009 (Kreidermacher, Kramer Ridge), bird communities displayed both similarities and differences between the years (Table 5). Species richness and diversity varied between the years, but in opposite directions at the two sites. There were no significant differences in bird density between the two years at either the Kreidermacher site ($t_4 = 0.392$, $P = 0.709$) or Kramer Ridge ($t_4 = 0.216$, $P = 0.836$). However, bird communities were significantly different (Bray-Curtis index <0.6) between the years at both sites.

Table 5. Bird community characteristics for two grassland study sites ruveyed in Winona County, Minnesota, 2008 and 2009. Standard deviations are in parentheses.

CHARACTERISTIC	GRASSLAND SITES			
	Kreidermacher		Kramer Ridge	
	2008	2009	2008	2009
Bird species richness	14	19	13	8
Simpson diversity	0.772	0.896	0.799	0.772
Density (birds/ha)	18.2 (10.8)	15.5 (6.8)	11.9 (7.8)	13.0 (4.3)
Bray-Curtis index	0.420		0.335	

Vegetation structure (visual obstruction) (ANOVA $F_{5,54} = 8.30$, $P < 0.001$) and litter depth (ANOVA $F_{5,114} = 24.19$, $P < 0.001$) both differed significantly among the bird survey sites (Table 3). Vegetation structure was greatest at the wettest sites (Burns Valley Creek, SE Tech) and lowest at the driest site (Valley Oaks). There was no significant difference between restored sites and old-field sites with respect to either vegetation structure or litter depth (Fig. 1). Litter depths were significantly thicker at sites with no recent management compared to sites with regular harvest or mowing (Fig. 2).

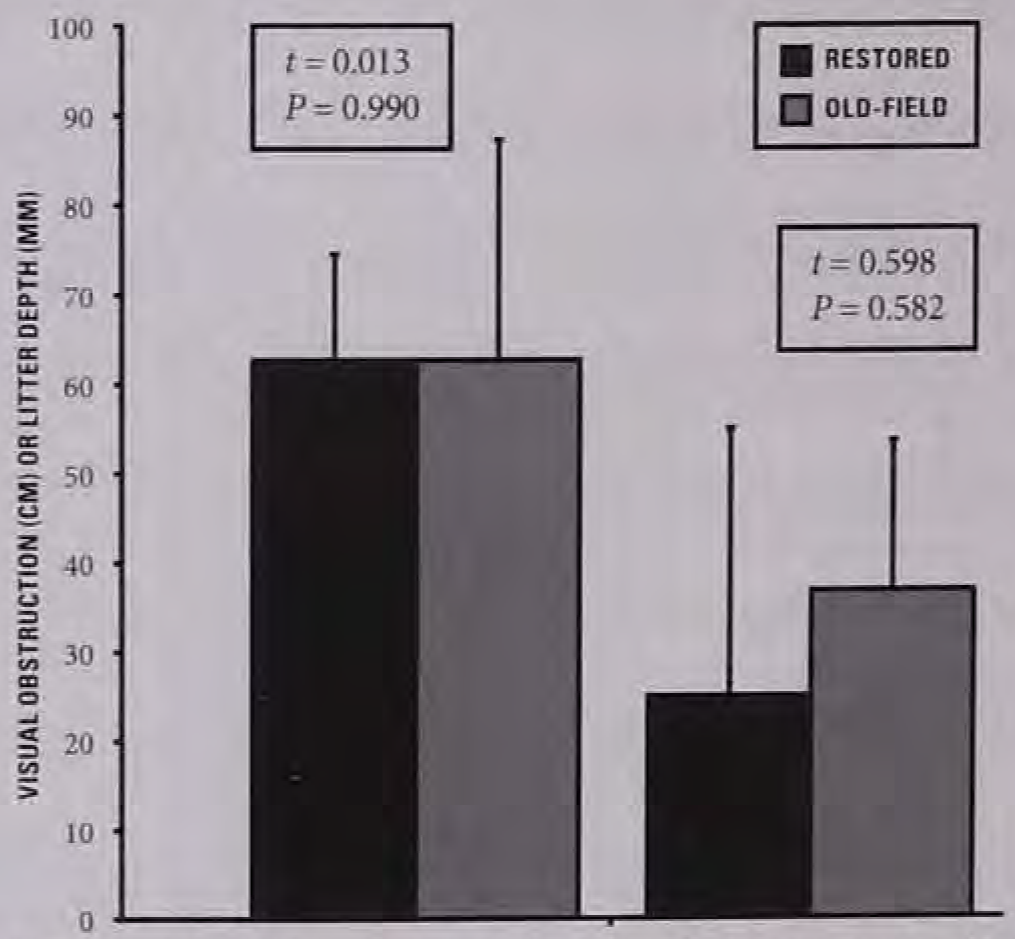
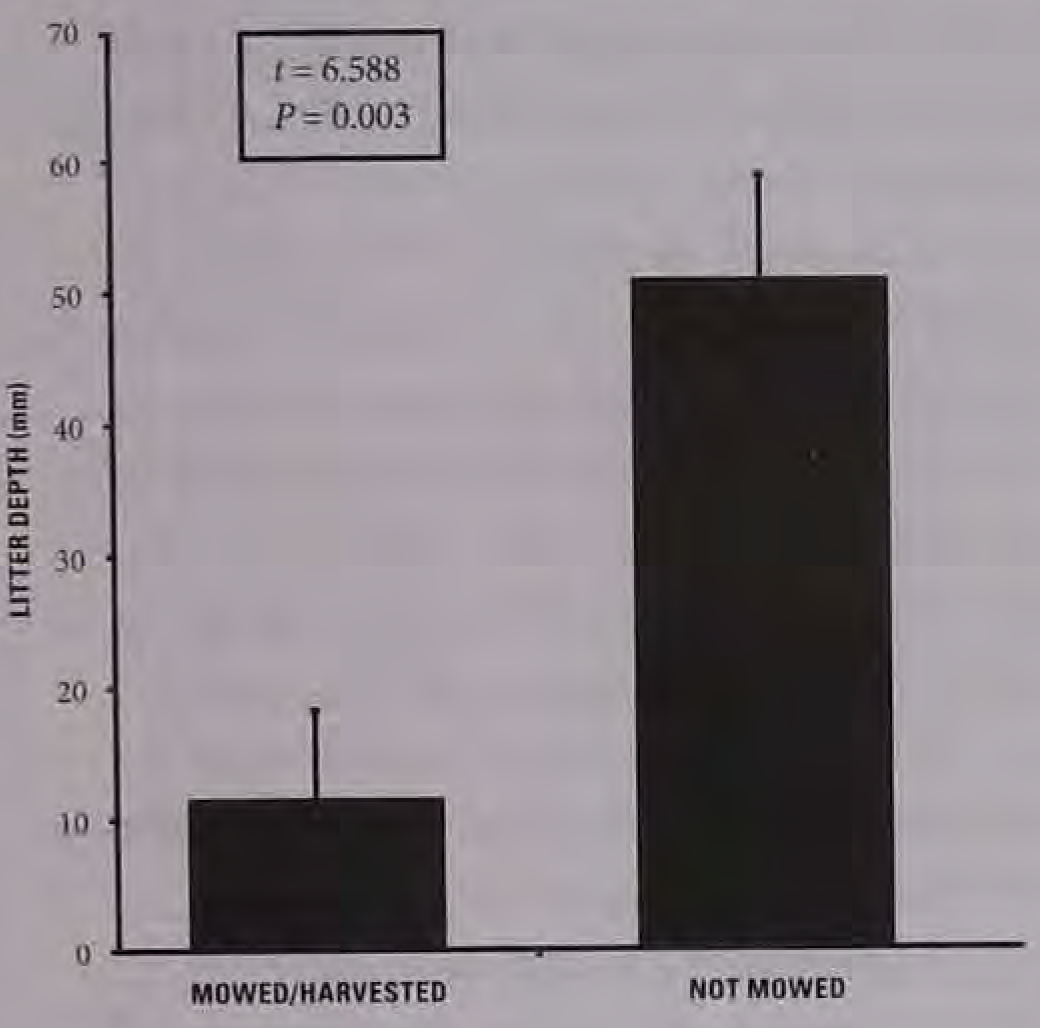


Figure 1. Vegetation structure (cm; visual obstruction reading) and litter depth (mm) at restored grassland and old-field bird survey sites. T-test comparison results are included above bars. Bars and vertical lines represent means and standard deviations, respectively.

Figure 2. Litter depths (mm) at brd survey grassland sites with and without mowing/harvest. T-test comparison results are included above bars. Bars and vertical lines represent means and standard deviations respectively.



DISCUSSION

Although the grassland habitats surveyed in this study varied in size, shape, vegetation composition and structure, degree of isolation, management type and frequency, and surrounding land use, most had similar densities of birds. Total bird densities in grasslands in Winona County also were similar to bird densities reported in grasslands over a wider geographic area (Warner 1994, Fritcher et al. 2004, Winter et al. 2005, Ahlering et al. 2006, Bakker et al. 2006, Peitz 2007, Rahmig et al. 2009, Skagen and Yackel Adams 2010), suggesting that restored grasslands and old-field habitats in southeastern Minnesota were as successful in attracting birds, in general, as are native prairies, managed grasslands on federal lands, and CRP fields elsewhere in Minnesota and in other states. However, the general lack of native grassland birds in the study grasslands suggests that these small grasslands may not be very suitable for grassland species.

In this study, only the three largest sites (3 ha and larger) attracted grassland bird species. Although several species of grassland birds, especially passerines, have small territories, many species require large tracts of land to reproduce successfully and/or sustain local populations (Samson 1980; Dechant et al. 1998; 1999a, b, c, d, e; Herkert 1998; Swanson 1998; Hull 2000). Many regional grassland species that either were not observed during the present study (eastern meadowlark, *Sturnella magna*; Henslow's sparrow, *Ammodramus henslowii*; savannah sparrow, *Passerculus sandwichensis*) or were observed in very low numbers (grasshopper sparrow, western meadowlark) typically occupy grasslands larger (> 10 ha) than those examined in this study, or they are very sensitive to grassland fragmentation and isolation (Dechant et al. 1998, 1999e, Herkert 1998, Swanson 1998, Hull 2000). Small, fragmented, and/or isolated grasslands are unattractive to many grassland birds (Samson 1980, Knopf 1994, Herkert et al. 2003; but see Ahlering et al. 2006), likely because nesting birds often suffer greater reproductive failure due to higher rates of nest predation and brood parasitism in these smaller grassland patches (Warner 1994, Howard et al. 2001, Cunningham 2005, Shochat et al. 2005, Skagen et al. 2005). No attempts were made in the present study to assess nesting attempts or nest success of grassland birds on the study sites, although adult bobolinks and dickcissels were observed feeding fledglings on at least two study sites. Future studies of grassland birds in this region should include larger (10-100 ha) restored grasslands and old-field habitats and some assessment of nesting success.

Vegetation structure and litter are important habitat components for many nesting grassland birds (e.g., Warner 1994, Cunningham 2005, Rahmig et al. 2009, Skagen and Yackel Adams 2010), although diversity in these characteristics (e.g., presence or absence of woody vegetation, thickness or absence of litter layer) is important to support the greatest diversity of grassland birds (Fritcher et al. 2004, Winter et al. 2005, Bakker et al. 2006, Rahmig et al. 2009). Varying management of the study sites, especially annual mowing or biomass harvest, may have had a detrimental effect on grassland birds by reducing the litter layer and/or disrupting nesting. For example, most grassland birds that are common to southeastern Minnesota prefer a well-developed (i.e., thick) litter layer (Dechant et al. 1998, 1999a, b, c, d, e; Herkert 1998; Swanson 1998; Hull 2000). Significantly reduced litter layers were typical of mowed/harvested sites in this study, and this may have reduced grassland bird use of these habitats relative to other areas. However, the site with annual biomass harvest (Borsari and Onwueme 2008) and the second lowest litter depth had the same density of grassland birds (3.96 birds/ha) as the site with the thickest litter, suggesting that other factors (habitat size and/or shape, isolation, vegetation structure, presence of woody vegetation nearby) may be able to compensate for one substandard habitat characteristic.

Both rural and urban grasslands, as well as restored and old-field habitats, provided habitat for small numbers of grassland birds in southeastern Minnesota. Native grassland birds can thrive in some urban grasslands, even while sharing habitats with increasing numbers of naturalized bird species (Engle et al. 1999, Marzluff et al. 2001, Jones and Bock 2002). Restored and old-field sites had similar vegetation structure and litter depths, allowing both types of grasslands to meet some of the basic structural habitat needs of grassland birds. However, the plant communities of restored and old-field habitats differed dramatically, and this may affect food resources (i.e., seeds, insects) needed by grassland birds. In general, it appears that grasslands in southeastern Minnesota, regardless of their management, may attract native grassland birds, as long as sites are large enough to meet minimum area requirements of some of these species.

Bird communities at the study grasslands were dominated by species usually associated with woody vegetation, including shrub, tree, and cavity nesters (Ehrlich et al. 1988). Although shrubs and trees were not present within any of the study grasslands, woody vegetation adjacent to all of the sites likely influenced the use of these grasslands by non-grassland birds. The majority of grassland birds that breed in southeastern Minnesota are either intolerant of woody vegetation within their nesting habitats or experience reduced nesting success when shrubs and trees increase in abundance (Dechant et al. 1998, 1999a, b, c, d, e; Herkert 1998; Swanson 1998; Hull 2000). Regardless of the plant community or management of the grasslands in this study, only 50% of small restored grasslands and old-field habitats in southeastern Minnesota attracted even small numbers of native grassland birds, likely resulting from a combination of small size and proximity of woody vegetation. Protection or restoration of larger tracts of grassland habitat may be necessary to attract and sustain grassland bird populations within this region.

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Appendix 1. Common names (alphabetical order) and scientific names of birds and plants observed during surveys of six grassland sites in Winona County, Minnesota, summers 2008 and 2009.

BIRDS	
COMMON NAME	**SCIENTIFIC NAME**
American Crow	*Corvus brachyrhynchos*
American Goldfinch	*Carduelis tristis*
American Robin	*Turdus migratorius*
Bald Eagle	*Haliaeetus leucocephalus*
Bank Swallow	*Riparia riparia*
Barn Swallow	*Hirundo rustica*
Black-capped Chickadee	*Poecile atricapillus*
Blue Jay	*Cyanocitta cristata*
Bobolink	*Dolichonyx oryzivorus*
Brown-headed Cowbird	*Molothrus ater*
Cedar Waxwing	*Bombycilla cedrorum*
Chipping Sparrow	*Spizella passerina*
Common Grackle	*Quiscalus quiscula*
Common Yellowthroat	*Geothlypis trichas*
Dickcissel	*Spiza americana*
Downy Woodpecker	*Picoides pubescens*
Eastern Bluebird	*Sialia sialis*
European Starling	*Sturnus vulgaris*
Field Sparrow	*Spizella pusilla*
Grasshopper Sparrow	*Ammodramus savannarum*
Gray Catbird	*Dumatella carolinensis*
House Finch	*Carpodacus mexicanus*
House Sparrow	*Passer domesticus*
House Wren	*Troglodytes aedon*
Indigo Bunting	*Passerina cyanea*
Mourning Dove	*Zenaida macroura*
Red-bellied Woodpecker	*Melanerpes carolinus*
Red-tailed Hawk	*Buteo jamaicensis*
Red-winged Blackbird	*Agelaius phoeniceus*
Ring-necked Pheasant	*Phasianus colchicus*
Rose-breasted Grosbeak	*Pheucticus ludovicianus*
Ruby-throated Hummingbird	*Archilochus colubris*
Sandhill Crane	*Grus canadensis*
Song Sparrow	*Melospiza melodia*
Tree Swallow	*Tachycineta bicolor*
Turkey Vulture	*Cathartes aura*
Western Meadowlark	*Sturnella neglecta*
Yellow Warbler	*Dendroica petechia*

PLANTS	
COMMON NAME	**SCIENTIFIC NAME**
Big bluestem	*Andropogon gerardii*
Black mustard	*Brassica nigra*
Black-eyed susan	*Rudbeckia hirta*
Burdock	*Arctium minus*
Canada thistle	*Cirsium arvense*
Canada wild rye	*Elymus canadensis*
Common milkweed	*Asclepias syriaca*
Compass plant	*Silphium laciniatum*
Goldenrod spp.	*Solidago spp.*
Gray-headed coneflower	*Ratibida pinnata*
Indian grass	*Sorghastrum nutans*
Lamb's quarters	*Chenopodium album*
Little bluestem	*Andropogon scoparius*
Common mullein	*Verbascum thapsus*
Oxeye sunflower	*Heliopsis helianthoides*
Partridge pea	*Cassia fasciculata*
Prairie brome	*Bromus kalmii*
Prairie coneflower	*Ratibida columnifera*
Prairie spiderwort	*Tradescantia bracteata*
Purple coneflower	*Echinacea purpurea*
Purple prairie clover	*Petalostemum purpureum*
Queen Anne's lace	*Daucus carota*
Red clover	*Trifolium pratense*
Reed canarygrass	*Phalaris arundinacea*
Rough bedstraw	*Galium asprellum*
Sedge spp.	*Carex spp.*
Side-oats grama	*Bouteloua curtipendula*
Smooth brome	*Bromus inermis*
Virginia wild rye	*Elymus virginicus*
White sweet clover	*Melilotus alba*
White wild indigo	*Baptisia alba*
Wild bergamot	*Monarda fistulosa*
Wild parsnip	*Pastinaca sativa*
Yellow sweet clover	*Melilotus officinalis*

DEVELOPING PREDICTIVE MODELS FOR ENDANGERED NATIVE PRAIRIE PLANT SPECIES: SMALL WHITE LADY'S SLIPPER (*Cypripedium candidum*), WESTERN PRAIRIE FRINGED ORCHID (*Platanthera praeclara*) AND WESTERN SILVERY ASTER (*Symphiotricum sericeum*)

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Abstract: We studied 109 tallgrass prairie sites in southern Manitoba, Canada, in an effort to identify factors accounting for the presence of three endangered native prairie plant species: the small white lady's slipper (*Cypripedium candidum*), western prairie fringed orchid (*Platanthera praeclara*), and western silvery aster (*Symphiotricum sericeum*). The objectives of our study were to identify factors that could be used to predict those species' distributions, and refine those factors into a model useful to land managers. The intent was to make field inventories more efficient in the future, leading to more effective management, conservation, and restoration. Methods included a combination of field investigation and GIS analysis. We discovered no clear indications regarding modeling small white lady's slipper occurrence. However, we made significant strides in developing predictive models for the other two species. Using our model, we found seven confirmed new locations for the western silvery aster. We also identified one potential new area for western prairie fringed orchids. We plan further analysis of project data and additional field work to refine the predictive models.

INTRODUCTION

Tallgrass prairie habitat within the Prairies and Boreal Plains Ecozones in southern Manitoba, Canada, has been greatly reduced by human encroachment. Seven recognized plant Species At Risk (SAR), federally and/or provincially, occur in tallgrass prairie habitats in Manitoba. Three of these are high priority SAR in Manitoba: the western prairie fringed orchid (*Platanthera praeclara*)--endangered; small white lady's slipper (*Cypripedium candidum*)--endangered; and western silvery aster (*Symphiotrichum sericeum*)--threatened. The critical habitats of these three species have not been well defined. Systematic surveys for these species and their remaining habitats have not been completed in Manitoba to date.

Predictive models (PMs) for critical habitat are statistical tools used to estimate the likelihood that a particular species may or may not occur in a given landscape. Increasingly, landowners and conservation agencies are required to determine the presence or absence of SAR on lands they own or are responsible for. The standard practice followed in typical environmental impact assessments is to conduct an arbitrary survey for known plant Species At Risk. These often are expensive and inefficient, especially if large blocks of land are being assessed. PMs hold the promise of increasing efficiency for environmental impact assessments by focusing survey efforts on areas most likely to hold SAR. As well, predictive models are important tools to locate new populations of plant SAR and develop overall strategies to best protect and restore them.

This project aimed to prepare practical, landscape-based predictive models of the critical habitat for three plant SAR in Manitoba: the western prairie fringed-orchid (WPFO), small white lady's slipper (SWLS), and western silvery aster (WSA), and to develop techniques and protocols to apply in the preparation of PMs for other plant species at risk. The known location and habitat information for these species was compiled and evaluated against the known landscape information (surficial geology, soil types, climate, and land use) to develop preliminary predictive models for their critical habitat. Field surveys were conducted to gather information on plant species associated with our target SAR, to identify other geophysical aspects of their habitat and to test our initial predictive models.

Earlier work by Prairie Habitats Inc. (Collicutt and Morgan 2009) has identified a strong correlation between the WSA and a particular surficial geological feature, Glacial Lake Agassiz Beach Ridges (GLBR). Working with a preliminary predictive model, we located three new locations for this species in southern Manitoba during a single day's field work on September 7, 2006. (Figure 1)

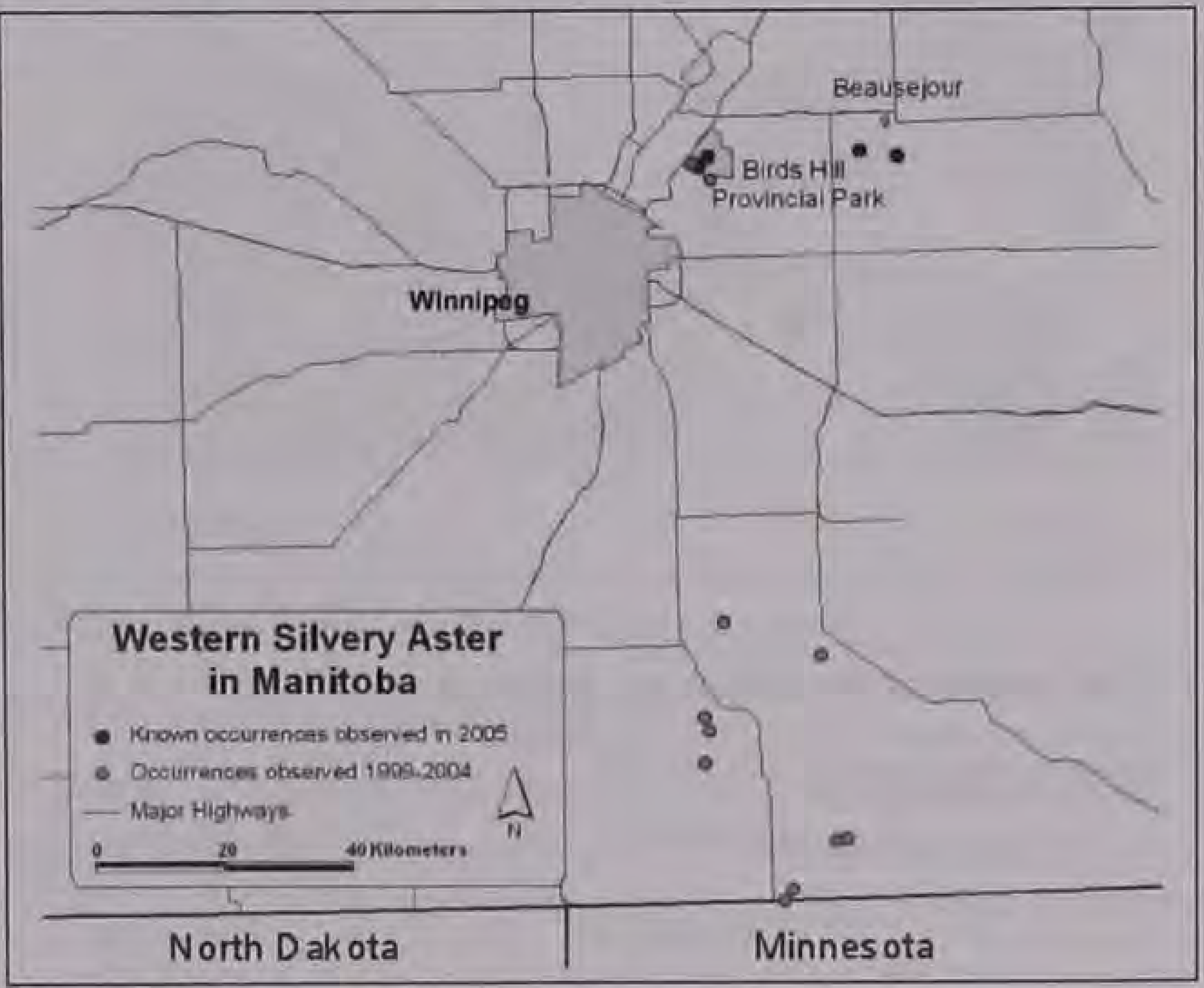

Figure 1. Previous locations of Western Silvery Aster in Manitoba.

Our preliminary examination of the known locations of WPFO in Manitoba, Minnesota and North Dakota (Figure 2) also lead us to believe that their distribution may be linked to a particular geological feature. Since its discovery in Manitoba in 1984, the small region of the province where it occurs (only 48 sq. km) has been surveyed intensively to monitor orchid populations. Little work has been done, however, to determine why the species occurs where it does. Many other areas of apparently similar tallgrass prairie habitat inexplicably do not contain WPFO.

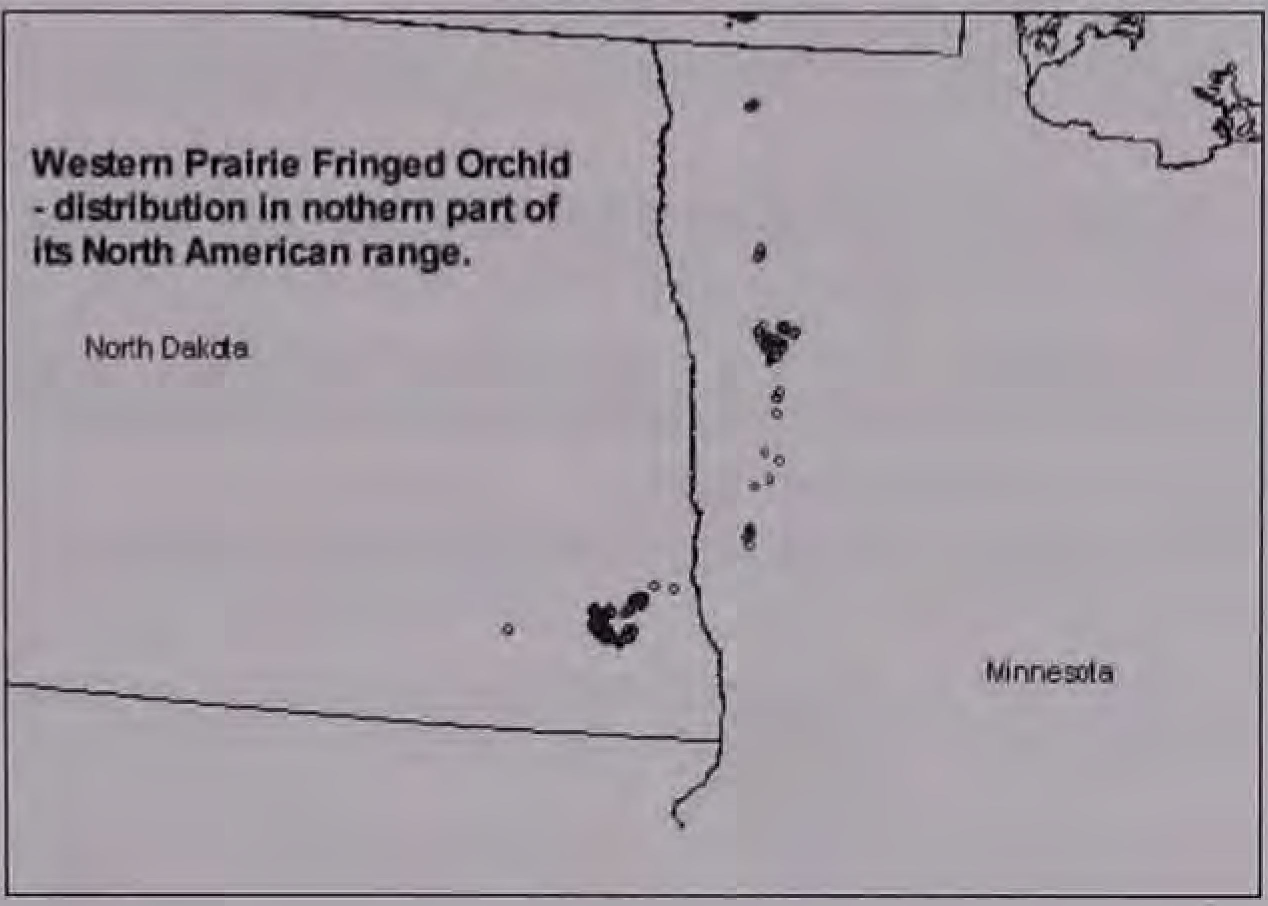

Figure 2. Known western prairie fringed orchid distribution in MB, MN, and & ND.

Research in southern North Dakota has suggested a strong correlation between soil surface moisture and orchid distribution within areas of its occurrence in the Sheyenne National Grasslands (Wolken et al. 2001). Here baltic rush (*Juncus balticus*) and hedge nettle (*Stachys palustris*) percentage of canopy cover were found to correlate positively with WPFO occurrence. Soluble soil magnesium levels and standing water levels in WPFO swales further influenced WPFO distribution in this study. A model with these four factors successfully described 84% of existing study area swales that contained WPFO.

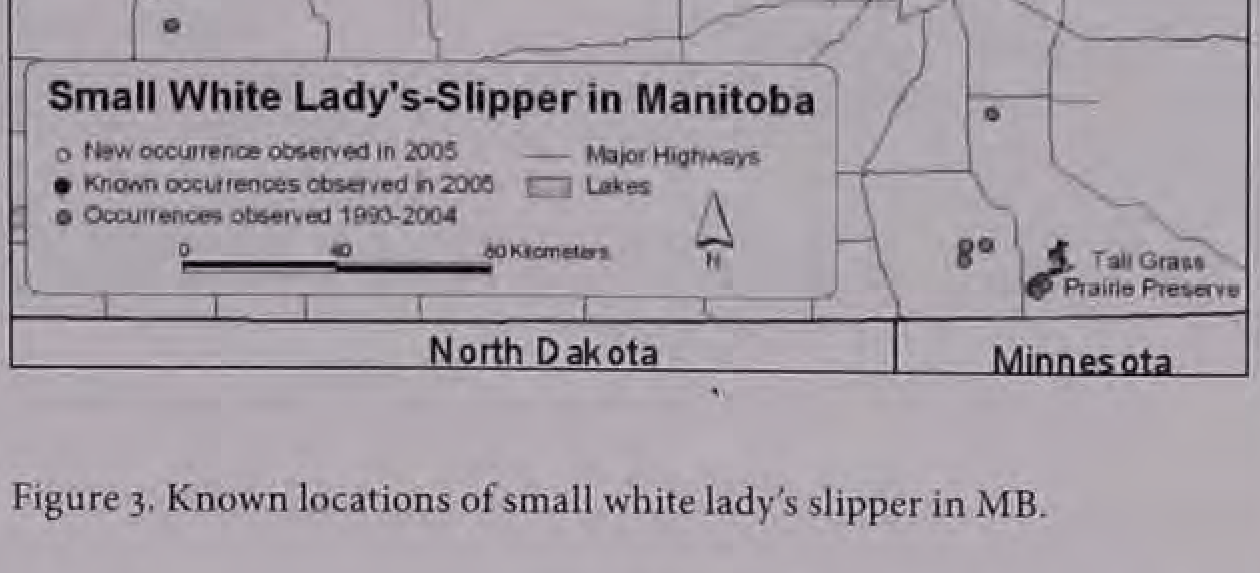

Figure 3. Known locations of small white lady's slipper in MB.

One recent predictive modeling study, stimulated by our research (Garner 2010), already has added new insight into the distribution of SWLS in Manitoba (Figure 3). This study examined soil, associated vegetation, and adjacent land use factors available from maps from Manitoba Conservation (Manitoba Land Initiative 2010). It predicted a higher possibility of finding additional SWLS sites near existing known areas, and quantified the factors that may be responsible for this species' occurrence. As the present paper goes to press (May 2011), ground-truthing has not yet begun to validate the Garner (2010) predictive model. Most other literature on predictive models examined by us to date have dealt with ecosystem modeling, not individual species.

MATERIALS AND METHODS

The most recent information on the known habitat of the three target SAR was obtained from Manitoba Conservation (Foster and Hamel 2006, Foster and Reimer 2007, Foster 2008). From this and our own experiences with these species it was established that all three SAR tended to be found in association with high-quality native prairie communities.

Geo-referenced information was analyzed using an ArcView GIS software-compatible program called MapWindow. Location data for our target species were overlain with landscape-scale geographical information to search for environmental attributes associated with known distributions of these species. Plots of known populations against the underlying environmental attributes served to develop the larger, landscape-based aspect of our preliminary PMs. Field testing of our preliminary PMs was not pursued extensively.

Field sampling was conducted in areas known to harbor target species or in selected high-quality native prairie controls. Specific sampling sites in known areas of target species habitat were selected in two manners. In some cases individual specimens of a target species were selected as center points for sampling radii. When target species were not used as centers for sampling, locations were selected at random within areas of known habitat. At control sites, sampling locations were selected at random. All plant species were identified and recorded within a 1 m radius of the selected point, then again within a 5 m radius. Additional plant species within the selected habitat (up to 50 m away), but outside the formal sampling radii, also were noted. All sampling locations were recorded as GPS waypoints and some were marked with small wooden stakes. Notations were made of any significant factors relating to each sampling site.

Field surveys were conducted mid-May to early June for SWLS, late June to mid-July for WPFO, and late August to mid-September for WSA. Some SWLS and WPFO sites were resampled later in the season to better identify late-season-blooming plants. Some WSA sites were sampled early in the year and again during the flowering period in late summer. WSA was easy to locate before its blooming season, unlike the other two species.

SMALL WHITE LADY'S SLIPPER

Analysis of available geo-referenced information did not reveal any significant trends or explanations to the distribution of SWLS in Manitoba, so we have not yet developed a geographic component of the predictive model for this species. More effort, however, is required to complete this analysis. One PM-based search of 12 native prairies along a proposed pipeline route in southwestern Manitoba was conducted for SWLS in 2007-08.

WESTERN SILVERY ASTER

Analysis of available geo-referenced information revealed a strong association of WSA with glacial lake beach ridges (GLBR, 6b in Table 1) specifically, and with glacial lake sand/gravel deposits (3, 3a, 3b, 6, 6a, 6b in Table 1.) in general.

Table 1. Definitions of surficial geological features associated with WSA (from: Manitoba Mineral Resources 2006).

	PROXIMAL GLACIOFLUVIAL SEDIMENTS:
3	Sand and gravel; 1-20m thick; occur in belts with single or multiple esker ridges, kames, and kettle holes; proximal sediments deposited by meltwater in contact with glacial ice.
3A	Sand and gravel predominantly derived from Precambrian rocks.
3B	Sand and gravel predominantly derived from Paleozoic rocks.
	GLACIAL LAKE SHORELINE SEDIMENTS:
6	Sand and gravel; 1-5m thick; beach ridges, spits, bars, and nearshore sand and gravel >1m thick; sediments reworked by the wave action of glacial Lake Agassiz.
6A	Sandy nearshore sediments.
6B	Sand and gravel beach ridges.

Our preliminary predictive model for WSA consisted of searching for glacial lake beach ridges that still held high quality native prairie communities. On two occasions, September 11 and October 1, 2007, we made attempts to locate new locations for WSA by visiting selected sites, based on the presence of glacial lake beach ridges. Sites which held high-quality native prairie plant communities were inspected closely, while sites that had been disturbed extensively or were agricultural fields were not inspected. Of 12 sites deemed as potential for WSA, four new locations for this species were located in 2007. Three new locations for WSA had been located using essentially the same PM on September 6, 2006 (Figures 4 and 5).

WESTERN PRAIRIE FRINGED ORCHID

Analysis of available geo-referenced information revealed a fairly strong association of WPFO for certain very calcareous soil associations in the region of this species' known distribution. There are additional underlying geological features that seem to support the idea of this region being fairly unique both in soil drainage and chemistry, and underlying geology. The known WPFO range is in a region of extremely calcareous soils overlaying calcareous glacial till which in turn overlays a distinct dolomitic limestone region. Additional areas of very calcareous soils were identified in this general region of SE Manitoba that merit investigation as possible habitat for WPFO. No PM-based surveys for new locations of WPFO were carried out due to time and budget restrictions. Further field work is planned in 2011 to examine one potential WPFO area predicted by the model along the Minnesota border south of Vita, MB. Figure 6 presents the distribution of WPFO in relation to soil, surficial, and deep geological features.

Table 2. Locations of the newly discovered WSA patches (also shown in Fig. 4).

LOCATION	SITE NAME	DATE	UTM LOCATION
Gardenton (east)	001_NEW_WSA	01-OCT-07 10:40:40AM	14 U 673114 5440390
St. Malo (north)	003_NEW_WSA	11-SEP-07 7:19:48PM	14 U 650834 5468915
Woodmore (south)	004_NEW_WSA	01-OCT-07 1:56:38PM	14 U 648920 5441035
St. Malo (north)	005_NEW_WSA	01-OCT-07 7:06:36PM	14 U 652846 5471586
Gardenton (south)	054_NEW_WSA	07-SEP-06 1:21:43PM	14 U 663237 5431519
Gardenton (south)	055_NEW_WSA	07-SEP-06 1:58:10PM	14 U 663463 5432439
St. Malo	058_NEW_WSA	07-SEP-06 6:11:29PM	14 U 649749 5465635

Figure 4. Seven newly discovered WSA locations in MB and their close correlation with glacial lake beach ridges from predictive model.

SPECIES ASSOCIATION RESULTS

A total of 109 different sites were sampled, 24 in WSA habitat, 28 in SWLS habitat, 29 in WPFO habitat, and 28 control sites in similar habitat but without the target species. Forty-one of these were sampled twice to get a better picture of season-long species associations. Data from both samples were amalgamated into a single record.

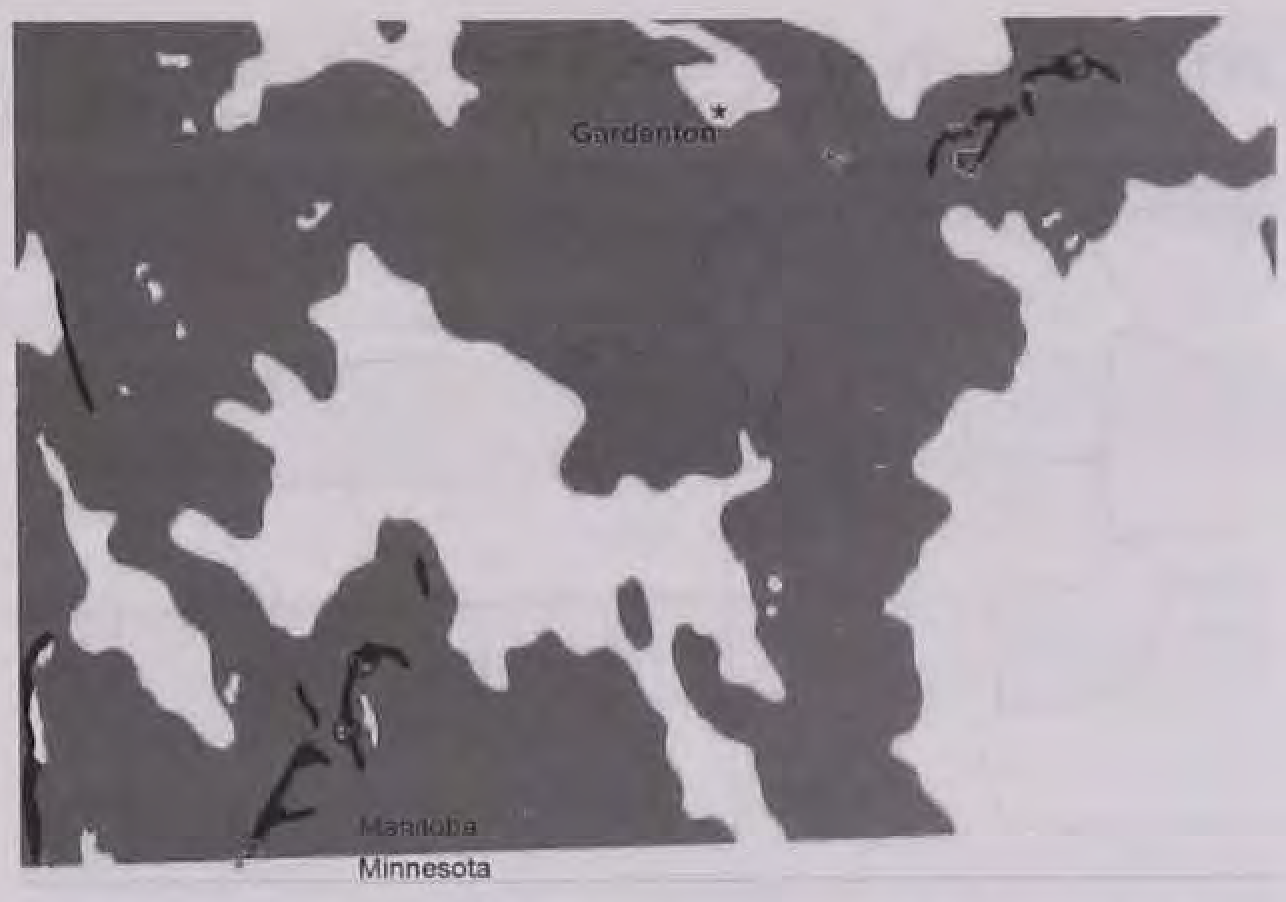

Figure 5. Close-up of three new WSA locations near Gardenton, MB, from predictive model.

Species association data was collected as separate 1 m and 5 m radius plots surrounding the target species or controls. All records for each site sampled, however, were combined. In the end our data consisted of a species list for each of the 109 sites: some of these sites were centered on individuals of our target species; some were in areas known to harbor our target species, but did not contain the target species; and some were in randomly selected high-quality native prairie communities not known to contain the target species (control sites).

Although we conducted a number of analyses on our species association data, the most valuable of these proved to be the simplest. For each of the target species' habitats we determined the frequency of occurrence of all species recorded (number times a species was recorded / total number samples). This produced a list of associated species and their percent frequency of occurrence for each target species. Sorting the data by percent frequency of occurrence gave us a list of species that were closely associated with each targeted SAR.

Upon inspection of these lists it became apparent that a number of species were found commonly in all three target species sampling sites, as well as in the control sites. For example, big bluestem and stiff goldenrod (*Solidago rigida*) had high frequencies of occurrence in all three target species sites and controls. This ranged from 83% to 93% for big bluestem and 63% to 93% for stiff goldenrod. As such they may be considered indicators of high-quality native prairie, but they provide no means to distinguish between habitat types of the target species. They have little value as indicators to distinguish habitat for any one of the target SAR.

A simple extension of this analysis, however, proved very revealing. As a comparative measure we combined all our data from WSA, SWLS, WPFO, and control sites and determined an overall frequency of occurrence for all species

Figure 6. Distribution of WPFO in MB in relation to soil, surficial and deep geological features.

recorded in all. The percentage of frequency occurrence of each species for the combined data was then subtracted from that species' percentage occurrence within each of the target species data. This served to highlight plant species that had a high association with each particular target species, and sorted out species that had high or low associations with all habitat types.

A total of 293 species of plants was recorded in the sampling. Percent occurrence of these in the combined data ranged from 0.9% for several species found only once in all sampling to 87% for big bluestem (*Andropogon gerardii*), the most commonly encountered plant in all our sampling sites.

DISCUSSION

Our analyses of the geophysical features of southern Manitoba as they relate to the distribution of our target SAR are far from complete. Several factors make these analyses daunting. One of the main problems is that the levels of information available for different regions of southern Manitoba are not the same. For example, there is highly detailed surficial geology information available for a region of southeastern Manitoba, but not for the rest of the province. All the known locations of two of our target species, WSA and WPFO, fall within this southeast region, but the third species, SWLS, has populations in western Manitoba where available surficial geology information is much less detailed. An additional factor is the lack of resources and time in the current study to examine fully the large amount of data and do additional field work. An extension of the project's budget was just granted in late 2010. Further analysis is ongoing as this paper goes to press, and more field testing of the models is planned for the 2011 season.

SMALL WHITE LADY'S SLIPPER

SWLSs are generally found only in sites supporting high-quality native prairie plant communities. A number of the known locations of SWLS, however, are roadside rights-of-way, which demonstrates that the species is capable of recolonizing disturbed locations. Why this species does not spread more aggressively, like the closely related yellow lady's slipper, is unknown. At the current state of our analyses we have been unable to find any particular geophysical features that seem strongly associated with the presence of SWLS and that would help to explain the limited distribution of this species. Further insights might be gained through a more thorough analysis of our extensive field data, existing geospatial information, and additional field work.

SWLS once had a wide historical distribution in Canada across southern Manitoba and southeastern Saskatchewan. Current locations, though much reduced due to native prairie habitat loss, still reflect some of the species' adaptability. Of the three species in this study, SWLS was the least predictable. It occupied a range of habitats from high-quality native prairie to pastures and roadsides adjacent (or not) to existing native prairies. It tended to occur with a wide variety of native and some nonnative species, none of which appear to be very good at predicting the presence of SWLS.

Some success was apparent, however, in using the model to locate two potential SWLS sites found late in the 2007 field season. Although they turned out to have the closely related yellow lady's slipper (*Cypripedium calceolus*) in 2008, they did show many similarities, especially in slope position, to some of the existing SWLS sites. It appears that teasing out the factors responsible for SWLS distribution, if possible, will be more complicated than doing so far either of the other two species. Garner (2010) already has made strides in this area in Manitoba, though field-testing the model remains to be done.

Our species association data support the idea that SWLS requires habitats that are high-quality native prairie plant communities. The species found most commonly in association with SWLS are well representative of mesic tallgrass prairie. Further review of our data, however, shows that this list of associated species may have limited value as indicator species as part of a predictive model for SWLS. This species has a much broader range in southern Manitoba than does either WPFO or WSA. Note the overall lower figures for the SWLS corrected associations, as compared to figures found in the WSA (Figure 8) and WPFO (Figure 9). Eight of the top ten associated species for SWLS are below 30%, while all the top ten for the other two species are above 30% (Figure 10). At this time we are unable to formulate a reliable PM for locating critical habitat for the SWLS.

Figure 8. Top 10 corrected WSA associated species.

Figure 9. Top 10 corrected WSA associated species.

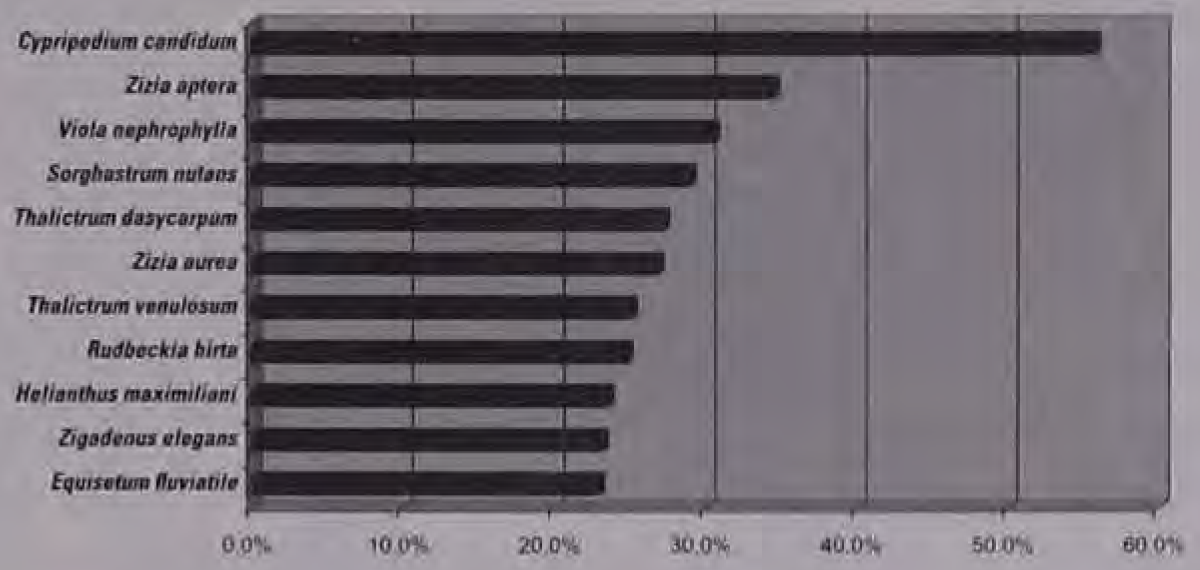

Figure 10. Top 10 corrected WSA associated species.

WESTERN SILVERY ASTER

WSA occurs in areas with sandy, gravelly soils. Our analyses of the available geospatial information show a strong correlation between the distribution of known WSA locations and glacial lake sand and gravel deposits. In particular Glacial Lake Agassiz Beach Ridge subsurface deposits hold the most potential.

The mere presence of glacial lake beach ridge sand and gravel deposits, however, is not necessarily an indicator of potential WSA habitat. Over much of southeastern Manitoba where these deposits are in regions of fairly flat terrain they have developed soils that permitted agricultural development. This has lead to the removal of the original prairie plant communities. Sites where WSA occurs now tend to display significant physical relief that has precluded agricultural use. In our searches for new WSA populations it quickly became apparent that glacial lake beach ridges with low relief were often incorporated into crop fields. All sites where we did find new WSA populations were distinct hills or ridges that had little soil development and appeared to host relatively undisturbed prairie plant communities. They often were associated with sand and aggregate mining operations. Some glacial lake beach ridges also held only forested communities and therefore did not currently represent habitat for WSA.

Species association data for WSA indicate, as expected, that this species is most often found within xeric native prairie plant communities. Species listed in Figure 8 are good indicators for the presence of WSA. Our work has led to an effective predictive model for WSA. Though not formally quantified as yet, this predictive model suggests that landscapes characterized by raised sand/gravel deposits supporting xeric native prairie plant species have a high probability of harboring WSA.

The geographical range of WSA in Manitoba is not well defined. Extant populations occur east of the Red River from just northeast of Winnipeg to the U.S. border (Figure 1). The predictive model developed in this project affords an opportunity to conduct a formal survey to determine the extent of this species' range in Manitoba. Once this is known and all extant populations have been identified, this information could be used to further the endangered species recovery plan for WSA.

WESTERN PRAIRIE FRINGED ORCHID

The restricted geographical range of the WPFO in southeastern Manitoba has been fairly well documented, but the underlying reasons for its limited range are not known. Our analysis of the available geospatial information suggest that the small region known to harbor WPFO may be geologically distinct from surrounding regions and this may be a factor in the restricted range of this species. Species association data for WPFO support the idea that this species is limited to very wet prairie communities. The plant species with the highest corrected association values for WPFO are all indicative of wet native prairie plant communities (Figure 8).

The presence of this species in roadside rights-of-way show it is capable of recolonizing disturbed sites. Yet it shows no signs of spreading from its known range. This may be taken as evidence that some soil, drainage, or other geological factors exist in this region that WPFO is dependent on. There is a close correlation of the WPFO range and regions of extremely calcareous soils, the soils likely the result of underlying geological conditions. Enough evidence exists to suggest that further soil/geological analyses should be undertaken in concert with soil moisture and drainage patterns to further refine a PM for the critical habitat of this species.

For a predictive model for WPFO we suggest that it is found in areas of moist, high-quality native prairie plant communities with extremely calcareous soils. The species listed in Figure 8 should be considered as good potential indicators for the presence of WPFO in Manitoba, and possibly northern Minnesota and North Dakota.

CONCLUSIONS

Quaternary beach ridge deposits correlated nearly 100% with WSA distribution. Seven previously unknown WSA sites for Manitoba and one potential new area for WPFO were located using our preliminary PMs. The success of the model in predicting WSA distribution and helping find possible new WPFO locations has shown the validity of the predictive model approach. It bodes well for further discoveries on these and other species. Additional research into the habitat requirements of SWLS, however, is required to develop a PM for this species. We hope the approach outlined here will lead to more-efficient inventory, better protection, more-informed management of existing populations, and enhanced strategies for maintaining existing native prairie habitats and expanding their current range with suitable reintroductions.

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GENETIC VARIATION IN THE ILLINOIS-THREATENED HILL PRAIRIE LARKSPUR

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Abstract: *Delphinium carolinianum* Walter is a threatened wildflower in the state of Illinois, where it is limited to isolated hill prairies along the Mississippi River. Isolated populations often experience little to no gene flow with other populations, and this may cause inbreeding depression that ultimately leads to extinction. Through use of amplified fragment length polymorphism (AFLP) molecular markers, we assessed the patterns of genetic variation within and between populations of *D. carolinianum* to better understand the stability of extant populations of this rare wildflower. Our data show that high levels of genetic variation are present within the populations studied, yet it is unclear whether this variation is due to current gene flow or preservation of historic variation.

Key Words / Search Terms: AFLP analyses, *Delphinium carolinianum*, rare plants, hill prairies, fragmented populations, population genetics

INTRODUCTION

Delphinium carolinianum (Ranunculaceae) is commonly called wild blue larkspur and grows in the Midwestern, southeastern, and south central United States (USDA, NRCS 2010). The plant is a perennial, 0.6-0.9 meters tall, which flowers in June, then sets seed and goes dormant for the remainder of the growing season. In Illinois as elsewhere, *D. carolinianum* is commonly called wild blue larkspur (Mohlenbrock 2002). However, in this study, we use hill prairie larkspur as its common name because in Illinois it is found mostly on hill prairies and the flowers are more white than blue. The hill prairie larkspur is currently found in three counties along the Mississippi River but was previously reported to be found in six counties across Illinois (Figure 1). The populations within each county are highly isolated, and many lack long-term protection status because they are located on private property. Our demographic data provided the Illinois Endangered Species Protection Board with the information necessary to place *D. carolinianum* on the state-threatened species list (IESPB 2009). *Delphinium carolinianum* is considered endangered in Florida and threatened in Kentucky (USDA, NRCS 2010).

A hill prairie is a dry grassland patch located on the west or southwest slope of an elevation (Evers 1955). Hill prairies are one of the least disturbed prairies left in the United States because the steep slopes where they are located are not suitable for conversion to farmland. Agriculture and development have replaced the surrounding natural areas, causing the habitat of the hill prairie larkspur to become highly fragmented into small "hill prairie islands." The lack of conservation within and around the hill prairies

Figure 1. Map of Illinois with documented current and historical locations of the hill prairie larkspur. Current locations indicated with circle, historical locations indicated with triangle. Historical location data taken from Mohlenbrock and Ladd (1978).

has allowed invasive species to encroach. Some of these species include dogwood (*Cornus drummondii* C.A. Mey.), bush honeysuckle (*Lonicera maackii* (Rupr.) Maxim.), and red cedar (*Juniperus virginiana* L.). Management and preservation must be done on hill prairies to help prevent *D. carolinianum* from becoming extirpated.

Genetic data provide conservationists and land managers with information to aid in preserving biodiversity within fragmented ecosystems. Fragmentation of populations can result in genetic drift, inbreeding, and reduced gene flow (Young et al. 1996). All of these reduce the amount of genetic variation present in a population, and increase the chances of having deleterious alleles become fixed in the population. The restoration of gene flow between fragmented populations can alleviate these effects by introducing new alleles to those populations lacking genetic diversity. Despite these generally observed trends in population response to fragmentation, fragmentation has sometimes been observed to increase gene flow in plant populations through the exchange of alleles between fragment populations (Hamrick 1994, Young et al. 1993).

We undertook this study to answer three questions: (1) What patterns of variation are present in Illinois populations of *D. carolinianum*? (2) How has habitat fragmentation influenced this variation? and (3) How much gene flow occurs between these populations?

MATERIALS AND METHODS

Fieldwork was conducted during the summers of 2005-2010. When a population of hill prairie larkspur was found, a handheld global positioning system (GPS) was used to mark the location. Throughout the month of June demographic data were collected from known hill prairie sites in Henderson, Pike, and Calhoun counties. We conducted a thorough search of the appropriate habitat (loess hill prairie/limestone glade), including around the edges of these communities. We used two or three people and spread out, conducting informal transect lines while walking across these communities, and taking extra care to look more carefully where plants were found or previously found and in areas with similar conditions and associated plants as those areas where larkspurs were found. Population size, number of flowering individuals, life-cycle stage, seedling recruitments, and species associates were recorded for each population. Leaf material was also haphazardly collected and placed in a paper coin envelope (1 leaf per individual, 30 individuals per population, as allowed by population size). The envelopes were placed in plastic sandwich bags containing Drierite (W. A. Hammond Drierite Co., Xenia, Ohio) to dry leaf material for future DNA analyses.

DNA was extracted from dried leaf tissue using a Wizard Kit (Promega, Madison, WI). Voucher specimens for each population will be deposited in the Field Museum of Natural History, Chicago. AFLP molecular markers were used to assess patterns of variation within and between fragmented larkspur populations. Methods for AFLP analysis generally followed Vos et al. (1995) and a modified protocol of the Applied Biosystems Plant Genome kit developed by M. Gitzendanner (U of FL; personal communication), with the following modifications: Genomic DNA, 2 µl of a 1:20 dilution with water, was digested for 3 hrs at 37°C with 0.25 µl of *EcoRI* (12U/µl; Fisher Scientific, Pittsburgh, PA, or New England Biolabs, Beverly, MA), 0.25 µl *MseI* (10U/µl; New England Biolabs), 0.5 µl each of the supplied enzyme buffers, and 1 µl BSA (1 mg/ml) brought to a final reaction volume of 10 µl. The digest reaction was stopped with a 65°C soak for 20 min. A 10 µl ligation reaction containing 0.5 µl of T4 DNA ligase (3U/µl; Fisher Scientific or New England Biolabs), 2 µl 10X T4 Ligase buffer, 1.8 µl *MseI* adapter (50 µM; 5'-GACGATGAGTCCTGAG-3' and 5'-TACTCAGGACTCAT-3'), and 1.8 µl *EcoRI* adapter (5 µM; 5'-CTCGTAGACTGCGTACC-3' and 5'-AATTGGTACGCAGTCTAC-3') was then added to the completed restriction digest and run at 25°C for 3 hrs followed by 10 min at 70°C. These reactions were then diluted with 120 µl of sterile water.

The first selective amplification was conducted in 20 µl reaction volumes containing 4 µl of the diluted restriction-ligation reaction, 0.2 µl Taq DNA polymerase (5U/µl; Sigma Chemical Co., St. Louis, MO), 2 µl 10X PCR buffer (Sigma), 2.8 µl 25mM MgCl2, 1.6 µl 10mM dNTPs, 1.2 µl *EcoRI*+1A primer (5µM; 5'-GACTGCGTACCAATTCA-3'), and 1.2 µl *MseI*+1C primer (5µM; 5'-GACGATGAGTCCTGAGTAAC-3'). Reactions were heated to 72°C for 2 min, then cycled 20 times at 94°C for 30 s, 56°C for 30 s, 72°C for 120 s, and then held at 60°C for 30 min. These reactions were diluted in 250 µl of sterile water and used in the second selective amplification step.

The second amplification was performed in 10 µl reactions containing 2.5 µl diluted +1 PCR product, 0.1 µl Amplitaq Gold DNA polymerase (5U/µl Applied Biosystems, Foster City, CA), 1 µl 10X Amplitaq PCR Buffer (Applied Biosystems), 1.2 µl 25mM MgCl2, 0.8 µl 10mM dNTPs, 0.3 µl of each *EcoRI*+4 primer (1.65µM each; 5'-[6-FAM] GACTGCGTACCAATTCACAT-3'; 5'-[NED] GACTGCGTACCAATTCAAGT-3'; 5'-[VIC] GACTGCGTACCAATTCAACT-3'; 5'-[PET] GACTGCGTACCAATTCACCT-3'), and 0.25 µl *MseI*+4 primer (5µM; 5'-GACGATGAGTCCTGAGTAACTTA-3').

We chose +4 primers because *Delphinium* has a large genome, making a more selective primer necessary to avoid homoplasy (Althoff et al. 2007). The *EcoRI* +4 primers were fluorescently labeled for visualization on an automated DNA analyzer. Single reactions contained four primer combinations because each primer was labeled with a different wavelength of dye (i.e., each *EcoRI* +4 primer with the *MseI* +4 primer). The reactions were held at 94°C for 2 min, then cycled 10 times starting at 94°C for 30 s, 65°C for 30 s, 72°C for 2 min, with a reduction in the annealing temperature by 1°C per cycle. Reactions were then cycled 36 times at 94°C for 30 s, 56°C for 30 s, 72°C for 2 min, followed by a 30-min 60°C hold.

The +4 PCR reactions were sent to the Biotechnology Resource Center, Cornell University, Ithaca, NY, to be run on an Applied BioSystems 3730xl DNA Analyzer. Samples were prepared by mixing 1 µl of the +4 reaction with 0.2 µl LIZ 500 size standard (Applied Biosystems) and 9.8 µl HiDi formamide (Applied Biosystems).

Results were analyzed with GeneMapper (v4.0; Applied Biosystems). The program's default settings were used, except only fragments from 100-500 base pairs (bp) were analyzed, the allele calling threshold was set to 50, common alleles were not deleted, and advanced peak detected was selected.

DATA ANALYSIS

Fragments smaller than 100 bp were removed from the study to avoid fragment-size homoplasy (Althoff et al. 2007). In the entire data set fragments present in more than 95% of individuals were removed from the analysis as well. We used these strict culling measures to assure we removed fixed fragments from the analysis. Other AFLP studies involving plants followed similar culling measures to assure meaningful polymorphism data were studied (Zhan et al. 2009). After culling, the remaining fragments were analyzed with AFLP-SURV (Vekemans 2002) and Structure v.2.2.3 (Pritchard et al. 2000, following analysis design of Hipp and Weber 2008). AFLP-SURV was used to calculate F-statistics between and within counties. Structure was used to look at how the variation of AFLP markers cluster individuals into populations.

RESULTS AND DISCUSSION

Our fieldwork resulted in our finding two additional populations (North New Canton and Kopp's Glade; Table 1). Kopp's Glade in Calhoun County is unique because it is a limestone glade and not a hill prairie. The total number of Illinois populations currently known is nine.

Table 1. Sites sampled and *D. carolinianum* population counts of number of individual plants by year.

COUNTY	POPULATION	OWNERSHIP	2005	2006	2007	2008	2009	2010
Henderson	Bald Bluff	Private			136	60	165	12
Henderson	Tartan Bluff	Private	2	30	128	295	286	29
Henderson	Rt 34	Private	60			53		10
Henderson	Dallas City	Public	30			18		0
Pike	North New Canton	Private					2	0
Pike	Grubb Hollow	INPC/DNR	22	2	38	85		83
Pike	Houson	Private	0	0	50		107	
Pike	Walnut Grove	Private	0	4				
Calhoun	Kopp's Glade	Private				250		27

Population size fluctuated dramatically from year to year (Table 1). This could be due to individuals staying dormant for a season, as has been demonstrated in California (Lewis and Epling 1959). After the culling measures 337 AFLP markers remained from the 1,023 generated. The percentage of polymorphic makers in the hill prairie larkspur in Illinois ranged from 58.2% to 77.2%. Pike County had the lowest percentage of polymorphic markers (58.2%), followed by Calhoun (71.8%), and then Henderson (77.2%). While these levels were higher than those of other *Delphinium* species studied (Koontz et al. 2001), AFLPs generate higher levels of polymorphism than the allozymes used in previous studies on *Delphinium* (e.g., Koontz et al. 2001, Richter et al. 1994). The levels of polymorphism provide baseline data on the genetic diversity of the Illinois populations and can be used in future studies of this species.

The average Fst value across all populations of hill prairie larkspur studied was 0.0606 (range: 0.0285-0.1046). This indicates that genetic variation is partitioned among populations and not within an individual population. Comparisons within counties show higher levels of gene flow (Table 2) than populations in different counties. This signifies that there is or has been gene flow occurring between populations.

Table 2. The average Fst values between counties.

POPULATION	TARTAN BLUFF	DALLAS CITY	GRUBB HOLLOW	HOUSON	KOPP'S GLADE
Bald Bluff	.0480	.0285	.1046	.0718	.0518
Tartan Bluff		.0533	.1027	.0685	.0688
Dallas City			.1023	.0734	.0611
Grubb Hollow				.0354	.0880
Houson					.0493

The Structure results show variable patterns between populations (Figure 2). The clustering of individuals from different counties supports gene flow between counties, but some of the clustering does not follow what would be expected by isolation-by-distance models. For example, clustering of individuals from Calhoun County with individuals from Henderson County and vice versa is nonintuitive, because the counties are separated by a large geographic distance. This brings up the question of how far apart *Delphinium* populations can be and still experience gene flow.

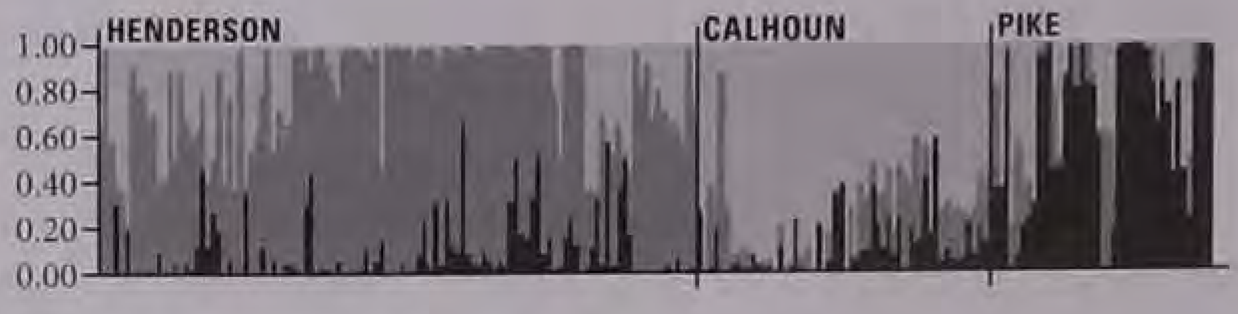

Figure 2. STRUCTURE Clustering with K=3. Each bar represents an individual sampled and shading shows portion of sampled AFLP markers that correspond to a given cluster. Samples generally clustered by their county of origin as shown by the predominance of a particular shade for each county; however some individuals within a county cluster with individuals from other counties.

Hill prairie larkspur can experience gene flow via pollination or seed dispersal. Little is known about seed dispersal in hill prairie larkspur regarding vectors or ranges for these species. However, pollination biology is known. The larkspur is primarily pollinated by bumblebees (*Bombus* spp.). A survey by Greenleaf et al. (2007) of foraging distance of five different species of bumblebees showed foraging distances ranging from 0.1-10 km. The studied bumblebees were of similar size to the native species from Illinois hill prairies. Some populations within counties fall into the 10 km range but most do not. This makes current pollen-mediated gene flow seem to be a highly unlikely source of gene flow. Furthermore, a negative correlation with distance would discern pollen-mediated gene flow but there seems to be no trend between similar genetic variation and distance between sites.

The results of this study suggest two possibilities: (1) Shared variation is the result of current gene flow by an unknown agent, or (2) shared variation is the result of preserved historical variation prior to population fragmentation. In order to conserve this threatened species, we must find out which possibility is the reality.

If current gene flow is the source of shared variation, we would expect these populations to remain genetically stable in the future. Previous studies have shown fragmentation to result in increased variation between populations. Small amounts of gene flow between populations can greatly reduce the loss of alleles due to genetic drift (Young et al. 1996). Bee foraging-distance data indicate that many of the distances are too great for bumblebees to be the source of pollen-mediated gene flow. More data are required to determine if gene flow is current and by which vector this flow is occurring.

If the gene flow is not current and is a result of preserved historic gene flow, we would expect the fragmented populations to succumb eventually to loss of diversity due to allele fixation and a loss of heterozygosity as a result of genetic drift. This could ultimately have a fatal effect on the fragmented populations from an accumulation of deleterious alleles. If this is the case, management must to be applied to the fragmented populations to restore gene flow or maintain the levels of genetic variation.

Previous studies have shown that *Delphinium* may have the ability to maintain sufficient genetic diversity under fragmented populations (Koontz et al. 2001, Richter et al. 1994). It appears that some individuals use different methods of coping with the effects of fragmentation. Lewis and Epling (1959) observed that several Californian *Delphinium* species remain dormant as seeds or rootstocks. This could serve as a means of maintaining genetic diversity by allowing different plants to breed between generations (Lewis and Epling 1959).

Delphinium carolinianum populations in Illinois appear to have high levels of genetic variation. The source of diversity does not seem to be current, at least not via pollen-mediated gene flow. Application of more-sensitive tests and codominant markers to these populations will allow us to determine heterozygosity levels. We may also be able to determine whether this species has mechanisms for preservation of genetic diversity. This information will be valuable for the conservation of native biodiversity for this species as well as for other plant species.

FUTURE STUDIES

While AFLP markers provide excellent data on distribution of variation between populations, their dominant nature (inability to detect heterozygous individuals) makes them problematic for determining gene flow. In populations at equilibrium between mutation and drift, AFLP data can be used to calculate gene flow via the indirect method. The fragmented sites investigated are almost certainly not at equilibrium between drift and gene flow, so conventional population genetic formulae cannot be used to approximate current rate of gene flow (Young et al. 1996, Ouborg et al. 1999). Thus, estimations of gene flow require direct measurements of heterozygosity with codominant (heterozygotes can be distinguished from dominant homozygotes) markers such as allozymes or microsatellites. For example, microsatellites detect more variation and can provide estimates of gene flow over the past 10-100 generations (Selkoe and Toonen 2006).

Microsatellites are currently being developed. We hope these data will allow us to determine heterozygosity and ultimately whether or not gene flow is current or historical. Field observations will also be continued to document new populations and to monitor the highly fluctuating numbers from year to year. Ecological studies are needed to determine what is triggering the variation in population size. Given the range of *Delphinium carolinianum* we would like to expand the sampling outside of Illinois to see whether our Illinois data are normal or the exception for the species as a whole.

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INITIAL SURVEY OF SOIL INVERTEBRATES IN A DISTURBED OKLAHOMA GRASSLAND

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Abstract: Soil invertebrates are a vital part of prairie ecosystems. Because soil invertebrates are small, belong to different taxonomic groups, and are difficult to observe, they have not been widely studied. Also, invertebrate populations vary between different regions, and over the course of a year. Relatively little is known about the effect of prairie-restoration efforts on the diversity or population sizes of soil invertebrates. We are currently in the process of restoring an upland site in southeastern Oklahoma to prairie. In addition to monitoring vegetation changes over time, I began sampling soil invertebrates from the site in 2009. Twenty soil samples from the site were taken in spring, summer, and fall. Invertebrates were extracted using both a Berlese funnel procedure and a flotation procedure. Invertebrates were preserved in 70% ethanol until identification and counting. For analysis, order-level data were used, as not all soil invertebrates could be identified to family. I calculated percent frequency for each order of soil invertebrates, and calculated Shannon index and evenness for each season's sampling. The greatest number of invertebrates was found in summer 2010, the least in spring 2010. Fall 2009 had the highest H' (0.951). However, spring 2010 had greatest evenness (.78). Abundance of different groups differed between sampling periods, with ants, beetles, and earwigs reaching a peak in the summer and collembolans being more abundant in the spring. The survey will continue once restoration treatments (burning) are initiated on the site.

Key-words / Search Terms: soil, soil invertebrates, Oklahoma, disturbed grassland

INTRODUCTION

Soil invertebrates are a vital part of prairie ecosystems. They are involved with nutrient cycling, detritus removal, and interspecific interactions that could affect root growth, root herbivory, and soil chemistry. Herbivores and detritivores in the soil invertebrate community consume two to ten times more of the belowground standing crop than comparable communities of above-ground invertebrates remove of the above-ground standing crop (Tscharntke and Greiler 1995). The recovery of the soil invertebrate community is considered an important measure of the success of a prairie restoration (Brand and Dunn 1998, Kalisz and Powell 2000). However, because soil invertebrates tend to be small, belong to a variety of taxonomic groups, and are difficult to observe directly, their community ecology has not been widely studied. Soil food webs comprise many interactions and high complexity, and have been difficult to describe (Kalisz and Powell 2000); therefore, a single sampling period is insufficient to characterize the full diversity of a community. Community structure can vary with vegetational heterogeneity, soil moisture, and soil temperature (Jonas et al 2002).

There has long been concern (Gibson and Hulbert 1987, Panzer and Schwartz 1998, Nadeau et al. 2006) about the negative effects of fire on prairie insect populations. Fire may directly harm above-ground larvae, pupae, or adults (Gibson and Hulbert 1987) or affect vegetation availability and thus insect population size; these concerns have led to cessation or limitation of burning for prairie management. However, in some cases, animal populations show broad fluctuations without long-term negative effects (Brand 2002). Tscharntke and Greiler (1995) cited several studies (in the U.S. and Europe) that suggested that, in the long term, soil invertebrates tend to be more abundant on grassland sites that are grazed or burned, as opposed to unmanaged grassland area. Lussenhop (1976) found "no impact" of burning on soil arthropods a month after the burning treatment was applied. The current study involved sampling the soil invertebrate community seasonally on land that is in the process of being restored to prairie. The current study provides a baseline of soil invertebrate diversity and abundance for comparison once burning treatments have begun. I hypothesized that diversity and abundance of invertebrates would be highest in the summer samplings.

MATERIALS AND METHODS

SITE DESCRIPTION

The field site is located near Roberta, OK (33° 55' 35" N, 096° 18' 09" W). This site is in an upland area and is underlain by a silty clay loam (Crockett-Durant complex, a Vertic Argiustoll). Prior to 2001, the site was heavily grazed by cattle but has had no grazing on it since. Long time local residents cannot recall the site's ever having been plowed. The entire site is 32.4 hectares; a small subsection of that was designated as an area to study prairie restoration intensively. The study area was divided into 5 rows of 8, 10 x 10 m plots, separated by narrow (1.5 m wide) walkways. Because some native vegetation was still present on the site, existing vegetation was not removed or killed. The site was seeded with a prairie species forb mix Caddo mix, Native American Seeds, Junction, TX) in 2006, but has not yet been burned. Vegetation sampling is ongoing, with prairie-plant species-frequency samples generally collected every spring and every fall. The dominant (in the sense of most frequently occurring) plant species on the site include bushy bluestem (*Andropogon glomeratus*), little bluestem (*Andropogon scoparius*), three-awn grass (*Aristida oligantha*), Scribner's panic grass (*Dichanthelium oliganthes* var. *scribnerianum*), heath aster (*Aster ericoides*), iva (*Iva annua*), and various species of goldenrod (*Solidago* spp.)

SAMPLING PROCEDURES

I began sampling for soil invertebrates in February 2009. Twenty of the 10 x 10 m plots were sampled, with every other plot being sampled to give fairly uniform coverage of the entire area. The same 20 plots were sampled each sampling period. Samples were collected using a standard bulb planter, resulting in cores that were 6.5 cm in diameter by 5 cm deep. A bulb planter was used because it gave shallower but broader samples than a standard soil core sampler would, and the top 5 cm of soil were what was desired for the sample. Five samples were collected per plot, one near each corner and one in the center. These samples were then combined by plot, for a total of 20 samples (one per plot), and were kept cool before being transported to the lab. Samples were collected in February, June, and October 2009, and in March, June, and October 2010 (October 2010 data and analysis not included in this paper).

INVERTEBRATE SAMPLING

Soil invertebrates were extracted using a modification of the Berlese funnel method. Heavy-duty foil two-quart casserole pans were modified to serve as funnels. Most of the bottom of each pan was cut out, and replaced with a 1 mm plastic mesh (plastic needlepoint canvas). A mesh of this size was used as a compromise between restricting movement of larger invertebrates and having large quantities of small soil particles fall into the alcohol preservative. These funnels were then placed on top of 1000 mL beakers with ~20 mL of 70% isopropyl alcohol. The 20 soil samples were each placed in a funnel for 24 hours with a light bulb shining on the soil. At the end of the 24-hour period, the alcohol and any invertebrates in it were transferred to 120 mL plastic specimen cups for storage. However, because of the large volume of soil and because the soil was often wet and tended to clump, I also sorted through each soil sample to search for invertebrates not captured during the extraction. Each soil sample was split into smaller portions, placed in petri dishes, soaked sufficiently with water so that any remaining invertebrates would float to the surface, and examined under 20x magnification using a dissecting microscope. Any invertebrates collected were added to the vial for that sample.

After the extraction and floatation procedures, each sample was examined at 20x magnification using a dissecting microscope. Soil invertebrates were identified as to order and counted. Soil invertebrates were identified using the "Kwik-Key to Soil Dwelling Invertebrates" (Meyer 1994). Initially, I attempted to separate some of the taxa to family, but as some orders were too difficult to separate reliably to family, for the final analysis classification was taken to the order level.

STATISTICAL ANALYSES

The invertebrate data for this study were evaluated using the Shannon-Weiner diversity index (computed using base 10 logarithms) and evenness (as computed [diversity/maximal diversity]*100) for each sampling period. Data were pooled for each sampling period for this analysis. Additionally, relative frequency values were computed for each order of invertebrates for each sampling period (again, data were pooled for sampling period). As these are baseline data that will eventually be compared to posttreatment data, only descriptive statistics were calculated.

To compare treatment periods (spring 2009 vs. spring 2010, and summer 2009 vs. summer 2010), I used a two-sample t-test designed specifically for use with Shannon-Weiner index values (Zar 2010). The variance is calculated differently for diversity-index data than would be with standard data, making some other statistical tests less applicable. Both the standard error and degrees of freedom for this test are calculated differently from a standard t-test: because of low variability and large n values, the degrees of freedom were effectively infinite.

RESULTS

INVERTEBRATE DIVERSITY

In general, the samples showed higher H' values in 2009 than in 2010, with fall 2009 having the highest H' (see Table 1). Diversity varied across seasons, but not greatly. Spring samples for both years showed lower diversity, but the lowest diversity of all samples was seen in summer 2010.

Order richness (number of different orders present in a sample) also varied across the seasons. Summer 2009 showed the highest order richness (21 orders represented), and the spring samples for both years showed the lowest richness (14 orders for spring 2010 and 16 for spring 2009).

Table 1. Order richness, Shannon diversity, and evenness of soil invertebrates. Shannon diversity is computed as -Σ pi (log10 pi), where pi is the proportion of invertebrates in each order class. Evenness is computed using the proportion of Shannon Diversity to maximal Shannon Diversity, which is computed as the base-10 logarithm of the number of orders.

SAMPLING PERIOD	ORDER RICHNESS	SHANNON DIVERSITY H'	MAXIMUM DIVERSITY H'MAX	EVENNESS J
Spring 2009	16	0.843	1.20	70%
Summer 2009	21	0.926	1.32	70%
Fall 2009	18	0.951	1.25	76%
Spring 2010	14	0.889	1.14	78%
Summer 2010	19	0.759	1.27	60%

Evenness was generally similar across samples, with highest evenness (78%) from spring 2010 and lowest (60%) seen in summer 2010. There were few clear trends in any of these numbers over time; differences observed were most likely related to seasonal weather differences.

I performed two-tailed, two-sample t-tests to compare diversity-index data for the two spring sampling periods and the two summer sampling periods. The H' values were not statistically significantly different between spring 2009 and spring 2010; however, they were significantly different between summer 2009 and summer 2010 (t = 4.63, d.f. = ∞, p<.001)

INVERTEBRATE ABUNDANCE

The largest number of individual invertebrate individuals was found in the summers of both years (454 in summer 2010 and 424 in summer 2009; see Table 2). The smallest number of invertebrate individuals, 172, was collected from the sample for spring 2010.

Generally, the same taxa had high abundance across all samples (Table 2). Mites (including orobatid, gamasid, and spider mites) showed high abundance in all samples. Springtails also showed high abundance in most samples. The abundance of earthworms and ants fluctuated over the course of the year; earthworms were absent from the summer 2010 samples, which were taken during a very dry

Table 2. Abundance and frequency values for soil invertebrates. Top five taxonomic groups by relative frequency (number individuals in taxon/total number of individuals found).

SAMPLING PERIOD	ABUNDANCE	RELATIVE FREQUENCY VALUES (%) FOR TOP FIVE ABUNDANT TAXONOMIC GROUPS				
Spring 2009	233	Acarina, 30.9%	Protura, 29.1%	Collembola, 9.9%	Nematoda, 8.2%	Diplura, 7.7%
Summer 2009	424	Diplura, 30.6%	Hymenoptera, 17.9%	Acarina, 13.2%	Nematoda, 11.1%	Collembola, 6.6%
Fall 2009	208	Protura, 22.6%	Acarina, 20.7%	Hymenoptera, 14.9%	Oligochaeta, 14.9%	Collembola, 4.8%
Spring 2010	172	Protura, 28.5%	Oligochaeta, 20.9%	Acarina, 12.7%	Collembola, 11.0%	Coleoptera, 9.9%
Summer 2010	454	Acarina, 46.9%	Collembola, 21.1%	Hymenoptera, 10.6%	Coleoptera, 6.4%	Arenae, 3.5%

period. Hymenopterans (ants) were abundant in the summers, but their numbers were low in the spring of both years. Other groups were in lower abundance but fluctuated over the course of the year. Homopterans and hemipterans showed up in the summer samples in small numbers, but were not found in spring samples. Earwigs (Dermaptera) were found only in the summer 2009 samples. Spiders (Arenae) were most abundant in the summer samples. Proturans were also abundant in most samples, although it is possible that some of the smaller collembolans were confused with proturans. Also, the high abundance of diplurans in summer 2009 may reflect their being confused with proturans.

There were some seasonal patterns. In spring 2009, mites were the most abundant group. In summer 2009, diplurans were the most abundant group. Ants, mites, and springtails also had high abundance. In fall 2009, proturans, mites (Acarina), earthworms (Oligochaeta), and ants (Hymenoptera) were the most abundant groups, and in spring 2010 proturans, earthworms, and mites had highest abundance. In summer 2010, earthworms were completely absent from all samples, and mites, springtails, and ants had the highest abundance. Ant numbers tended to be patchy and variable, with samples from some plots yielding 50 or more ants. Part of the reason the number of soil invertebrates was so high in the summers was the high abundance of ants (Table 2).

DISCUSSION

The trends seen in abundance and diversity of soil invertebrates at this site can most likely be related to seasonal patterns and rainfall patterns. Jonas et al. (2002) noted that June samplings of above-ground invertebrates had the highest species richness, abundance, and diversity. They note that if only one sample can be taken annually for monitoring purposes, it should be a June sample, because that typically the time of highest above-ground invertebrate abundance and activity. For both years in this study, the June sample had the highest number of individual soil invertebrates, as well as highest order richness. While order richness (absolute number of orders found) was high, order diversity (as measured by the Shannon-Weiner index) was significantly lower for summer 2010 than for summer 2009.

The decline in summer 2010 invertebrate diversity is most likely related to weather patterns. Weather conditions near the time of sampling in both spring and summer 2010 were dry and there had been no precipitation for several weeks (NOAA 2010). The summer 2009 sample was collected a few days after a rainfall, whereas the summer 2010 sample was collected after several weeks without rain. However, despite differences in weather (and in diversity; the summer 2009 sample showed moderately high diversity and the summer 2010 sample had the lowest H' of all samples examined, and was statistically significantly lower than the summer 2009 sample), summer seems to be the time of highest soil invertebrate abundance. Lussenhop (1976) states that soil temperature seems to affect soil arthropod density to a greater degree than soil moisture, with high temperatures leading to fewer arthropods being found in samples. However, he sampled only arthropods; the current sample also included annelids, nematodes, and other invertebrates that might also be affected by low soil moisture. There were no segmented worms found in the summer 2010 sample (they were abundant in many other samples), suggesting that the prolonged dry weather had caused them to move into lower layers of the soil horizon. Brand and Dunn (1998) noted that native prairie had higher soil moisture (as compared to sites in the process of being restored) and that this could be a factor in observed differences in soil invertebrate abundance. However, the two samples in my study collected during wetter times (fall 2009 and spring 2010) had lower overall soil invertebrate abundance than the sample collected during the driest time (summer 2010).

Other researchers have found patterns of abundance in remnant sites similar to what I found in a site currently undergoing restoration. Hamilton and Stathakis (1987), in a survey of Illinois remnant prairies, found that ants were the most abundant group, with mites and springtails also showing high abundance. Kalisz and Powell (2000) found 27 different orders of soil macrofauna in samples from Kentucky. They assessed the abundance on a dry-weight basis, with beetles (Coleoptera), earthworms, and ants being among the groups having highest biomass. (However, as these are also among the larger soil invertebrates, that may not be directly comparable with abundance values based on counts). Brand (2002) found that mites and springtails were the first and second most abundant groups in samples from Illinois. Also, Risser et al (1981) noted that in northern Oklahoma prairie, the abundant taxonomic groups included Acarina, Formicidae, Collembola, Diplura, Thysanoptera, Protura, and Symphyla. All these groups were found in my samples; many of them were among the most abundant taxa. In general, my results for order richness and seasonal patterns were similar to those found by other researchers working in prairie.

FUTURE RESEARCH

I plan to continue the study as prairie restoration proceeds on this site. In addition to the invertebrate sampling, the prairie vegetation sampling (not discussed in this paper) will continue. There are plans at this point to burn half the site, perhaps in spring 2011, however, it may be difficult to block the burning from the other half. Monitoring of the soil invertebrate community will continue as restoration activity progresses, particularly after fire is applied as a treatment, to determine what, if any, short-term and long-term effects burning has on soil invertebrates.

The vegetation of the site has not changed appreciably over the year and a half of invertebrate sampling (Corbett, unpublished data). Brand and Dunn (1998) note that vegetation heterogeneity tends to increase diversity of invertebrates. Presumably, if burning leads to changes in the vegetation structure, diversity might change as a result. Tscharntke and Greiler (1995) stated that soil invertebrates tend to be more abundant on sites that are grazed or mowed, as compared to sites that are not. (They mainly cited research from Europe, where burning is a less common maintenance tool than in North America). Lussenhop (1976), after applying a burn treatment to Wisconsin prairie, found a greater number of herbivore or detritivore arthropods in the burned area than in the unburned control.

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RECENT REDISCOVERY OF RARE PLANTS OF EPHEMERAL RAINWATER POOLS IN SIOUX QUARTZITE PRAIRIES IN SOUTHWESTERN MINNESOTA

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Abstract: Sioux quartzite bedrock exposures are a distinctive feature of prairies and pastures in parts of the Prairie Coteau of southwest Minnesota, southeast South Dakota, and northwest Iowa. In April to mid-June, undisturbed bedrock exposures form short-lived rainwater pools ranging from 0.25 m^2 to 10 m^2 in size that harbor a distinct aquatic plant flora that does not occur in more-permanent wetlands. Recent surveys in Minnesota documented several locations of these species, including *Crassula aquatica, Marsilea vestita, Isoetes melanopoda, Heteranthera limosa, Bacopa rotundifolia, Eleocharis wolfii, Limosella aquatica, Plantago elongata, Elatine triandra,* and *Callitriche heterophylla. Crassula aquatica* had not been collected in the region since the early 20th century. Several other species had three or fewer collections in the state. These small-statured and mostly ephemeral species may occur in bedrock pools elsewhere in the Coteau but are unlikely to be located without targeted searches. Management of prairies containing bedrock exposures should take into account bedrock pool microsites in addition to the prairie matrix in which they occur.

Key Words / Search Terms: Prairie Coteau, vernal pools, *Crassula aquatica, Marsilea vestita, Isoetes melanopoda, Heteranthera limosa, Bacopa rotundifolia, Eleocharis wolfii, Limosella aquatica, Plantago elongata, Elatine triandra, Callitriche heterophylla.*

INTRODUCTION

The Prairie Coteau is an area of regionally high land in southwest Minnesota, southeast South Dakota, and northwest Iowa that sits between the paths followed by the James and Des Moines Lobes during the most recent glaciation (Figure 1). West of the Bemis Moraine, the high ridge deposited along the western margin of the Des Moines Lobe, is a region that geologists estimate has not been covered by a glacier for 400,000 to 600,000 years (Bierman et al. 1999). The resulting long period of surface erosion of this region exposed ridges and flat expanses of Sioux quartzite bedrock, which at the time of European settlement were embedded within a matrix of native prairies. The Minnesota Geological Survey mapped 23,800 acres of Sioux quartzite near the surface on the Prairie Coteau in Minnesota, all in Rock and Pipestone counties (Patterson 1995). This map unit consists of shallow soil over bedrock dotted with numerous bedrock exposures. Depressions in bedrock surfaces, ranging up to 10 m^2 in size, accumulate snowmelt and rainwater to form numerous temporary pools that are too short-lived to support permanent wetland vegetation. Some of these pools, however, are deep enough to last for a month or two and support an unusual aquatic flora that includes species rarely recorded in other microhabitats in the region. Several of these species, or closely related species in the same genus, occur in ephemeral pool systems elsewhere in the U.S., including the vernal pools of California's Central valley, as well as in other regions of the world, such as Australia and the Andes (Keeley et al. 1996).

In Minnesota, few collections of several rare ephemeral pool species had been made before a recent, more systematic survey for these species by botanists of the Minnesota County Biological Survey (MCBS), a project of the Minnesota Department of Natural Resources (MNDNR).

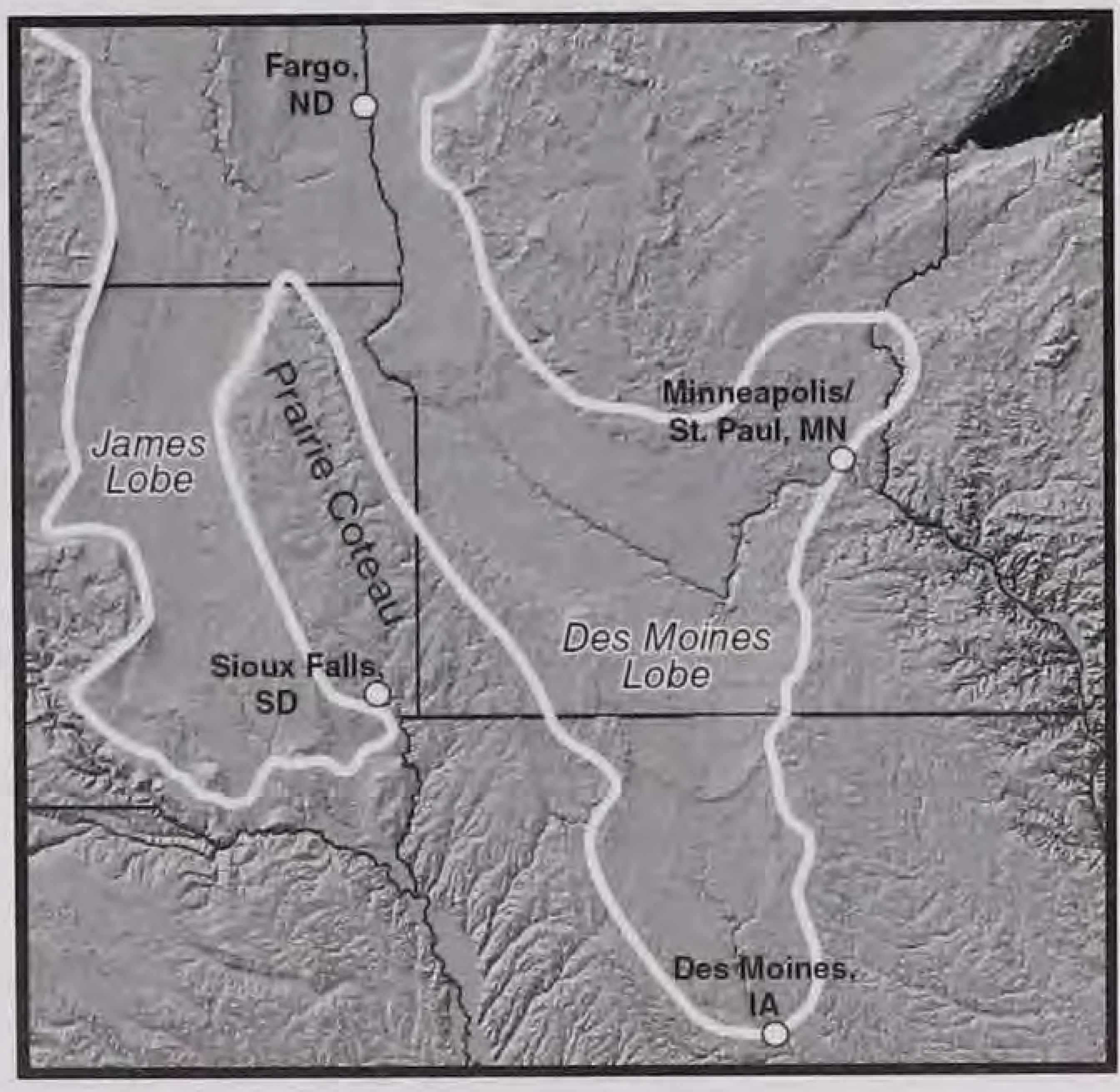

Figure 1. Location of the Prairie Coteau and the paths of the James and Des Moines Lobes of the late Wisconsinan glaciation.

METHODS

MCBS has surveyed native plant communities and rare species remaining in Minnesota on a county by county basis beginning in 1987. Survey sites are identified from aerial photography, as well as through other information including geological maps, past plant collections, and soil surveys. Where landowner permission was obtained, sites with rock outcrops in Prairie Coteau counties were surveyed in the field from 2006 to 2008. Field visits were conducted in May through June when ephemeral rock pools were most likely to contain actively growing plants. Plants were identified following the nomenclature of *Flora of North America* (FNA Ed. Comm. 1993+), and of

Gleason and Cronquist (1991) for taxa not yet treated in FNA. Voucher specimens were collected to document new locations of rare plants and deposited in the University of Minnesota Herbarium of the Bell Museum of Natural History. Native plant communities, including rock outcrops, were classified following the *Field Guide to the Native Plant Communities of Minnesota* (MNDNR 2005) and evaluated for ecological quality and condition. Digitized boundaries of intact rock outcrop plant communities and locations of rare species occurrences were entered into the Minnesota Natural Heritage Information System, maintained by the Minnesota Department of Natural Resources. GIS polygons of native plant communities are available online from the MNDNR's Data Deli (www.deli.dnr.state.mn.us).

RESULTS

Most bedrock exposures occur within a matrix of other native plant communities in the Sioux quartzite regions of the Prairie Coteau. On deeper soils over bedrock, mesic prairie dominated by *Andropogon gerardii* and *Sorghastrum nutans* still remains in a few sites where grazing has not been too severe, or in nature preserves where the prairie is being managed. Depressions within these prairies support permanent wetlands that typically include *Spartina pectinata, Scirpus pallidus, Leersia oryzoides, Persicaria punctata, Persicaria amphibia, Eleocharis compressa, Juncus interior, and Alisma triviale* on less-disturbed areas; or *Phalaris arundinacea* on disturbed sites. *Carex stricta* is common in pools receiving groundwater input. Also, dry prairie is common in shallow soils adjacent to bedrock exposures and is typically dominated by *Schizachyrium scoparium, Bouteloua gracilis,* and *Bouteloua curtipendula.* Most of the rock outcrop sites on private land are grazed by cattle, with some sites so severely trampled that very little of the native flora survives. Broadcast spraying of herbicide to control invasive weeds in pastures is also a common practice that severely affects floristic diversity.

MCBS mapped a total of 814 acres (272 polygons, averaging 3 acres per polygon) of intact (native-dominated) rock outcrop plant communities on exposed Sioux quartzite bedrock surfaces on the Prairie Coteau west of the Bemis Moraine in Pipestone and Rock counties (Figure 2). Approximately half of these mapped areas consist of primary communities on dry microhabitats of bare rock surfaces, rock crevices, and the margins of rock exposures, typically dominated by lichens, bryophytes, and *Selaginella rupestris.* These areas include a variety of typical drought-tolerant species such as *Phemeranthus* (formerly *Talinum*) *parviflorus, Buchloe dactyloides, Opuntia fragilis,* and occasionally the rare *Opuntia macrorhiza.*

Though they are numerous, ephemeral pools on bedrock outcrops on the Prairie Coteau cover approximately half of the mapped rock outcrop vegetation and were not mapped separately. Most ephemeral pools are too shallow and short-lived to sustain plant life. Those pools deep enough to support aquatic or semiaquatic plants typically contain a variety of plant species, mostly annuals, growing in standing water, on pool margins, or in moist sediments after pool evaporation. In addition to the rare species described below, common ephemeral pool associates include *Myosotis verna, Myosurus minimus, Callitriche palustris, Gratiola neglecta, Lindernia dubia, Veronica peregrina, Triodanis perfoliata, Plagiobothrys scouleri, Alopecurus carolinianus, Eleocharis acicularis,* and *Eleocharis ovata.*

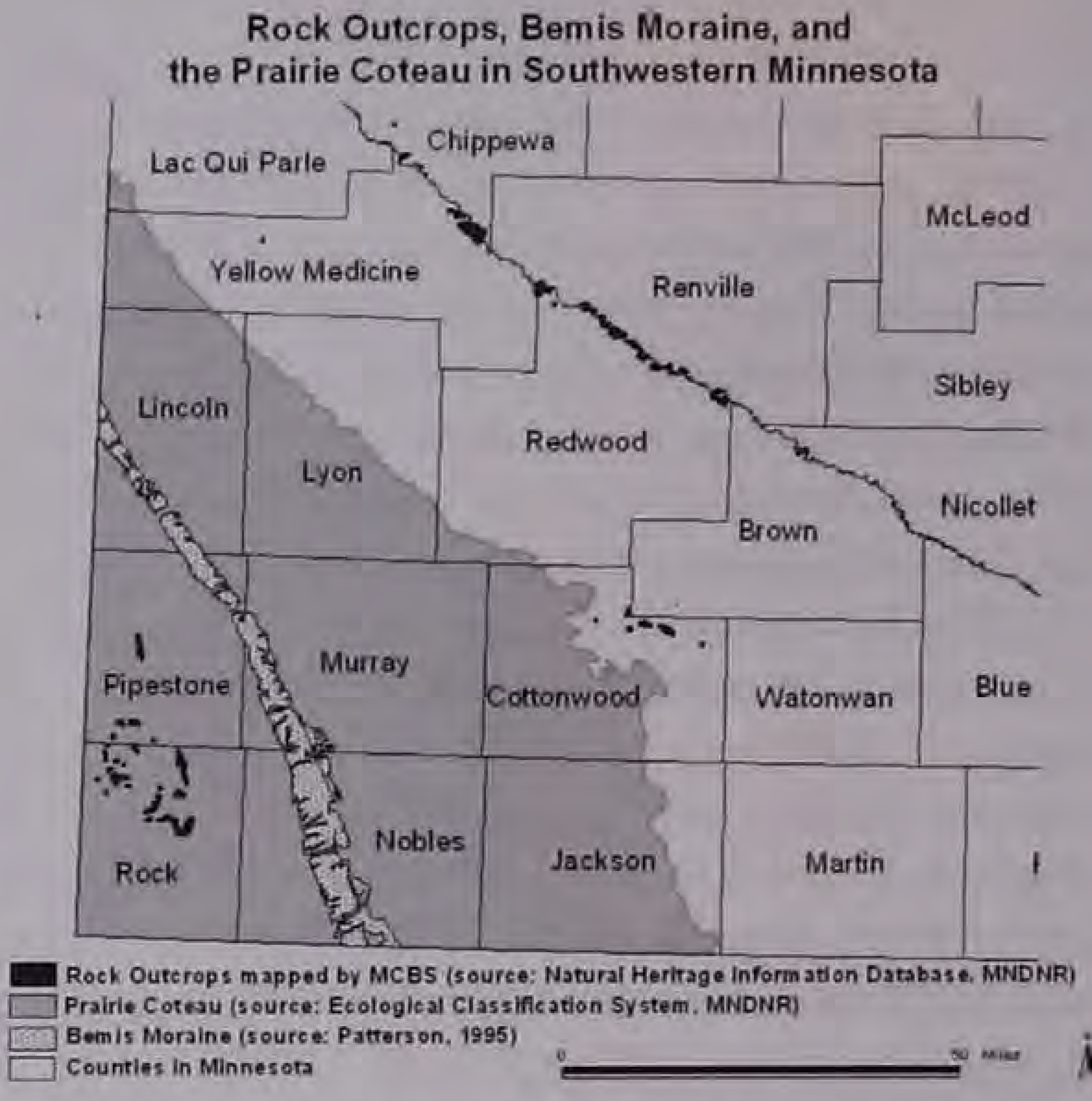

Figure 2. Locations of rock outcrops, the Bemis Moraine, Minnesota counties, and the Prairie Coteau in Southwestern Minnesota.

Once the pool sediments dry out, typically in early June, most of the plant species associated with the seasonal wetland disappear. For much of the rest of the year, all that remains of the former pools are baked, cracked accumulations of bare silt in rock depressions, which conceal the seeds and perennating parts of aquatic or semiaquatic plants. Scattered plants growing on drying pool sediments in midsummer include *Cyperus squarrosus* and *Ambrosia psilostachya,* the nonnative species *Portulaca oleracea* and *Plantago patagonica,* and the rare *Cyperus acuminatus.* Pools with active aquatic plants were also documented in September in 2006, but we did not know whether the plants had been sustained throughout the season or regenerated with the return of rainfall in the autumn.

Observations on several new occurrences of rare plant species recently documented in bedrock pools on Minnesota's portion of the Prairie Coteau are recorded in more detail below. Additional information and photos of these species can be accessed in the MNDNR's Rare Species Guide on the MNDNR website (www.dnr.state.mn.us/rsg/).

PYGMYWEED (*Crassula aquatica*)

Prior to the MCBS survey, *Crassula aquatica* (Crassulaceae) had been collected once in southwest Minnesota in 1945 (in a location that has since been destroyed), and more recently from the margin of Namakan Lake in Voyageurs National Park in far northern Minnesota (MNDNR 2011a). MCBS documented four new locations for this aquatic succulent plant species in Rock and Pipestone counties. In bedrock pools, individual *Crassula aquatica* plants reach a maximum height of two centimeters at maturity and are very easily overlooked. While pools are hydrated, these plants are often buried in soft sediments on the bottom of pools and scarcely visible. When stranded in mud after pools evaporate, these plants grow upright for a couple of weeks, then flower and set seed. In the vernal pools of California, this species was found to perform the Crassulacean acid metabolism (CAM) mode of photosynthesis while it is a submerged aquatic plant (as do species of Isoetes) (Keeley 1999). CAM is the typical mechanism for carbon uptake in desert succulent plants, in which plants take up and store carbon overnight and thereby keep the stomata closed and minimize water loss via transpiration during the day. In ephemeral pools, dissolved carbon can be scarce during the day, due in part to uptake by aquatic plants. By taking up carbon at night, submerged *Crassula aquatica* plants may avoid competition for dissolved carbon during the day. Interestingly, the plants were found to revert to C_3 photosynthesis when they became aerial plants stranded in mud after pools evaporated (Keeley 1999).

HAIRY WATER CLOVER (*Marsilea vestita*)

Marsilea vestita (Marsileaceae) was previously known from three locations in the state, two of which are historical collections that no longer exist (MNDNR 2011a). MCBS documented fifteen new locations of this species in Rock and Pipestone counties. When pools are hydrated, this heterosporous fern forms floating leaves in the shape of four-leaved clovers. After pools evaporate, the plants sprawl on the drying sediments with thicker, aerial leaves that last for a few weeks. The plants produce hard, highly drought-resistant sporocarps that have been found to be viable for many decades (Johnson 1985).

PRAIRIE QUILLWORT (*Isoetes melanopoda*)

Isoetes melanopoda (Isoetaceae) was previously known from two recent and one historical location in Minnesota (MNDNR 2011a). MCBS documented ten new locations, all in Rock and Pipestone counties. This aquatic, heterosporous fern ally roots in shallow silt in the bottom of bedrock pools, with grasslike leaves typically emerging above the pool surface. The plants mature, develop characteristic blackened bases, and persist for several weeks in drying pool sediments. Unlike many other ephemeral pool species, Isoetes has a perennial rootstock that survives the severe desiccation of the pool sediments. The southwestern Minnesota population appears to be disjunct, 250 km from the nearest population (MNDNR 2011a), though this species was collected from a single location in 1924 in southeastern South Dakota (South Dakota Department of Game, Fish, and Parks 2009).

WOLF'S SPIKE-RUSH (*Eleocharis wolfii*)

Eleocharis wolfii (Cyperaceae) had previously been collected in just three locations in the state, all prior to 1968 in moist areas bordering prairie wetlands where they have not been relocated (MNDNR 2011a). MCBS discovered this species growing on the margins of bedrock pools and documented five new locations, three of which are on the margins of Sioux quartzite pools in Rock and Pipestone counties. *Eleocharis wolfii* frequently grows among other *Eleocharis* species, namely *E. compressa* and *E. acicularis*; it may be distinguished from the similar *E. acicularis* by its taller, flattened, twisted culms typically 1 mm or more wide. *Eleocharis wolfii* is listed as rare in many states where it occurs, though recent field surveys suggest it is overlooked and not as rare as previously thought (McKenzie et al. 2009).

MUD PLANTAIN (*Heteranthera limosa*)

Heteranthera limosa (Pontederiaceae) was previously known from only two locations in Minnesota (MNDNR 2011a), though it is common or weedy in other parts of the U.S. MCBS documented three new populations of this species in Rock County and one outside of the Coteau on a rock outcrop along the Minnesota River in Lac Qui Parle County. In Minnesota, small germinating seedlings of this species appear in hydrated pools in mid-May but this larger-statured species takes longer to develop to fully mature plants than do other, much smaller, outcrop pool species. In 2007 and 2008, years with rapid dry-up of pools in early June, most observed seedlings of this species died out in drying mud before maturing and flowering. The only flowering plants that were recorded in 2007 and 2008 were in deeper rock pools in drainages on the sides of bedrock ridges that are fed by runoff or small springs, and persist longer into the summer than the typical pools on flat bedrock depressions.

WATER HYSSOP (*Bacopa rotundifolia*)

Bacopa rotundifolia (Scrophulariaceae) was previously known from seven locations in Minnesota (MNDNR 2011a), all but a couple of which are in bedrock pools, though it is common in other parts of the U.S. MCBS documented fourteen new locations of this species along the Minnesota River or on the Prairie Coteau. This annual species rapidly develops and completes flowering and seed formation while pools are hydrated. In several locations, the plants were observed in just a single pool among many other pools—often with several hundred plants densely packed into a very small area. The plants disappear rapidly once the pool evaporates and sediments begin to desiccate. In Minnesota, this species has also been recorded in seasonally; inundated mud flats not associated with bedrock, including a wet spot surrounded by cultivated land (MNDNR 2011b).

MUDWORT (*Limosella aquatica*)

Prior to MCBS, *Limosella aquatica* (Scrophulariaceae) was known from six locations in the state, including two historical locations in Kittson County in northwestern Minnesota (MNDNR 2011a). MCBS documented fourteen new locations of this species in bedrock pools in Rock and Pipestone counties. This small annual plant is a circumboreal species of exposed mud flats. In hydrated pools, it has thin, floating leaves like tiny, 5-mm-wide lily pads, but when stranded in mud it morphs into a more succulent-leaved terrestrial form with tiny white flowers.

SLENDER PLANTAIN (*Plantago elongata*)

Plantago elongata (Plantaginaceae) was known only from two historical collections and one more recent collection in Rock and Pipestone counties (MNDNR 2011a). MCBS found eight new locations of this species on the margins of rock pools and in seasonally wet zones between pools in Rock and Pipestone counties, including some large populations. This annual stemless plantain is a western species that reaches the easternmost edge of its United States range in southwest Minnesota (MNDNR 2011a). Unlike most of the rare ephemeral pool species, several *Plantago elongata* populations were documented on outcrops that receive heavy grazing pressure.

THREE-STAMENED WATERWORT (*Elatine triandra*)

Elatine triandra (Elatinaceae) is a rare aquatic species found in bedrock pools in southwestern Minnesota, as well as on the margins of lakes in northeastern Minnesota. Prior to MCBS, four locations were known from southwestern Minnesota outcrops, including historical collections (MNDNR 2011b). MCBS documented five new locations of this species in Sioux quartzite bedrock pools in Rock and Pipestone counties. In southwest Minnesota, individuals of this species are rarely more than two centimeters long. Like *Crassula aquatica*, *Elatine triandra* is mostly hidden in the soft sediments at the bottom of hydrated outcrop pools. Once the pools evaporate, *Elatine triandra* lies prostrate on the sediment surface and forms tiny pink flowers for two weeks or so while the former pool's sediments subside and desiccate.

LARGER WATER-STARWORT (*Callitriche heterophylla*)

Callitriche heterophylla (Plantaginaceae) was known from six historical locations in southwestern Minnesota as well as in lakes in northeastern Minnesota (MNDNR 2011a). MCBS found one new location of *Callitriche heterophylla* in the Prairie Coteau region, whereas numerous locations were observed for the similar and common Callitriche palustris. These Callitriche species form submerged leaves and rosettes of leaves that float on the pool surface. Once pools evaporate, the plants sprawl on wet mud but do not persist for long in desiccating sediments.

DISCUSSION

Some of the rare species of rock pools in Minnesota were also historically collected from the margins of prairie lakes or potholes that are not associated with bedrock outcrops, such as *Marsilea vestita*, *Bacopa rotundifolia*, *Eleocharis wolfii*, and *Limosella aquatica*. Few such historical collections of these species exist, however, suggesting that these species have always been somewhat rare in Minnesota. Over the last two centuries, nearly 99% of Minnesota's prairies and associated wetlands were drained and cultivated, with the result that these rare species may appear today to be more specific to bedrock pool habitats than they once were, and that these bedrock pools constitute the last refuge of these species in Minnesota. Bedrock exposures are quite uncommon in southwestern Minnesota, ephemeral rock pools suitable for aquatic/semiaquatic plant life are even rarer, and the rare plants of ephemeral pools are among the rarest plants in the state.

The largest and most secure populations of rare ephemeral pool species in Minnesota are in a few natural areas in Rock and Pipestone counties, such as Blue Mounds State Park, Touch the Sky National Wildlife Refuge, and Pipestone National Monument. Some rare species also occur in pastures on private land that are not overly grazed or sprayed with herbicide.

Before the MCBS survey, most natural area managers on the Prairie Coteau were unaware of the distinctive flora of ephemeral bedrock pools, as outcrop pools are little-known microhabitats containing tiny, somewhat obscure plants that are present for a brief time. These unusual components of the rich diversity of the Prairie Coteau's prairie landscape would likely not be identified without targeted searches at the right time of year. For most of the year, ephemeral pools look like baked dust and likely appear to have no significance to managers and landowners.

Rock pools on private land continue to be threatened by poor grazing practices, widespread herbicide applications, and rock mining. Continued public education about these places and assistance for landowners who would consider protection options should be a priority. Management of these sites should include avoiding any physical disturbance to the rock depressions that support these pools, including when they are dry and all that remains is bare dirt on the rock surface.

ACKNOWLEDGMENTS

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ROSINWEED GALL WASP RESPONSE TO FIRE

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Abstract: Rosinweed (*Silphium integrifolium*), and to a lesser extent cup-plant (*S. perfoliatum*), appear to be the exclusive hosts for a Cynipidae (*Hymenoptera*) gall-forming wasp, *Antistrophus silphii*. This wasp forms distinctive terminal stem galls that can be as large as a golf ball. The wasps have an annual life cycle and spend 95% of it within the galls, making them exceedingly vulnerable to fire. In addition, their host plants are now very limited on the landscape compared to presettlement times, making the wasp a species of conservation concern, especially in light of their vulnerability to fire. To test *A. silphii* recovery from a spring fire, I laid out a sampling grid, in September 2008, across a 2-ha (5-acre) site which had healthy populations of both rosinweed and its gall wasp. A portion of the site had been burned in April 2008. I counted the number of rosinweed stems and terminal stem galls per unit area across the site, in both burned and unburned areas. The site averaged more than 400 galls per acre, an infestation rate per stem of over 2%. Surprisingly, I found no reduction in galls within the burned areas. In fact, the rate of stems with galls was nearly 4 times greater in the burned area than the unburned, and there was no decline in gall occurrence with increasing distance from the unburned refugia, up to the 80 m maximum distance afford by the site. Possible explanations for these findings are varied and warrant further investigation.

Key Words/Search Terms: insects, tallgrass prairie, *Antistrophus silphii*, Hymenoptera, Cynipidae, *Silphium integrifolium*, conservation

INTRODUCTION

Fire is an integral ecological process in tallgrass prairie and thus critical to the maintenance of prairie vegetation (Gleason 1913, Curtis 1959, Vogl 1974, Kucera 1981, Henderson 1982, Towne and Owensby 1984, Axelrod 1985, Knapp and Seastedt 1986, Pyne 1986, Gibson and Hulbert 1987, Hulbert 1988, Collins and Wallace 1990, Leach and Givnish 1996, Bowles et al. 2003, Rooney and Leach 2010). However, there are many prairie-specialist insect species (insects dependent upon remnant prairie or restored prairie vegetation), which are negatively affected by fire (Opler 1981, Panzer 1988, Reed 1997, Harper et al. 2000, Panzer and Schwartz 2000, Dietrich and Voegtlin 2001, Swengel 2001, Panzer 2002, Panzer 2003, Tooker and Hanks 2004). If not given enough time to recover, or unburned refugia to recover from, some of these species may be eliminated by the use of fire in managing prairies. Therefore, in order to safeguard the entire prairie ecosystem, it is important to know how those insects that are both prairie-restricted and sensitive to fire respond to, and recover from, fire.

Working in central Illinois in the late 1880s, as the last of the tallgrass prairies were succumbing to the plow and cow, Dr. Clarence P. Gillette, along with collaborator Mr. C. A. Hart, discovered a species of small (3-4 mm) gall-forming wasp associated with *S. integrifolium* (rosinweed) and *S. perfoliatum* (cup-plant). Gillette (1891) published a description of this new species, *Antistrophus silphii* Gil. (Hymenoptera: Cynipidae).

Figure 1. Adult *Antistrophus silphii* gall wasp reared from a rosinweed (*Silphium integrifolium*) terminal stem gall in Wisconsin. Photo by S. Sauer.

Gillette concluded that adult specimens of this wasp (Figure 1) reared from either rosinweed or cup-plant were one and the same species. However, recent unpublished work by Zhiwei Liu and colleagues at Eastern Illinois University is demonstrating that the terminal stem gall wasps found on rosinweed and cup-plant (*A. silphii*) may actually be two different, but morphologically very similar, species. This is based on host selection data (Ginder and Liu 2010), and an ongoing project looking at genetic markers (personal communication).

Figure 2. Terminal stem gall on rosinweed (*Silphium integrifolium*). Photo by R. Henderson.

Gillette described *A. silphii* as forming abrupt, large (2.5-4.5 cm) subglobular swellings (galls) at the tips of the stems of rosinweed and cup-plant (Figure 2). Each gall may support up to 20 or 30 wasp larvae (Fay and Hartnett 1991, Fay and Samenus 1993). Gillette found that the wasps overwintered as pupae in individual cells within the pith of the galls (Figure 3), emerging as adults in May or June. The adults are active for only a brief period of time. Approximately 95% of the wasp's annual life cycle is spent within the galls. See Fay and Hartnett (1991) for a complete description of the wasp's habits.

Figure 3. Cross-section of rosinweed terminal stem gall showing *Antistrophus silphii* larvae chambers. Photo by S. Sauer.

The wasp's life history makes it extremely vulnerable to fire. Fay and Samenus (1993), working in Kansas, found survival through spring fire by *A. silphii* to be very rare. They concluded that wasps must reestablish after fire almost exclusively by immigration from unburned areas. Therefore, for the conservation of this prairie-dependent species, it is important to know how far the wasps disperse and how quickly they recolonize post-fire.

Rosinweed terminal stem gall wasps appear to be uncommon in Wisconsin, and are often low in numbers when present. Therefore, when an opportunity presented itself, at the largest known *A. silphii* population in the state, to investigate gall density in relation to fire, I took it. My research questions were: (1) What is the difference in gall density between recently burned and unburned areas? And (2) Within burned areas, is there a correlation between gall density and distance from unburned refugia?

METHODS

The study took place at a privately owned site known as Underwood Prairie, located in south-central Wisconsin, in eastern Iowa County, 20 miles southwest of Madison. It is an 8-ha (20-acre) parcel that has been managed by The Prairie Enthusiasts since 1999, and is adjacent to The Prairie Enthusiasts' 530-acre Mounds View Grassland Preserve, which consists of remnant prairie sod in various stages of recovery and nonnative cool-season fields being gradually restored to prairie. The study site consists of high-quality remnant dry-mesic prairie (Figure 4) and buffer lands planted to prairie. Management has consisted of rotational fire, mowing, and removal of weeds, trees, and brush. Much of the most-degraded areas had been burned annually for 5 to 10 years prior to the study.

Figure 4. Underwood Prairie (Iowa Co., Wisconsin) in early July. Photo by R. Henderson.

Within the site, a 2.1-ha (5.2-ac) study area was selected, which had good populations of rosinweed and terminal stem galls. The study area was approximately 40% remnant prairie and 60% restored prairie. The soils are thin to moderately deep silt loam over fractured dolomite bedrock. The study area has a west- and southwest-facing aspect with a 5% to 10% slope. In April 2008, 70% of the study area was burned.

In early October 2008, I laid out a sampling grid (Figure 5). Survey transects (150 meters long) were run from east to

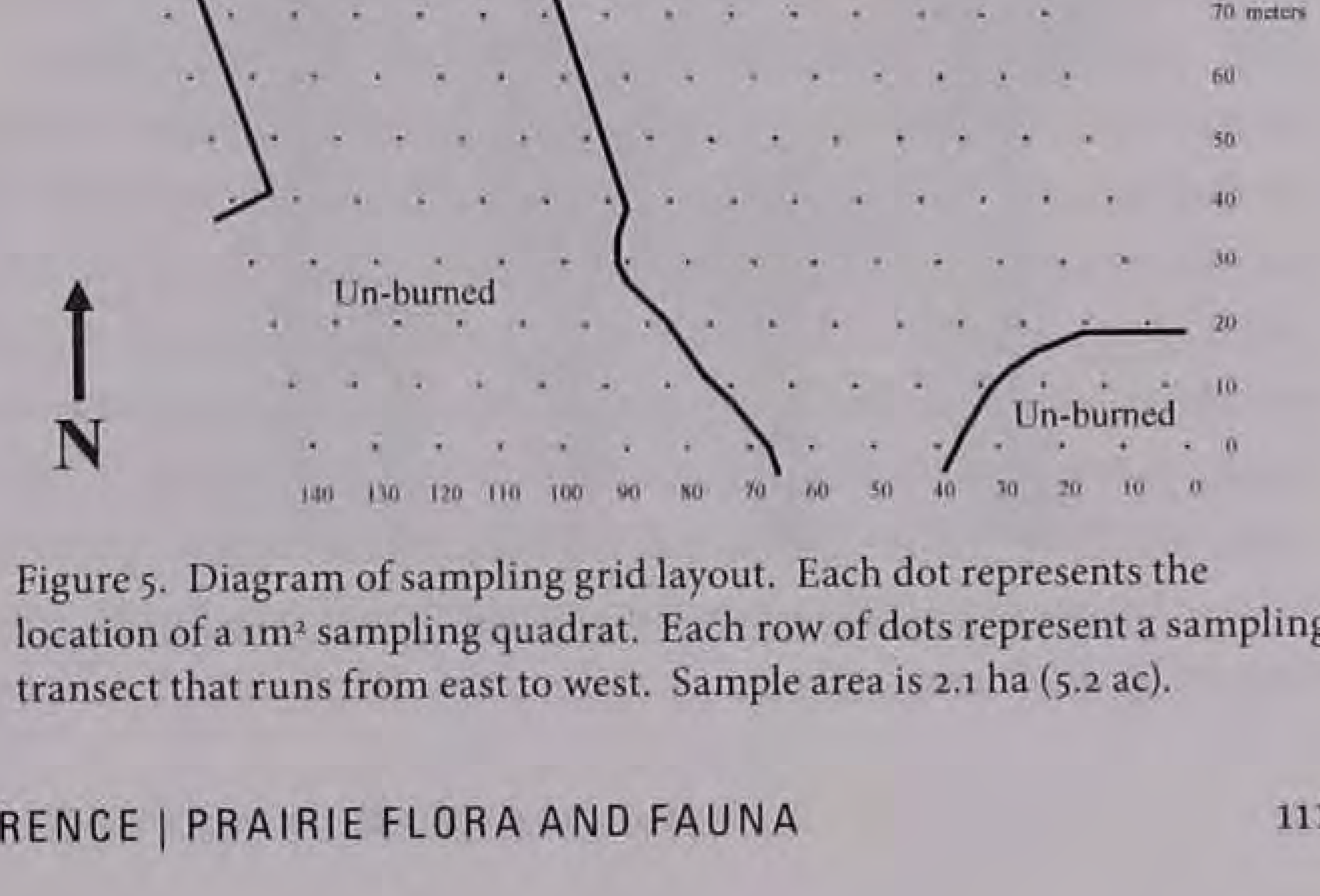

Figure 5. Diagram of sampling grid layout. Each dot represents the location of a 1m² sampling quadrat. Each row of dots represent a sampling transect that runs from east to west. Sample area is 2.1 ha (5.2 ac).

west at 10-meter intervals. At every 10 m along a transect line, a one square-meter quadrat frame was placed on the ground (Figure 6). The number of rosinweed stems and galls present were counted and recorded for each quadrat. In addition, for a distance of 3 m on each side of the transect line (6 m total width) along each 10 m length of the line between each 1m² quadrat survey point, all rosinweed stem galls were counted and recorded (a plot size of 60 m²) (Figure 6). At each sample point, I recorded whether or not it had been burned in April. This method resulted in 60% of the study area being sampled with the 60 m² plots and 1% with the 1 m² plots. The area east of the study area was nonnative cool-season grass fields with no rosinweed present. The areas north and west had rosinweed and were burned that spring. The area south had rosinweed and was not burned.

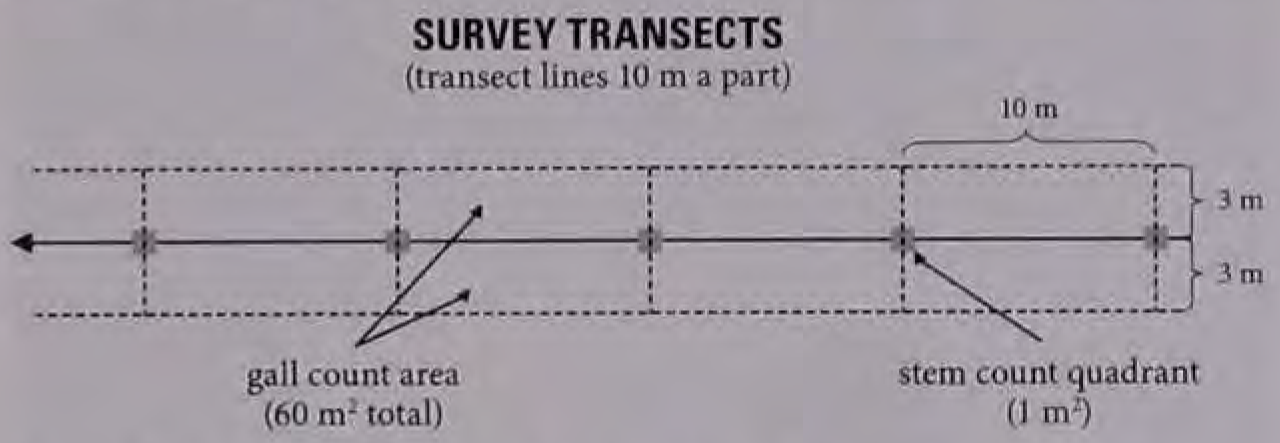

Figure 6. Diagram of a survey transect line. Lines were 10 m apart.

Individual rosinweed plants produce from one to 100 shoots or stems, with 20-50 being most typical (Fay and Hartnett 1991, Fay et al. 1996). When the plants are growing in close proximity to each other, it is often difficult to determine where one individual ends and another begins. Therefore, I made no attempt to count individual rosinweed plants or galls produced per plant. Instead, I counted numbers of shoots. A maximum of one gall is produced per shoot (stem) and it is always located at the terminal end of the stem. Most of the time, its presence precludes flower development. The galls can vary in size and shape (Figure 7), but are generally very obvious and easy to count. It seems logical to assume that the larger the gall the more wasp larvae present, but such a correlation has yet to be documented. There also is uncertainty about parasitism rates in the larvae, which can be high. Fay et al. (1996) found approximately 1/3 of the wasp larvae to be parasitized. Therefore, equating gall size with number of adult wasps produced may prove difficult.

Figure 7. Variation in size and shape of rosinweed terminal stem galls. Photo by R. Henderson.

Table 1. Rosinweed stem and wasp-gall densities. N = number of quadrats sampled.

	GALLS/HA	STEMS/ HA	% OF STEMS WITH GALLS	N
ENTIRE STUDY AREA	1,012	44,251	2.3 %	207
BURNED PORTION	1,311	48,874	2.7 %	151
UN-BURNED PORTION	205	31,786	0.7 %	56

RESULTS

The study area averaged 4.43 rosinweed shoots (stems) per 1 m², but they were not evenly distributed (Table 1). The unburned portions had 1/3 fewer stems than the burned (Figure 8). Overall, an estimated 1,012 galls per ha (410/ac) were produced within the study area. Far more galls were produced in the burned areas than the unburned, even correcting for differences in rosinweed density. The frequency of rosinweed stems with galls was nearly four times higher in the burned areas than the unburned (Figure 9).

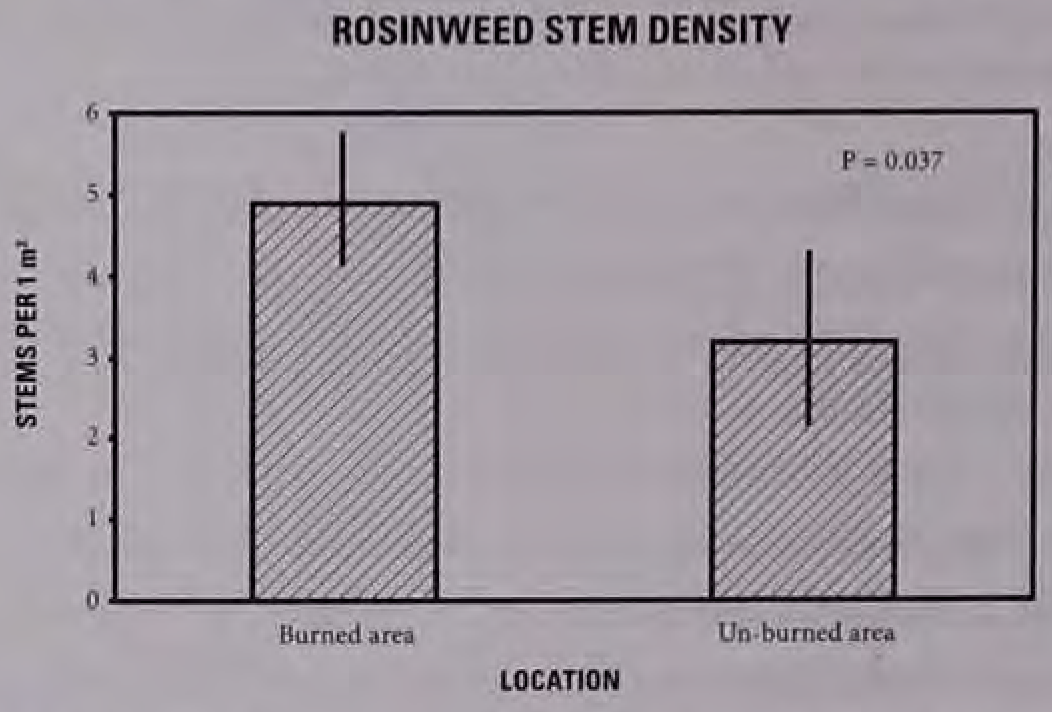

Figure 8. Density of rosinweed stems within burned (N=151) and unburned (N=56) areas. Bars indicate the 95% confidence interval.

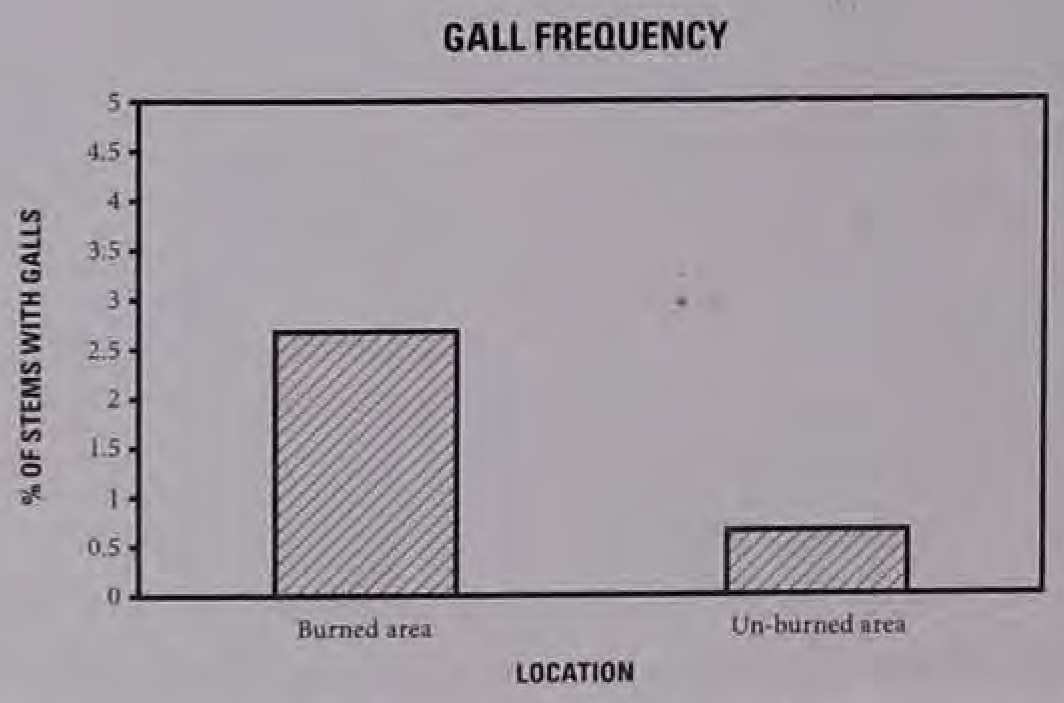

Figure 9. Frequency of galls per stem within burned and unburned areas.

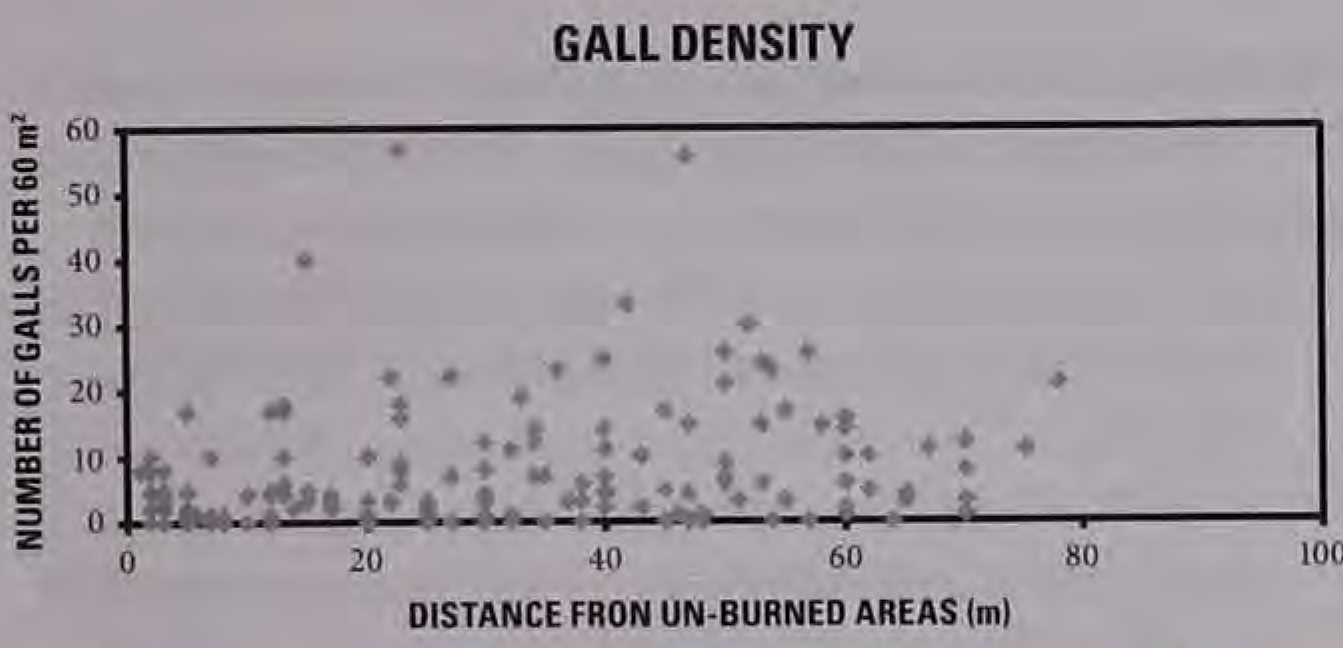

Figure 10. Gall density in burned areas plotted against distance from unburned refugia. No correlation was found.

No edge effect was detected on gall density going from unburned refugia into the burned areas. Within the 80-m maximum distance out from unburned areas available at the study site, no correlation was detected between distance from unburned refuge and density of *A. silphii* galls (Figure 10).

DISCUSSION

The fact that the galls were spread evenly across the burned area suggests that 80 m is an insufficient distance for detecting declines in colonization rates of *A. silphii* with increasing distance from unburned refugia. It is possible that some wasps survived the fire in situ, within pockets of low fire intensity, as described by Fay and Samenus (1993), but the uniformity of the fuel load and the high intensity of the April 2008 fire made this unlikely at this site. In addition, the sheer number of galls across the entire burned area (estimated 1,930 galls) suggests that large numbers of wasps must have come from the unburned area. Nonetheless, it is possible that a few galls may have escaped the heat of the fire sufficiently enough to allow some wasps to emerge within the burn area, but the numbers would have been very low. Even though galls have been reported to have up to as many as 30 wasp larvae chambers per gall (Fay and Hartnett 1991), the averages are far less when it comes to adult wasps produced per gall. Fay and Samenus (1993) found a mean of 12.9 adults emerging per gall. Fay et al. (1996) reported only 1.5 adults per gall on average.

I have observed *A. silphii* galls on a recently established rosinweed population located on a site that has been burned annually since the rosinweeds were introduced (three or four years). The population was introduced via seed, and is 200 m from the nearest source of rosinweed galls. This suggests that even 200 m may not be a significant obstacle to *A. silphii* recolonization. The work done on *A. silphii* at Konza Prairie in Kansas (Fay and Hartnett 1991, Fay and Samenus 1993, Fay et al. 1996) was done on an old field invaded by rosinweed that was burned annually, at least during the years of their studies. The fact that this site produced ample galls for their investigations, suggests that *A. silphii* has a very good ability to recolonize following fire. Unfortunately, the authors did not state clearly the size of the annually burned field, or indicate the maximum or typical distance the wasps seemed to be able to disperse across. Research is needed to quantify both the typical and maximum dispersal distances of *A. silphii* from unburned refugia into adjacent burned areas. These data are critical to determining what portion of rosinweed gall-wasp habitat may be burned without risking loss of the wasp.

Given the obvious vulnerability of *A. silphii* to fire (Fay and Samenus 1993), it was surprising to find gall infestation rates to be nearly four times higher in the burned areas than in the unburned. There are a number of possible explanations for this, but additional investigation is required to adequately assess the true cause.

First, the results may be a simple fluke of annual variation. Repeating the study in more years and at other sites should adequately address annual variability.

Second, there may be a difference in host plant productivity, food value, or palatability between burned and unburned rosinweed that results in the wasps being attracted to burned areas over the unburned. Fay et al. (1996) found no effect of rosinweed vigor or fitness on *A. silphii* production or performance. Therefore, it is unlikely that wasps are attracted to burn areas because the rosinweeds are more productive. The explanation of differences in palatability, however, may yet have some merit.

Third, the wasps may simply be attracted to areas of highest rosinweed stem density. If you recall, the burned areas in this study had over 50% higher stem density per unit area than the unburned areas. If stem density proves to be the key, then determining whether a difference in stem density is linked to fire history or some other factor unrelated to fire would be the next step in understanding the interactions of this system.

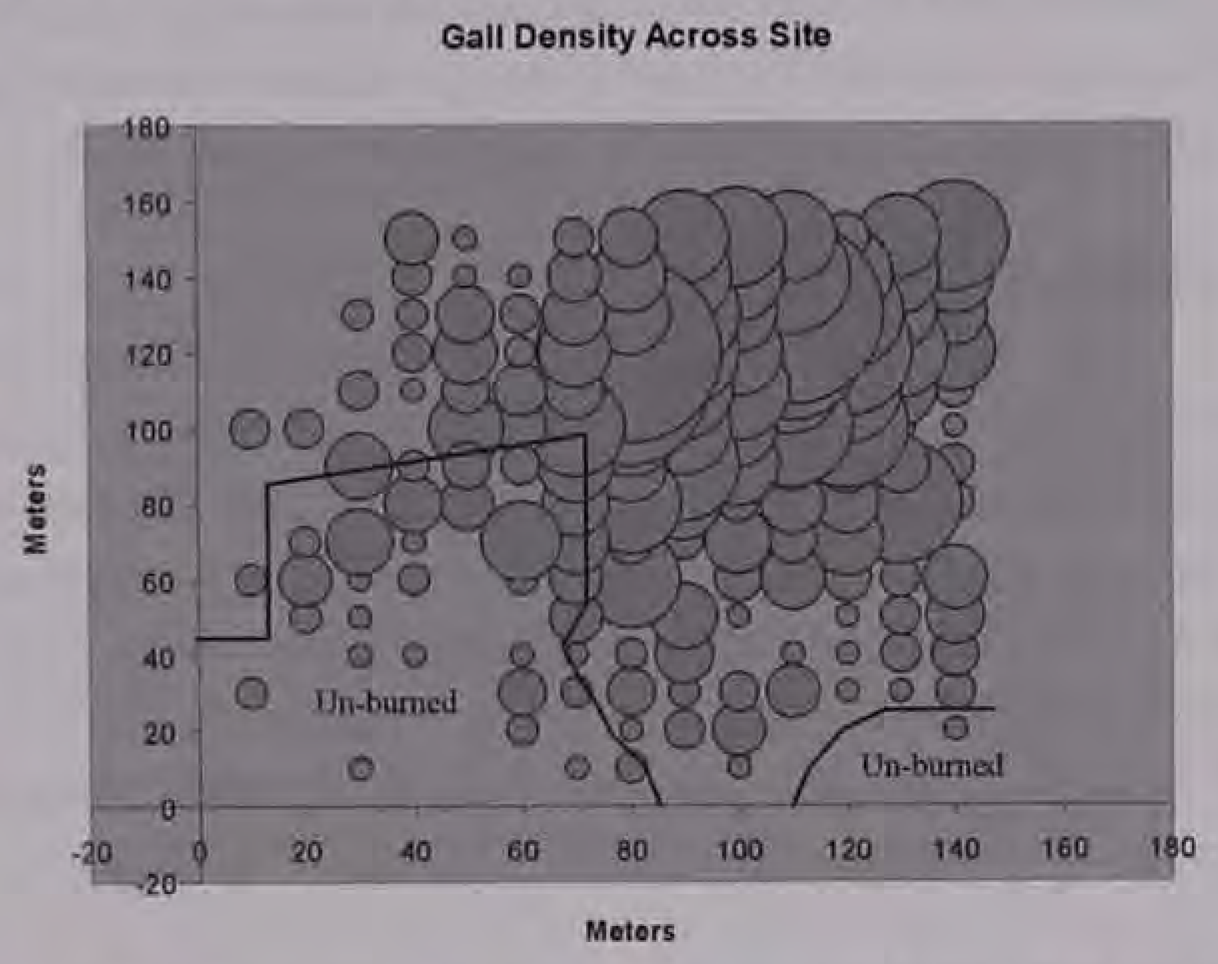

Figure 11. Bubble graph of gall density across sample area. The larger the bubble the more galls present.

Lastly, the wasps may be responding to topography. The study area sits upon a gradually sloping hillside that slopes to the west and south. A bubble graph of gall density across the study site (after transforming the rhomboid shape of the study area into a rectangle) shows the highest density of galls coinciding with the highest ground of the study area (Figure 11). A comparison of this chart with a bubble chart of stem density (Figure 12), suggests that topographic position may play more of a role than stem density in determining wasp-gall location. If the wasps have a tendency to "hill-top," this

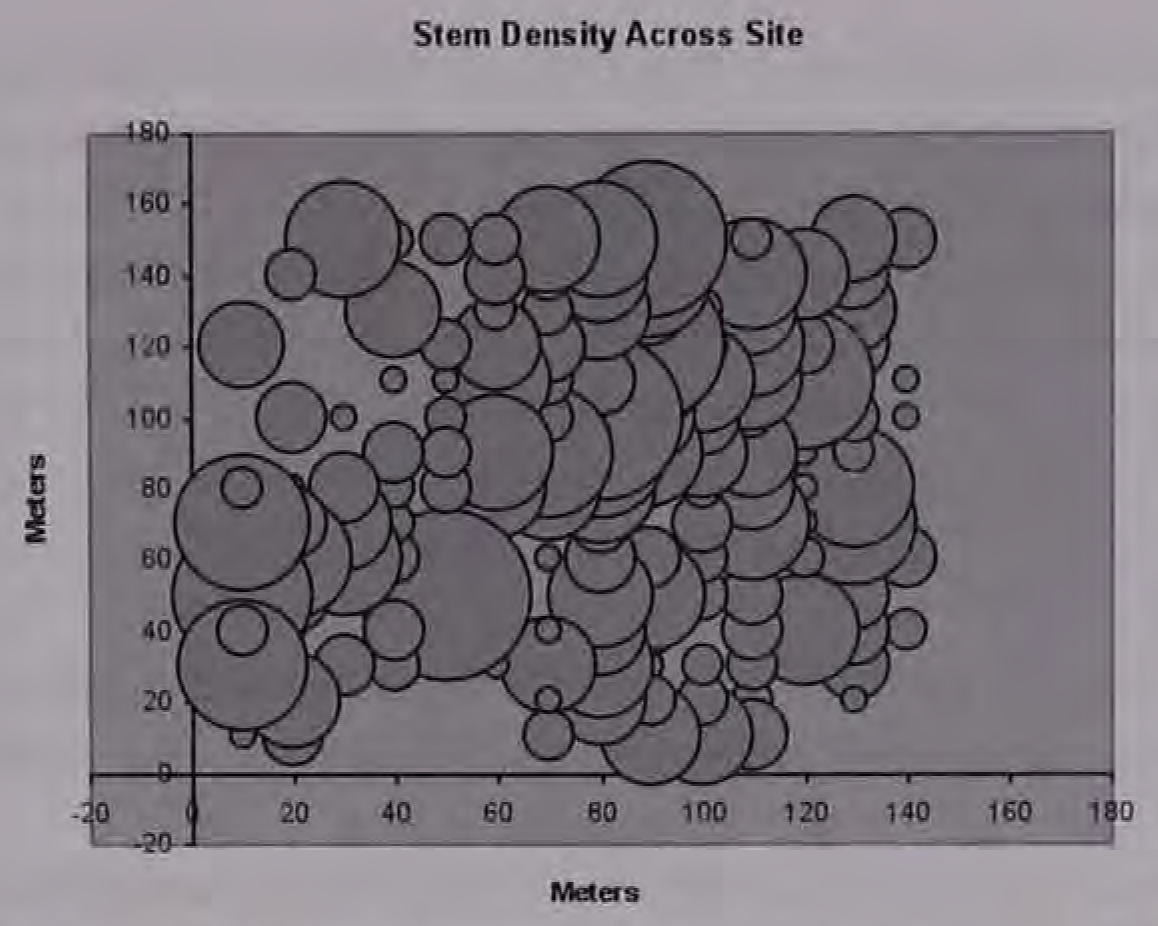

Figure 12. Bubble chart of rosinweed stems density across sample area. The larger the bubble the more stems.

alone might explain the higher rate of galls in the burned area, which includes all of the highest ground.

Another factor to consider in this study is that more galls per unit area may not equate to more wasps. The study did not address any difference in number of wasps produced per gall between burned and unburned areas. This is a variable that needs to be addressed before drawing any firm conclusions.

Should the apparent preference in this study of rosinweed terminal stem-gall wasps to use burned areas more than unburned prove to be real and consistent, then *A. silphii* may be an example of a species readily killed by fire that actually benefits from the occurrence of fire when applied appropriately. If so, this maybe a case of short-term loss for long-term gain, similar to the example of Fender's blue butterfly (Schultz and Crone 1998).

ACKNOWLEDGMENTS

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SILPHIUM GALL WASPS: LITTLE-KNOWN PRAIRIE SPECIALISTS

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Abstract: The original prairies of the Midwest have nearly vanished, and thus so have many of the insects linked to prairie vegetation. In 1891, five species of gall-forming wasps, *Antistrophus* spp. (Hymenoptera: Cynipidae), were described from various plants in the genus *Silphium* (compassplant, prairie dock, rosinweed, and cup-plant) growing in Illinois prairies. These wasps are thought to live only in *Silphium* spp., which are characteristic plants of our Midwest prairies. *Silphium* spp. once dominated the landscape, but now they are mostly limited to scattered remnants of original prairie and prairie plantings. For one hundred years following the description of these wasps, the scientific community paid them very little attention. Then starting in the early 1990s, researchers began to take interest in the group, describing additional species and investigating the wasps' ecology and response to fire. However, the distribution and conservation status of *Silphium* gall wasps are still very much a mystery. Recently, two species have been documented for the first time in Wisconsin, and more species are suspected of being present. Efforts to document the distribution and status of these uncommon prairie specialists in Wisconsin and adjoining states are being made.

Key Words/Search Terms: insects, tallgrass prairie, *Antistrophus*, Hymenoptera, Cynipidae, compassplant, prairie dock, rosinweed, cup-plant, conservation

DISCOVERY

Working in central Illinois in the late 1880s, as the last of the tallgrass prairies were succumbing to the plow and cow, Dr. Clarence P. Gillette, along with collaborators Mr. C.A. Hart and Mr. J. Marten, discovered several species of small (2-5 mm) gall-forming wasps associated with four plants of the genus *Silphium*: *S. laciniatum* (compassplant), *S. terebinthinaceum* (prairie dock), *S. integrifolium* (rosinweed), and *S. perfoliatum* (cup-plant). These were among the most dominant plants of the original eastern tallgrass prairies. Gillette and company made observations in the field and reared adult wasps from plant tissue. From this work, Gillette published descriptions of five Cynipid (Hymenoptera) wasps of the genus *Antistrophus* (Gillette 1891).

Antistrophus silphii

Gillette described this species as forming abrupt, large (2.5-4.5 cm) subglobular swellings (galls) at the tips of the stems of rosinweed and cup-plant (Figure 1). Each gall supported many wasp larvae. Gillette concluded that adult specimens (Figure 2) reared from both species of plants "were in every way identical with those from galls of the other species." He found that the wasps overwintered as pupae in individual cells within the pith of the galls, emerging as adults in May or June (Figure 3).

Figure 1. Terminal stem gall on rosinweed (*Silphium integrifolium*). Photo by R. Henderson.

Figure 2. Adult *Antistrophus silphii* gall wasp reared from a rosinweed (*Silphium integrifolium*) terminal stem gall in Wisconsin. Photo by S. Sauer.

Figure 3. Cross-section of rosinweed terminal stem gall showing *Antistrophus silphii* larvae chambers. Photo by S. Sauer.

Antistrophus laciniatus

Gillette described this species as forming individual, small (4-5 mm), egg-shaped galls occurring in clusters in the flower disks of compassplant (Figure 4). Each gall supports one wasp. The galls occupy the center of the flower disk among the male (sterile) florets. The flowers appear unaffected by the wasps, with the gall cluster becoming evident only as the flower head ripens and falls apart, revealing the galls.

Figure 4. *Antistrophus laciniatus* galls from the flower disks of compassplant (*Silphium laciniatum*) in WI. Photo by R. Henderson.

Antistrophus rufus and *A. minor*

Gillette reared both of these species from flower stalk stems of compassplant, but unlike the previous two species, there was no evidence of gall formation. Instead, he found that the wasps pupated in little cells within the pith just under the outer wall of the stem (Figure 5). *A. minor* was smaller and less numerous than *A. rufus*. Dr. Gillette, along with Mr. Hart, visited "fields" of compassplant and "found that the majority of the stems were more or less infested with cynipidous larvae, hundreds of which could, in some cases, be found in a single stem." They went on to investigate other *Silphium* species in the area and "found similar larval cells abundant in *S. perfoliatum*, *S. terebinthinaceum*, and *S. integrifolium*." However, they did not rear specimens from these stems to see what species of *Antistrophus* wasps were present.

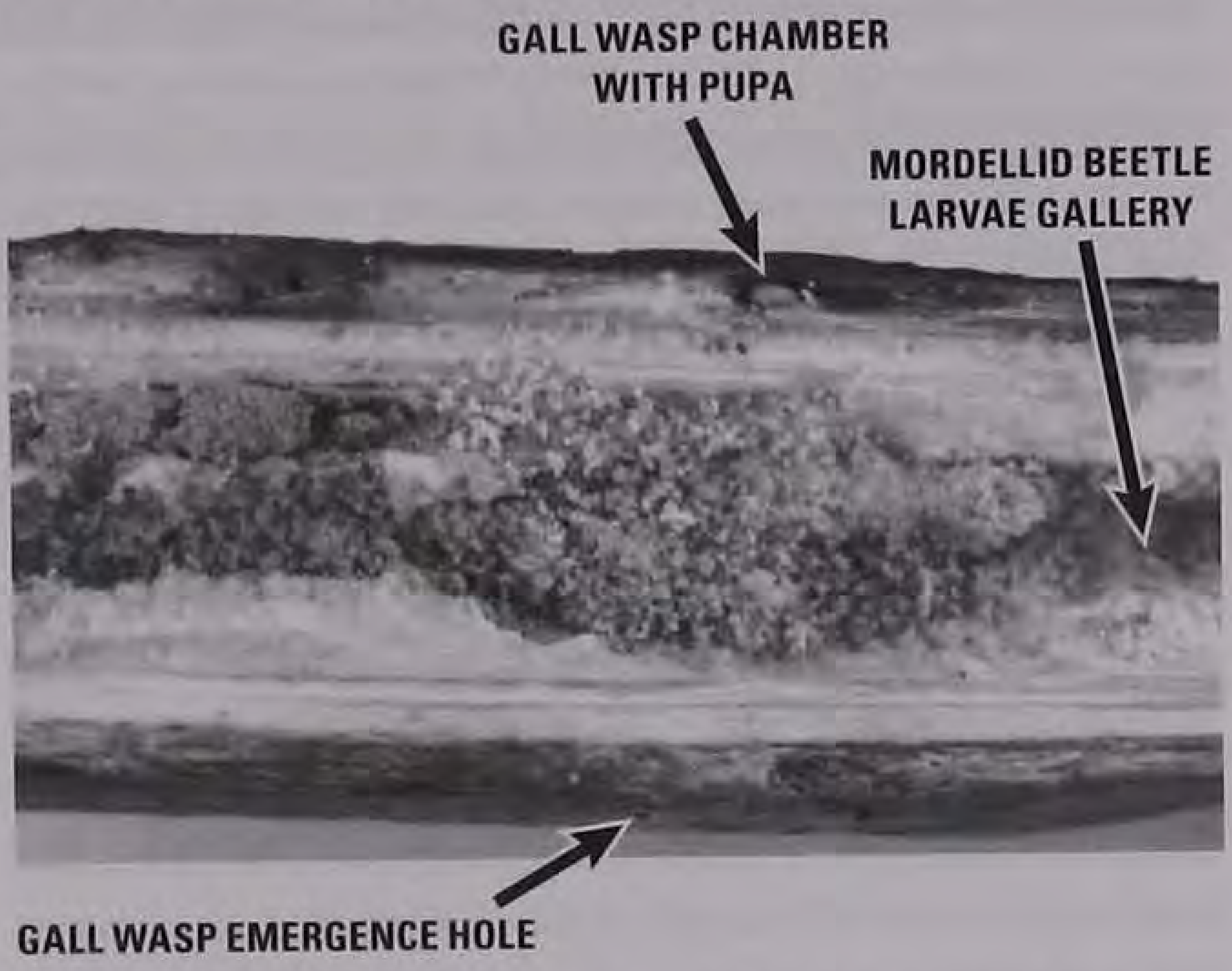

Figure 5. Cross-section of compassplant (*Silphium laciniatum*) stem inhabited by *Antistrophus* wasps and Mordellid beetle larvae. Photo by S. Sauer.

Antistrophus bicolor

Gillette did not offer much information about this species. He described it from a single specimen from Normal, Illinois. Both the host plant and gall of the wasp were unknown to him. Seventy years later, Weld (1959) reports the collection date of Gillette's type specimen as being July 6, 1884, and the accompanying accession catalogue as stating "from *Silphium integrifolium*."

FORGOTTEN AND REDISCOVERED

After Gillette's work, there appears to have been nothing published about these *Antistrophus* species for the next 35 years, except for some confusing name-changing and occasional misidentifications by taxonomists, and the passing along (sometimes with errors) of Gillette's original observations (Bassett 1900, Kieffer 1902, Beutenmuller 1910). For example, Beutenmuller (1910) listed all four species of *Silphium* as hosts for *A. rufus*. This is likely an error, since his only reference, Gillette (1891), confirmed *A. rufus* from only compassplant. The Antitrophus wasp larvae Gillette found in the stems of the other species were not reared out and identified.

In 1912, Lewis H. Weld began to collect specimens and make field observations of *Silphium* gall wasps in northeast Illinois (Weld 1926). He reaffirmed compass plant as the host for *A. laciniatus*, *A. rufus*, and *A. minor*, and documented *A. minor* as also using prairie dock. In 1915 and 1917, Weld documented *A. silphii* forming terminal stem galls on cup-plant in eastern Nebraska. This was the first published record of any *Silphium* gall wasp being found outside of Illinois. Weld also presented information indicating that Gillette's *A. bicolor* should be considered a synonym of an earlier described species (*Aulacidea harringtoni*). Weld found the plant genius *Lactuca* to be a host for this gall wasp. However, 30 years later, Weld (1959) treats *A. bicolor* as a distinct species. In the Catalog of Hymenoptera in America North of Mexico (Burks 1979), *Antistrophus bicolor* is listed as both a good species in its own right and as a synonym of *Aulacidea harringtoni* (same author citation for both listings!), with the host listed as *Lactuca* spp. Burks (1979) listed Illinois as the only location for *A. bicolor*, and, he states, "Said to produce galls on *Silphium integrifolium*," but he gave no reference for this. Gillette (1891), the only Illinois source, provided no information as to hosts or galls for *A. bicolor*. Weld (1959) reported the type specimen as coming from *S. integrifolium*. *A. bicolor* appears to be a poorly known species in need of both taxonomic and ecological work.

Burks's 1979 comments on *A. rufus* listed Kansas, in addition to Illinois, as a location for the species. This was the first reported record outside of Illinois for this species, and is presumably based on a museum specimen, but he provided no reference. He listed compassplant as the only host for *A. rufus*.

As far as we have been able to determine, it was not until 1991 that any new work on the ecology and habits of *Silphium* gall wasps was published since the works of Gillette and Weld, 100 and 80 years earlier, respectively. Then the activity of work on *Silphium* gall wasps picked up significantly and remains steady to the present. The "rediscovery" of these wasps started in 1988, with work at the Konza Prairie Research Natural Area in eastern Kansas, where Philip Fay and others began to investigate the ecology of *A. silphii* on rosinweed and its effect on that host plant (Fay and Hartnett 1991, Fay and Samenus 1993, Fay et al. 1996). Their research appears to constitute the first and only published reports of this gall wasp in Kansas. The authors reported it being very common in the area, with over a third of rosinweed shoots galled, and up to 30 larvae per gall (Fay and Hartnett 1991). The authors also reported an unknown gall in the disk florets of rosinweed at Konza. They speculated that it might be *A. laciniatus* (Fay et al. 1996), but did not rear out adult specimens to confirm. It appears that *A. laciniatus* has yet to be confirmed outside of Illinois.

In 1999, John Tooker and others began to investigate the ecology and habits of internal stem *Antistrophus* wasps in central Illinois, where the wasps were first discovered. They looked at the endophytic insect communities inhabiting the stems of compassplant and prairie dock (Tooker and Hanks 2004a). They found both *A. rufus* and *A. minor* to be common at all eight sites they surveyed and present in both species of *Silphium* (compassplant and prairie dock). However, *A. rufus* was three times more abundant than *A. minor*. The average density of internal galls per stem was 80 for compassplants and 62 for prairie dock. This appears to be the first published documentation of *A. rufus* using prairie dock. Tooker and Hanks also confirmed that *A. rufus* and *A. minor* behave just as Gillette (1891) described for *A. silphii*, in that the wasps overwinter as pupae in individual cells within the pith of the galls, in this case within the pith of the stems, and emerge as adults in May or June.

Tooker et al. (2004b) went on to study the genetic, morphological, and ecological differences of *A. rufus* populations using compassplant, prairie dock, and cup-plant. They concluded that each species of plant has its own species of wasp within what is now considered the *A. rufus* complex of species (an apparent example of co-evolution). The recognized species are now *A. rufus* (restricted to compassplant), *A. meganae* (restricted to prairie dock), and *A. jeanea* (restricted to cup-plant). This now makes six *Antistrophus* species confirmed as being restricted to prairie *Silphium* spp. But the story is still unfolding.

Recent work (yet to be published) by Zhiwei Liu and colleagues at Eastern Illinois University is demonstrating that the terminal stem gall wasps found on rosinweed and cup-plant (*A. silphii*) may actually consist of two different but morphologically very similar species, as was the case with the *A. rufus* complex. This is based on host selection data (Ginder and Liu 2010) and an ongoing project by Liu and colleagues looking at genetic markers (Z. Liu, personnel communication, 2010). If these putative species prove valid, that will make seven species of *Antistrophus* tied to prairie *Silphium* spp. But this still may not be the end of the story. Questions still remain about the host, ecology, and taxonomic status of *A. bicolor*, and no one has documented what *Antistrophus* species are in the stems of rosinweed, as first noted by Gillette (1891). Might this be yet another new species or simply *A. bicolor*? We have been unable to find published documentation of internal stem galls using rosinweed since Gillette's original observations. A complete list of species along with listings of published host associations and the states they have been reported in is provided in Table 1.

ASSOCIATED INSECTS

The recent work on *Silphium* gall wasps has revealed a whole community of endophytic (within a plant) insects associated with prairie *Silphium* spp., in some cases possibly dependent upon them. Fay et al. (1996) observed an undescribed species of Eurytoma wasp regularly parasitizing *A. silphii* wasps in rosinweed galls. Tooker and Hanks (2004b), during a detailed study of endopyhtic insects of compassplant and prairie dock in Illinois, found eight species of parasitoid wasps attacking *A. rufus* and *A. minor* gall wasps, or in some cases possibly attacking the parasitoid wasps attacking the *Antistrophus* wasps. They presented evidence that one of

Table 1. *Antistrophus* spp. (gall wasps) associated with prairie *Silphium* spp., along with documented host plant and location records.

WASP SPECIES	HOST PLANT	SOURCE						
		GILLETTE 1891	WELD 1926 & 1959	WINTERRINGER 1961	BURKS 1979	FAY et al. 1990	TOOKER et al. 2000	LIU 2010
A. bicolor	rosinweed?	IL (Normal)[1]	??[2]		IL in rosinweed[3]			
A. jeanae	cup-plant	IL					IL	
A. laciniatus	compassplant	IL (Champaign)	IL (Glenview)					
A. meganae	prairie dock	IL					IL	
A. minor	compassplant	IL	IL (Evanston)		IL		IL	
A. minor	prairie dock		IL (Evanston)				IL	
A. rufus	compassplant	IL	IL (Evanston)		IL & KS[3]		IL	
A. silphii	rosinweed	IL (Champaign)	IA[4]			KS		KS, MO, & IL
A. silphii	cup-plant	IL (Normal)	NE (Valley & Omaha) & IA[4]	IL	IL & NE			IL
Antistrophus sp.	rosinweed	IL						

[1] No rearing from stems or galls reported.

[2] Taxonomy unclear; reared from *Lactuca* (1926); reports the accession catalogue of the type specimen as being from rosinweed (1959).

[3] No source given, assumed to be from specimen labels or accession catalogues.

[4] Unclear as to whether the reported IA collection was from cup-plant or rosinweed.

the parasitoids, *Eurytoma lutea*, may be a specialist on *Antistrophus* wasps. They also consistently found the larvae of a mordellid (tumbling flower) beetle, *Mordellistena aethiops*, burrowing through the stems of compassplant and prairie dock (Figures 5 and 6) along with three species of parasitoid wasps that appear to attack this beetle. They demonstrated that *M. aethiops* larvae are omnivores, consistently eating both plant tissue and wasp larvae (Tooker and Hanks 2004c), which is unusual for the *Mordellistena* genus. Their larvae are normally reported as stem-boring herbivores (Ford and Jackman 1996).

Figure 6. Tumbling flower beetle, presumably *Mordellistena aethiops*, emerging from a *Silphium* stem. Photo by S. Sauer.

DISTRIBUTION, STATUS, AND FUTURE WORK

Since 1996 we have had a collaborative project looking into which insects species are associated with and dependent upon remnant prairies in Wisconsin and the Upper Midwest. We have also focused on the status, distribution, and response to management of these remnant-dependent species. However, since up to 2,000 species likely fall into this category, we have not been able to address all taxonomic groups, and thus we had paid relatively little attention to gall wasps. We were aware from the literature that there were gall wasps associated with *Silphium* spp., but they had been reported only from Illinois, Nebraska, and Kansas. There were no specimens or records from Wisconsin. So we did not spend time looking for them here until the fall of 2005, when the lead author (Henderson) noticed a large terminal stem gall on rosinweed in the Madison area and began to wonder if this could be the work of *A. silphii*. He had seen galls on rosinweed on rare occasions in the past, but had dismissed them as the likely product of generalist gall-formers. This time, however, he collected the gall and waited to see what emerged. Wasps emerged, and a year later we learned that they were indeed *A. silphii*. So in 2007, we began to look informally for terminal stem galls on rosinweed. We found them in a few more locations, but they were not at all common, except at a site called Underwood Prairie, west of Madison. Galls were common there but still far below the densities observed at Konza Prairie in Kansas where 35% of the rosinweed stems had galls (Fay and Harnett 1991). In comparison, only 2% of the stems at Underwood Prairie had galls. In 2007, we also collected a few stems of compassplant to see what they might hold. They produced what has tentatively been identified as *A. bicolor*. If this proves true, this will be not only the first documentation of this species in the state, but the first documentation of its using a *Silphium* species other than rosinweed.

Now knowing that at least some *Silphium* gall wasp species are in the state, we conducted an investigation to learn more about their distribution and status here. In early April of 2009, we visited some 40 sites in southern Wisconsin and collected stems of compassplant, prairie dock, rosinweed, and cup-plant, and looked for and collected terminal stem galls on rosinweed and cup-plant. We also solicited people to be on the lookout for, and to send us, terminal stem galls on *Silphium* spp. We also collected flower disk galls from compassplant at a few sites. Anyone collecting compassplant seed in Wisconsin knows that gall clusters in the center of their flower disks are a common occurrence. But apparently no one has tried to find out what is inside those galls. So we collected them in anticipation of finding *A. laciniatum*. We also made a few collections from a variety of *Silphium* spp. in Iowa since there does not appear to be any published documentation of *Silphium* gall wasps in Iowa either.

Identifications of the emerged materials have yet to be confirmed, but there appear to be many *Antistophus* specimens, a variety of parasitoid wasps (Figures 7 and 8), and many mordellid beetles, possibly *Mordellistena aethiops* (Figure 6). If the identification of *M. aethiops* is confirmed, this would be the first record of that beetle in Wisconsin. We found terminal stem galls on rosinweed at six sites in Wisconsin and two in Iowa, and on cup-plant at four sites in Wisconsin and two in Iowa. We found *Antistrophus* gall wasps in the stems of compassplant at 14 of 18 sites in Wisconsin and 5 of 7 in Iowa; in prairie dock at 18 of 20 sites in Wisconsin; in cup-plant at 3 of 4 sites in Wisconsin and 2 of 2 in Iowa; and in rosinweed at 9 of 11 sites in Wisconsin and possibly 4 of 4 sites in Iowa. Lastly, we found what appear to be *A. laciniatum* wasps at four sites in Wisconsin.

Figure 7. Parasitoid wasp (*Eurytomid* sp.) emerging from rosinweed terminal stem gall. Photo by S. Sauer.

From what we have observed so far, *A. silphii* appears to be the rarest of the *Silphium* gall wasps in the state of Wisconsin and may be worth consideration as a species of greatest conservation need (SGCN). In the case of the cup-plant gall, listing as state endangered or threatened may be warranted. Only a third of the rosinweed populations we checked had terminal stem galls, and with the exception of the Underwood Prairie site, the gall numbers were very low. Adding to this relative rarity is the fact that today, rosinweed is very limited in its occurrence on the landscape compared to its great prevalence 200 years ago, when prairie and savanna dominated the landscape of southern and western Wisconsin. Galls on cup-plant were extremely hard to find. After visiting 50 or more cup-plant populations, and getting word out to people to send us galls, we had only six galls from four sites to show for it. We estimate that only 1% or 2% of cup-plant populations in Wisconsin support this wasp. The cup-plant gall appears to be uncommon in Illinois as well. Winterringer (1961) described it as rather uncommon in Illinois in the 1950s. Dr. Liu of Eastern Illinois University considers the cup-plant gall to be much less common than the rosinweed gall, and it should probably be considered endangered (Z. Liu, personal communication, 2010).

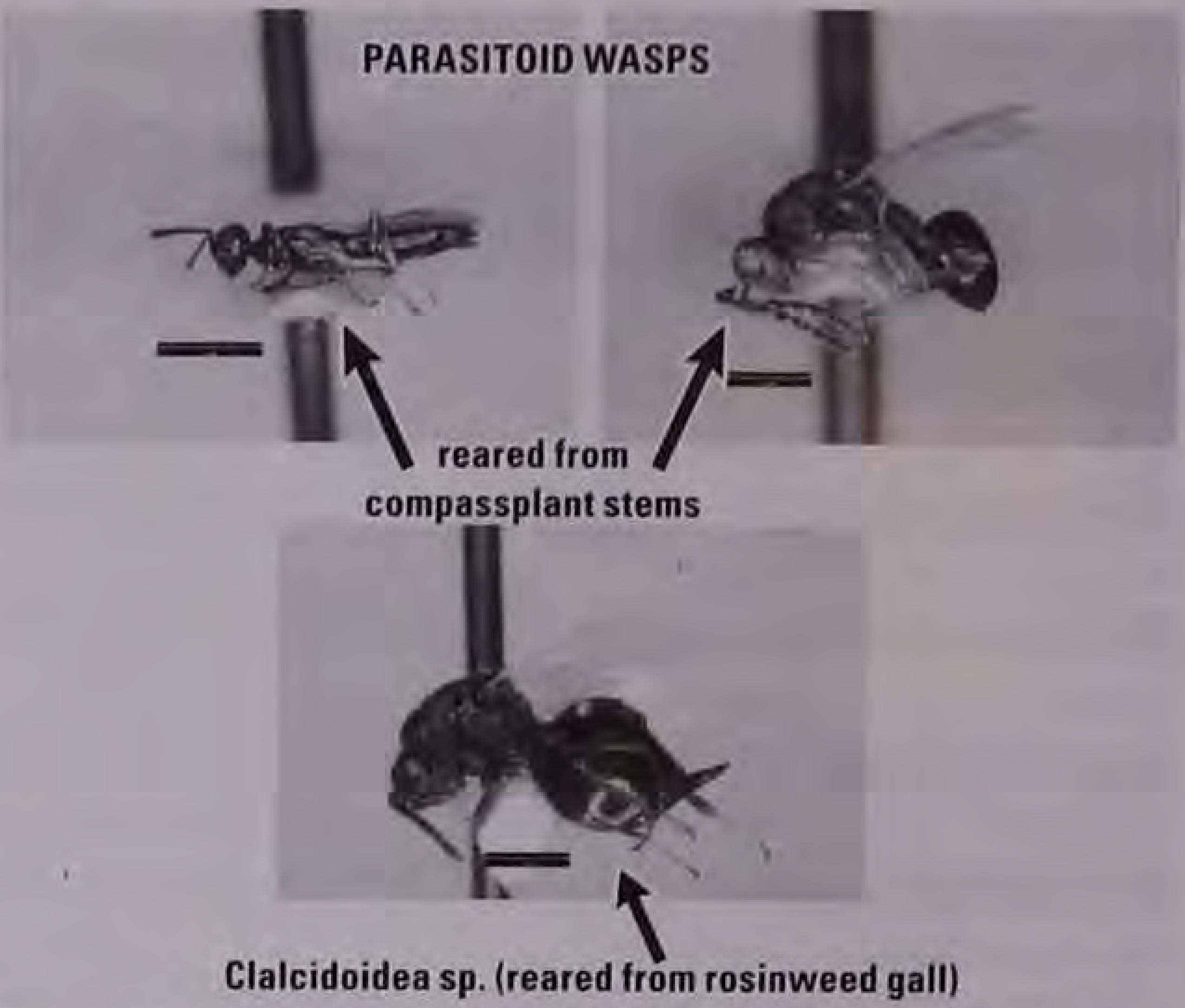

Figure 8. Parasitoid wasps reared from *Silphium* galls and stems in WI. Photos by S. Sauer.

We are currently working to get our material identified by specialists, and are seeking funding to (1) conduct a more thorough survey of *Silphium* gall wasps in the state, (2) conduct research on the effects of fire on these wasps and their associated endophytic insect communities, and (3) investigate the distribution and status of these wasps throughout their range.

This community of specialist insects shows that there is still much to learn and re-learn about the tallgrass prairie ecosystem and that a great diversity of species is a part of and dependent upon that ecosystem. Lastly, it is amazing to us that such great diversity still exists, given that only a small fraction of 1% of the original acreage of tallgrass prairie has survived and most in small isolated patches. It gives one hope that recovery and restoration of the system are still possible.

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SIZE CLASS COMPARISONS AND TREE SPECIES COMPOSITION OF BUR OAK (*QUERCUS MACROCARPA* MICHX.) SAVANNAS IN CENTRAL AND EASTERN NEBRASKA

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Abstract: Bur oak (*Quercus macrocarpa* Michx.) savannas have historically been documented in the loess hills mixed-grass prairies of central Nebraska, even though they are restricted to more mesic canyon and hillside sites. These savannas are often perceived as a detriment to cattle grazing and have occasionally been removed mechanically or sprayed with herbicides. Selected oak savanna upland sites bordering the Loup River Valley, Custer County, Nebraska, were analyzed by the point-centered quarter method during the 2009 growing season. Based on average importance values (IV), the dominant tree species in central Nebraska savannas were bur oak, green ash (*Fraxinus pennsylvanica*), and red cedar (*Juniperus virginiana*), while bur oak, red oak (*Quercus rubra*), and hophornbean (*Ostrya virginiana*) were the dominants on upland sites in eastern Nebraska. All bur oaks recorded in this study were arranged by size class. These data were compared to size class information derived from studies by the first author on upland forest sites in four counties bordering the lower Platte River Valley in eastern Nebraska. Significant size class differences ($P < 0.05$) were found between the bur oak populations of central Nebraska and those in eastern Nebraska. The presence of many smaller trees (size classes < 20 cm dbh) and the lack of the same in central Nebraska indicates that bur oak savannas are reproducing in the eastern Nebraska upland forests but are not being replaced in central Nebraska.

INTRODUCTION

Although the presettlement mixed-grass and tallgrass prairies of the Great Plains are often depicted as an endless "sea of grass," bur oak savannas are features of the prairie that are often overlooked (Figure 1). These scattered populations of bur oak are more common in the hills and valleys of the tallgrass prairie region, but will occasionally extend into the eastern mixed-grass prairie along river valleys or in more mesic, protected canyons. On mixed-grass sites, the oaks must adapt to cyclic drought conditions and prairie fires that occurred as often as every three to five years (Bragg 1985). Most woody plants are destroyed or seriously damaged by fire, but the corky inner bark of the bur oak provides a degree of protection. In the well-known words of Aldo Leopold (1949), "bur oak is the only tree that can stand up to a prairie fire and live."

Most studies of oak savannas have taken place in areas that originally bordered tallgrass prairie, such as the western Iowa Loess Hills, the driftless area of southwestern Wisconsin, and the lower Platte River Valley of eastern Nebraska (Klein and Cottam 1979, Rothenberger 1985, Rozmajzl 1988). This study documents and compares disjunct oak savanna sites occurring in the mixed-grass prairie of Custer County, Nebraska, to upland forest sites bordering the lower Platte River Valley of Nebraska, approximately 226 km (141 mi) to 287 km (178 mi) to the east (Figure 2). Most of the Custer County sites were grazed, whereas grazing was not a factor in the east. Even though all of the eastern Nebraska sites were ungrazed, these areas are subject to other kinds of disturbances, such as expanded cultivation, housing developments, and limestone quarries. Average annual precipitation ranges from 54.6 cm (21.5 in) in Custer County to 79 cm (31.1 in) at Fremont, Nebraska, on the lower Platte.

Figure 1. Trees at the savanna edge encroach on a mixed-grass prairie in central Nebraska.

In eastern Nebraska, the early land surveys (1855-1857) recorded the presence of oak savannas or small clusters of bur oak trees during presettlement times (Rozmajzl 1988, Rothenberger 1989). Most of these trees were gradually removed as

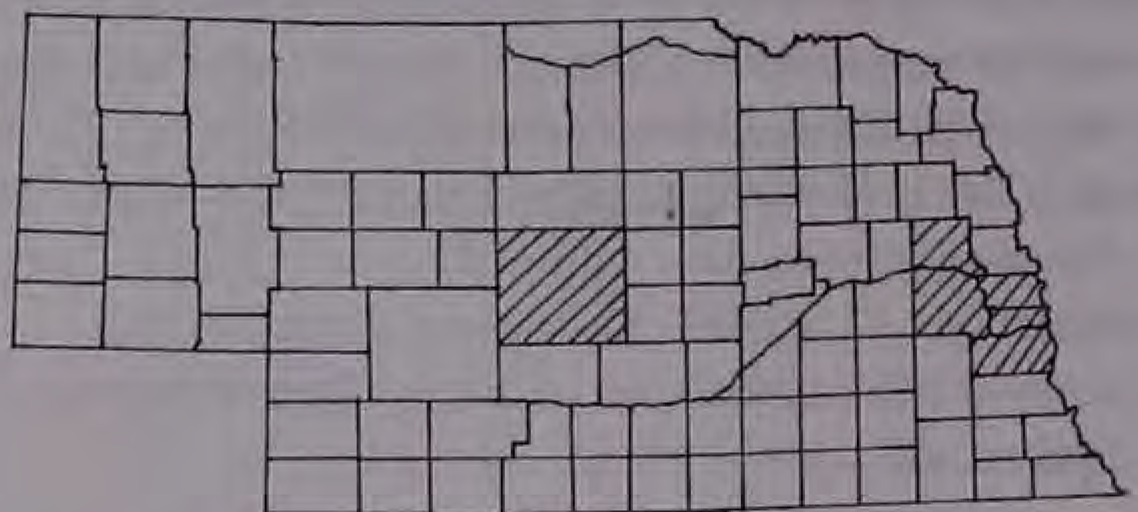

Figure 2. The locations of the two study areas: Custer County in central Nebraska and the five counties bordering the lower Platte River in eastern Nebraska.

these areas were homesteaded. Remaining trees on uplands are mostly associated with a river or stream and provided the opportunity to quantify these species during the original study in 1979-1985. Unfortunately, a number of these sites are very desirable for housing developments or acreages because of the short commuting distance to the Omaha metro area. In the context of savanna preservation/restoration, this study is of importance because data are provided from sites that are geographically separated from the oak-hickory forest region as originally described by Braun (1950). Future research should explore the effects of fire suppression, human activities, and grazing on these areas.

METHODS

Selected savanna sites in the prairies bordering the Loup River Valley, Custer County Nebraska, were sampled during the 2009 growing season by the point-centered quarter method. A total of 25 transects were run within approximately 20 different sites or locations. The nearest tree within each quarter with a diameter at breast height (dbh) > 2.54 cm was measured. Data recorded at each point included tree species, the point to tree center distances, and diameter at breast height (dbh) in cm. For each species, density/relative density, frequency/relative frequency, and dominance/relative dominance were calculated. Importance value (IV) for each tree present at a given site was determined by the sum of relative density + relative frequency + relative dominance (Cottam and Curtis 1956, Hovind and Rieck 1970). Data were gathered by the same method in the eastern Nebraska study, and density, frequency, dominance, and importance values were calculated. This study divided sites into three types: (1) riverbottom forest, (2) transitional sites, and (3) upland sites. The upland sites consisted mostly of forest openings dominated by bur oak and were most representative of true savanna habitat. Therefore, only tree species data from the nine upland sites were used as a comparison to the Custer County study. All tree measurements (dbh in cm) were placed into size classes set at 5 cm intervals between 0 and 100 cm dbh. For example, size class 1 = 0-5 cm dbh, size class 2 = 6-10 cm dbh, size class 3 = 11-15 cm dbh, up to size class #20 = 96 to 100 cm dbh. For each of the two studies, the mean dbh was calculated separately along with the standard error. A student t-test was utilized to establish 95% confidence intervals with the significance set at $P < 0.05$. Nomenclature follows *The Flora of Nebraska* (Kaul et al. 2006)

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RESULTS AND DISCUSSION

In Nebraska, significant variation in geography and precipitation from west to east results in habitats that are increasingly more hospitable for the development of woody vegetation. Therefore, differences in tree species composition between eastern and central Nebraska were expected. The eastern Nebraska sites are dominated by a higher diversity of hardwood species (16) than the central Nebraska sites that consist mostly of bur oak, green ash, American elm, and eastern red cedar (Table 1). Bur oak is the only known oak (*Quercus*) species that can adapt naturally to parts of central and western Nebraska, whereas three species of oak are prominent in the lower 40 km of the Platte River Valley. These are black oak (*Quercus velutina*), red oak, and bur oak. In extreme southeastern Nebraska, six oak species are known in the bluffs and uplands that border the Missouri River. In addition to those previously mentioned, these species are blackjack oak (*Quercus marilandica*), chestnut oak (*Quercus muhlenbergii*), and pin oak (*Quercus palustris*). Other hardwood species exclusive to eastern Nebraska are shagbark hickory (*Carya ovata*), bitternut hickory (*Carya cordiformis*), and hophornbean.

Table 1. Average importance values (IV) for tree species (all sites) sampled during the eastern Nebraska upland forest study and in Custer County oak savannas.

SPECIES	UPLAND FOREST OF EASTERN NEBRASKA AVE. IV	SAVANNAS OF CUSTER COUNTY AVE. IV
Acer negundo (box elder)	2.4	15.1
Amelanchier arborea (Juneberry)	0.7	
Carya cordiformis (bitternut hickory)	23.1	
Celtis occidentalis (hackberry)	16.3	5.3
Cornus drummondii (rough-leaved dogwood)	3.8	
Cornus sericea (red-osier dogwood)	0.4	
Fraxinus pennsylvanica var. subintegerrima (green ash)	15.1	43.6
Gymnocladus dioica (Kentucky coffee-tree)	3.2	
Juglans nigra (black walnut)	8.9	
Juniperus virginiana (red cedar)	17.7	20.7
Morus alba (white mulberry)	0.4	1.5
Morus rubra (red mulberry)	5.6	
Ostrya virginiana (hop hornbean)	33.9	
Populus deltoides (plains cottonwood)	0.6	10.5
Populus alba (white poplar)		5.9
Quercus rubra (red oak)	45.2	
Quercus macrocarpa (bur oak)	67.2	151.8
Quercus velutina (black oak)	21.3	
Robina pseudoacacia (black locust)		8.3
Salix amygdaloides (peach-leaf willow)		6.9
Tilia americana (American linden)	13.1	
Ulmus americana (American elm)	1.7	22.2
Ulmus rubra (slippery elm)	1.7	
Ulmus pumila (Siberian elm)		8.2

The average importance value of bur oak (151.8) in Custer County is much higher than the same value in eastern Nebraska (67.2) where bur oak competes with several upland hardwood tree species (Table 1). This comparison also reflects the status of bur oak as the dominant species in Custer County savannas. In addition to bur oak, the other dominant trees here are green ash, American elm (*Ulmus americana*), and red cedar, respectively. In our study, only 12 tree species were recorded at the Custer County sites compared to 20 total tree species in eastern Nebraska. Based on importance value, the dominant species in eastern Nebraska upland sites are bur oak, red oak, hophornbean, bitternut hickory, and black oak, respectively.

SIZE CLASSES

Size class data comparisons between eastern and central Nebraska produced some surprising results. In eastern Nebraska, the presence of smaller tree size classes and tree seedlings indicated that the populations were reproducing or at least maintaining their population density in these woody plant communities. In central Nebraska, the future of bur oak communities is uncertain. An average dbh size class of 41-45 cm ($\bar{x}$ = 44.4 cm ± 2.63; mean + 95% C.I.) was significantly different from the average size class of 21-25 cm ($\bar{x}$ = 25.0 cm; ± 2.73; mean + 95% C.I.) in eastern Nebraska (Figure 3). These numbers directly reflect the absence of smaller size classes of bur oak trees in Custer County as opposed to the initial perception that the oak trees here are larger in overall diameter. A comparison of tree densities and size classes recorded at both research locations is shown in Figure 4.

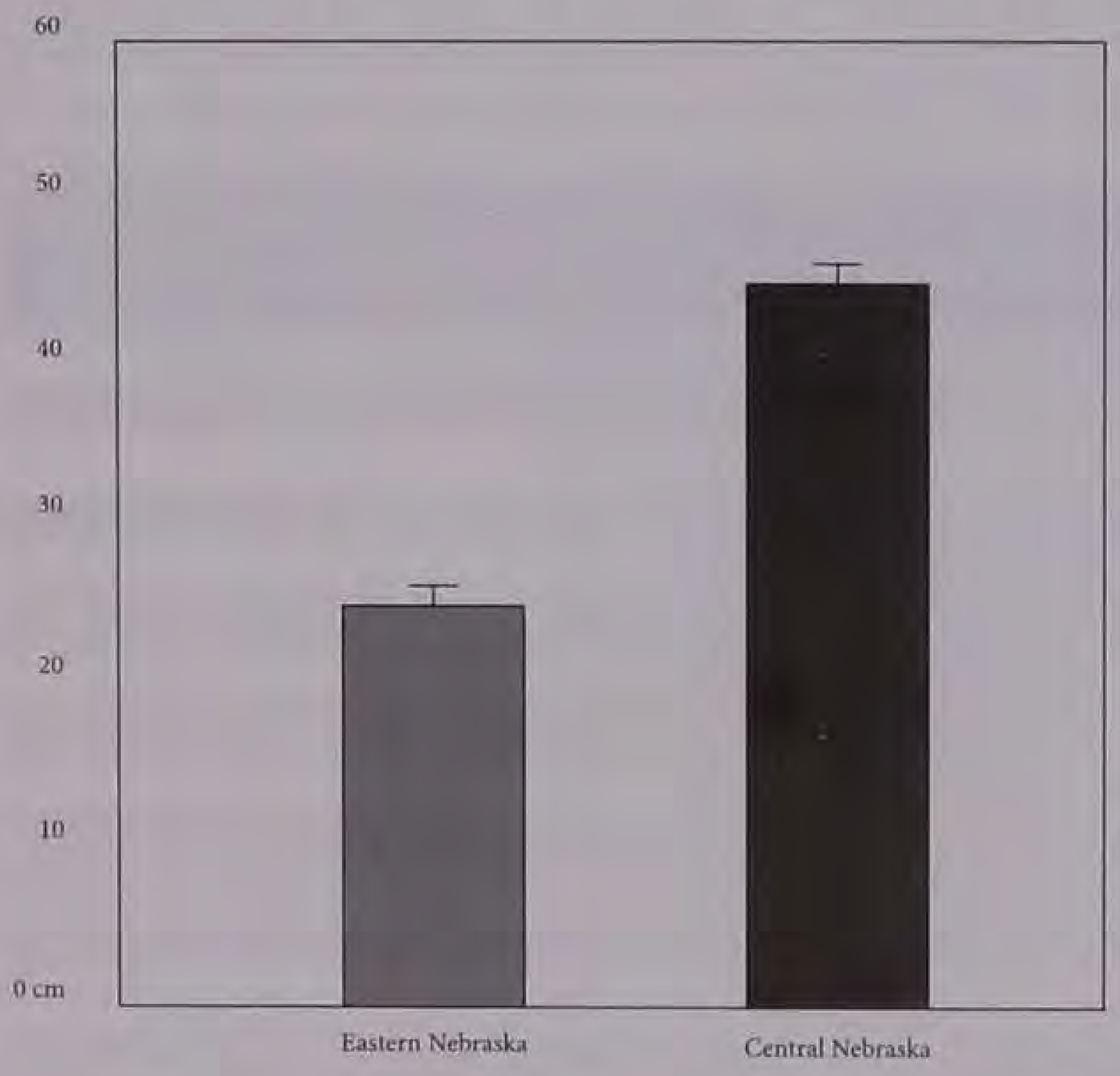

Figure 3. Mean tree diameter at breast height (dbh in cm) and standard errors calculated for bur oaks from eastern Nebraska sites (n = 94) and central Nebraska sites (n = 163).

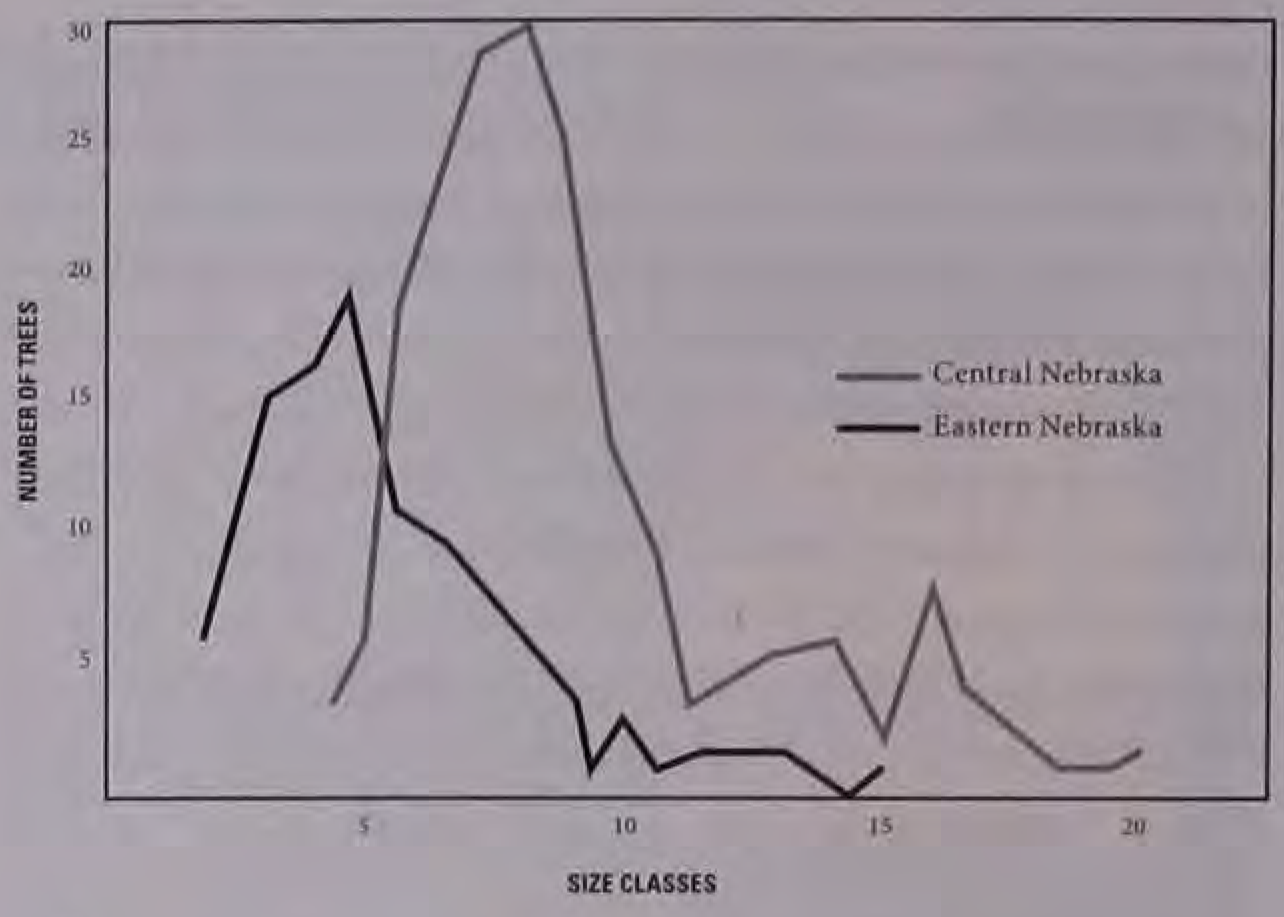

Figure 4. A comparison of the bur oak trees (number of trees per size class) documented in the eastern Nebraska and central Nebraska studies.

On central Nebraska grazing lands, bur oak is considered to be an invasive woody species and is occasionally cut down or sprayed with herbicides for the purpose of "grassland maintenance." Trampling and grazing also have an impact on tree seedlings. The presence of smaller size classes on eastern sites provides some evidence for tree replacement. Our results are consistent with research done by Beightol and Bragg (1993). Their study of an oak savanna in Seward County, Nebraska, demonstrated that bur oak in the smallest size class (< 10 cm dbh) was absent from one of their study transects and low in abundance in the others. This presettlement bur oak forest (Beightol and Bragg 1993) is apparently succeeding to one dominated by American elm, slippery elm (*Ulmus rubra*), and hackberry (*Celtis occidentalis*), just as green ash and American elm are succeeding bur oak on many of our study sites in Custer County. The majority of our study sites are privately owned, which jeopardizes their existence. During presettlement times, these savannas were mostly maintained by fire, which removed competitors, and the absence of human disturbance. For those savannas that are worthy of preservation, it is essential to work with individual landowners in order to achieve some degree of protection. For example, oak savanna preservation in Iowa is occurring at selected state parks and within the Loess Hills, which have been designated as a Natural National Landmark (Prior 2010). However, Nebraska has yet to recognize the unique significance of this habitat type.

PHYSICAL FACTORS

Originally, it was assumed that fruit (acorn) production and the conditions for germination were comparable between these two geographical areas. This may not be the case. The Custer County location is likely to experience severe drought conditions more frequently than is eastern Nebraska, and differences in soil composition (organic matter, soil moisture, micro- and macronutrients) could be critical. Differences in soils properties and geography of the two areas are of importance. The hillsides and canyons of Custer County are formed from deep deposits (up to ~30 m thick) of Peoria loess, the most extensive parent material in this area (Wilson et al. 1982). The eastern sites along the lower Platte River Valley occur on shallow loess deposits ranging in thickness from 0 to 5m underlain by Pennsylvanian limestones and shales (Burchett et al. 1975). These differences in parent material and soil formation factors were not measured in this study, but their effects on the development of woody vegetation provide potential for future research.

CONCLUSIONS

In the eastern Nebraska study, the abundance of bur oak in smaller size classes resulted in a much lower average dbh ($\bar{x}$ = 25.0 cm) than in the central Nebraska study ($\bar{x}$ = 44.4 cm). Average importance values for bur oak of 67.2 (eastern Nebraska) compared to 151.8 (central Nebraska) also reflected these size class differences. Although the sites in central Nebraska are succeeding more rapidly to replacement species, such as green ash and American elm, almost all post-settlement savannas in Nebraska are jeopardized by the lack of fire and the presence of human activities.

Even with disjunct oak savannas extending from eastern Nebraska into the plains, these vegetation types are not as common or as well developed as are the more typical, established savannas to the east. Some of the best examples of oak savannas in the north central region are those of western Iowa and in the driftless area of southeastern Minnesota and southwestern Wisconsin (Klein and Cottam 1979). Preservation of representative tracts of oak savanna in western Iowa has become a priority in recent years, beginning in 1986 with the designation by the U.S. Department of the Interior of ~ 4050 ha (10,000 ac) of this land as a National Natural Landmark (Prior 2010). These unique "islands" of woody vegetation in the prairie provide habitat for numerous species that are not typically found in grasslands. The preservation of the few extant bur oak savannas that extend into the drier central plains should also be a priority.

ACKNOWLEDGMENTS

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TWENTY-FIVE YEARS OF MONITORING THE WESTERN PRAIRIE FRINGED ORCHID (*Platanthera* Sheviak & Bowles) IN MINNESOTA

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Abstract: We summarize a nested monitoring protocol for western prairie fringed orchid (*Platanthera praeclara* Sheviak & Bowles) and present illustrative case examples of results that highlight benefits of complementary levels of monitoring. Flowering counts at full anthesis are the range-wide data standard for tracking the species' status. Minnesota harbors over 40% of plants in the United States. We annually count flowering plants at over 80% of Minnesota populations, with consistent counts at 20 sites for 17 years. Statewide, flowering plants peaked at over 10,000 in 1994. Average counts from 2006 to 2009 are 15% of counts from the four high years between 1993 and 1996. In 1985 and 1991 the Minnesota DNR established demographic transects across the species' latitudinal gradient to assess recruitment, mortality, dormancy, and age to first flowering. In a preliminary analysis of data from 1986 to 1994, up to 12% of plants in demographic plots were dormant each year, with over 30% experiencing dormancy of one to three years. Across all years at two Polk County sites, average life spans were 4-8 years, with individual plants surviving 22-26 years with episodes of dormancy. Earliest documented flowering of putative seedlings at two northwestern Minnesota populations is in the sixth aboveground year. Phenological observations at two sites for five and seven years indicate that plants can emerge as early as the first two weeks of April and can become senescent before normal monitoring dates in dry years, and that up to 38% of reproductive effort can be lost before seed dispersal.

Key Words/ Search Terms: Western prairie fringed orchid, *Platanthera praeclara*, monitoring, demography

INTRODUCTION

Western prairie fringed orchid (*Platanthera praeclara* Sheviak & Bowles), hereafter often referred to as "the orchid," is a showy prairie orchid with an open raceme of up to 24 white flowers, up to 2.5 cm (1 in) wide and 3.0 cm (1.2 in) tall, with a deeply fringed, three-lobed lip and slender nectar spurs up to 55 mm (2.1 in) in length (Sheviak 2002). The lip, or lower petal, of each flower is deeply three-lobed and fringed. The single, smooth stem is up to 85 cm (34 in) tall (Smith 1993). Flowering plants have three or more smooth, elongate leaves. Established nonflowering plants have 1-3 leaves of similar appearance. Leaves of seedlings exhibit underground protocorms (Sharma 2002, Sharma et al. 2003) and are generally less than 1 cm (0.4 in) wide and up to 18 cm (7 in) long (MNDNR unpublished data, Quintana-Ascencio and Menges 2005).

Western prairie fringed orchid is extant in Nebraska, Kansas, Missouri, Iowa, Minnesota, North Dakota, and Manitoba (US Fish and Wildlife Service 2009). It appears to have been recently extirpated in Oklahoma and was historically documented in South Dakota and Wyoming. The species is listed as threatened in the United States (USFWS 1989), endangered in Canada (Environment Canada 2006), endangered in Missouri, and threatened in Iowa, Minnesota, and Nebraska (USFWS 2009). The species was added to the IUCN Red List in 2008 (Goedeke et al 2008). The U.S. federal Recovery Plan (US Fish and Wildlife Service 1996) recognizes 104 extant populations of the western prairie fringed orchid, 58 of them in Minnesota. An updated summary of the species' distribution and numbers in the United States is included in the five-year review of the species' recovery status (US Fish and Wildlife Service 2009).

Minnesota populations historically occurred in all regions of the state that supported mesic prairie. The Recovery Plan (US Fish and Wildlife Service 1996) establishes goals by Ecoregion (Bailey et al. 1994). Minnesota populations occur in Ecoregion 223Na (Aspen Parkland), Ecoregion 222M (Eastern Broadleaf Forest), Ecoregion 251B (North Central Glaciated Plains), and Ecoregion 251A (Red River Valley Prairies and Aspen Parkland). Recognition of Ecoregion 223Na (Aspen Parkland) in the Lake Agassiz beach region of northwestern Minnesota postdates the recovery plan. Minnesota's largest populations lie at the southern end of this ecoregion. As of 2010, the Minnesota DNR tracks 91 Element Occurrences, three of them historical, and three not confirmed since 1990 (Natural Heritage Information System 2010).

Minnesota's orchid populations range from latitude 43° 41' in Rock County near the Iowa and South Dakota borders to 48° 44' in Kittson County, near the Manitoba border. Extant occurrences are present in nine counties (Natural Heritage Information System 2010). Some occurrences comprise numerous subpopulations. Forty-six source features (subpopulations) are part of the single large Polk County Interbeach metapopulation (Ecoregion 251A). Three sentinel monitoring sites are located in this metapopulation.

Sites in southwestern and northern Minnesota vary in physiographic setting. Monitoring sites in Ecoregion 253Na are wet to wet-mesic prairies, on level to gently sloping, loamy to clayey tills over sand. Many occur in subirrigated interbeach swales associated with the beach ridges of Glacial Lake Agassiz. Orchid sites in Ecoregion 253Na exhibit little relief and are subject to spring standing water, sometimes into early June. Northwestern Minnesota *P. praeclara* habitat is most similar to sites in the Gardenton-Vita area of southern Manitoba (Collicutt 1992, MNDNR unpublished data). Minnesota's sites all lack the hill-and-swale topography of the Sheyenne National Grassland, North Dakota, where populations tend to concentrate in distinct swales, many of which are intermittently paludified for the entire length of the growing season (Sieg and Wolken 1999).

Minnesota's extant populations in Ecoregion 251B are in the far southwestern part of the state on the Prairie Coteau. They are characterized by mesic soils shallowly overlying Sioux Quartzite bedrock. Standing water is an infrequent occurrence. It is likely that the necessary moisture for these populations is held in surface soils by the impervious underlying rock, suggesting that these sites are less well buffered from effects of drought than those in the north (Willson and Akyuz 2010).

Northwestern Minnesota sites are characterized by the presence of patches of willows (*Salix* spp.), bog birch (*Betula pumila*), and shrubby cinquefoil (*Dasiphora fruticosa*). It is unclear whether this shubby cover serves as a safe site for orchids, or whether the slightly moister conditions that support the shrubs maintain orchids through drier years, but observational evidence suggests that the presence of flowering orchids in northwestern Minnesota in dry years may be correlated with proximity to shrubs. A number of species present in northwestern sites have not been documented in association with the orchid in southwestern Minnesota. Among these are tufted hairgrass (*Deschampsia cespitosa*), marsh bellflower (*Campanula aparinoides*), prairie loosestrife (*Lysimachia quadriflora*), and seaside arrowgrass (*Triglochin maritima*). Conversely, several species growing in association with *P. praeclara* in the southwest have not been documented to co-occur directly with the orchid in the north. Among these species at Blue Mounds State Park are leadplant (*Amorpha canescens*), smooth aster (*Symphyotrichum laeve*), silky aster (*Symphyotrichum sericeum*), and birds-foot coreopsis (*Coreopsis palmata*). The population at Pipestone National Monument is dominated by little bluestem (*Schizachryrium scoparium*) (Willson and Akyuz 2010). The co-occurrence of these species with the orchid illustrates the drier overall habitat conditions in southwestern Minnesota.

Nearly all Minnesota sites have a history of grazing during the late nineteenth century and first seven decades of the twentieth. Only three sites are grazed today. The majority of sites are actively managed in Preserves and Wildlife Management Areas. Prescribed fire is the traditional tool of choice for prairie management at Minnesota orchid sites, used to control woody invasion in the north and infestations of smooth brome in the south. Smooth brome (*Bromus inermis*) is most effectively controlled if fires are timed to coincide with the period when the inflorescence is wrapped in or just emerging from the elongating leaf sheath (Willson 1992). Burns do not always occur at optimal management dates because of logistical considerations related to the availability of personnel when conditions are within prescription. Minnesota's two most northerly *P. praeclara* sites have not experienced any management since discovery of the orchids in the early 1990s and suffer from serious aspen invasion. None of the sites is managed exclusively for the orchids, but an ongoing experimental management study of orchid response to fire and haying at TNC's Pembina Trail Preserve dictates management of the state's largest population (Kiefer 2004).

The Minnesota Department of Natural Resources initiated monitoring of *P. praeclara* by establishing demographic transects in 1985. The purpose of this monitoring is to investigate the species' life history. At the time of its inception the study was unique because it included addition of newly observed plants on each subsequent year of observation.

Previously published studies of *P. praeclara* concentrate on the population in North Dakota's Sheyenne National Grassland (USFWS 2009). Sieg and Bjugstad (1994) report an overall 60% decline in 16 transects of marked flowering individuals for the five years between 1987 and 1991. Sieg and King (1995) followed plants in all life stages in ten-meter-wide belt transects from 1990 to 1994. On the basis of the low number of returning individuals, they concluded that the species is short-lived, with a half-life approximating 1-3 years, or that it exhibits periods of dormancy longer than could be detected in their study. Survivorship from year one to year two ranged from 14% to 73%, with only one of 523 plants reappearing each of the five years of the 1990-1994 study. Plants moved back and forth among life states, but once absent, the probability of remaining absent was high (82%-100%).

Willson and Akyuz (2010) assessed return rates of 30 marked flowering plants at Pipestone National Monument between 1995 and 2004 and compared their results with the fate of 30 flowering plants in the 1987 marked cohort at the Sheyenne National Grassland. Of marked Pipestone plants 83% returned the second year of observation, with 13% of plants present for only one year and 23% for only two years. Two plants survived through the entire length of the study. Forty-three percent of plants exhibited at least one episode of dormancy, with a maximum length of a single dormancy episode of three years (Willson and Akyuz 2010, appendix 1).

MATERIALS AND METHODS

The Minnesota Department of Natural Resources uses the nested orchid monitoring protocol depicted in Figure 1, with decreasing numbers of sites as intensity of monitoring becomes greater. All levels of monitoring are implemented at least at a minimum of one site in the north and one site in the south.

Level 1 monitoring tracks presence and absence in BIOTICS (Natural Heritage Information System 2010). For the purpose of our database, we continue to treat populations that have not been seen for a series of years as extant unless all potential habitat has been destroyed. The BIOTICS mapping standard of 1 kilometer is applied as the standard separation distance between element occurrences, but individual subpopulations and annual counts are tracked as source features, allowing users to cross reference consolidated occurrences in the BIOTICS system to observations treated as separate populations in the federal recovery plan.

Level 2 monitoring is an annual census of flowering plants conducted in collaboration with the Minnesota Chapter of The Nature Conservancy (TNC). Together, we count upwards of 80% of Minnesota plants and populations each year. Twenty sentinel populations have been continuously counted for the last 17 years. The Minnesota

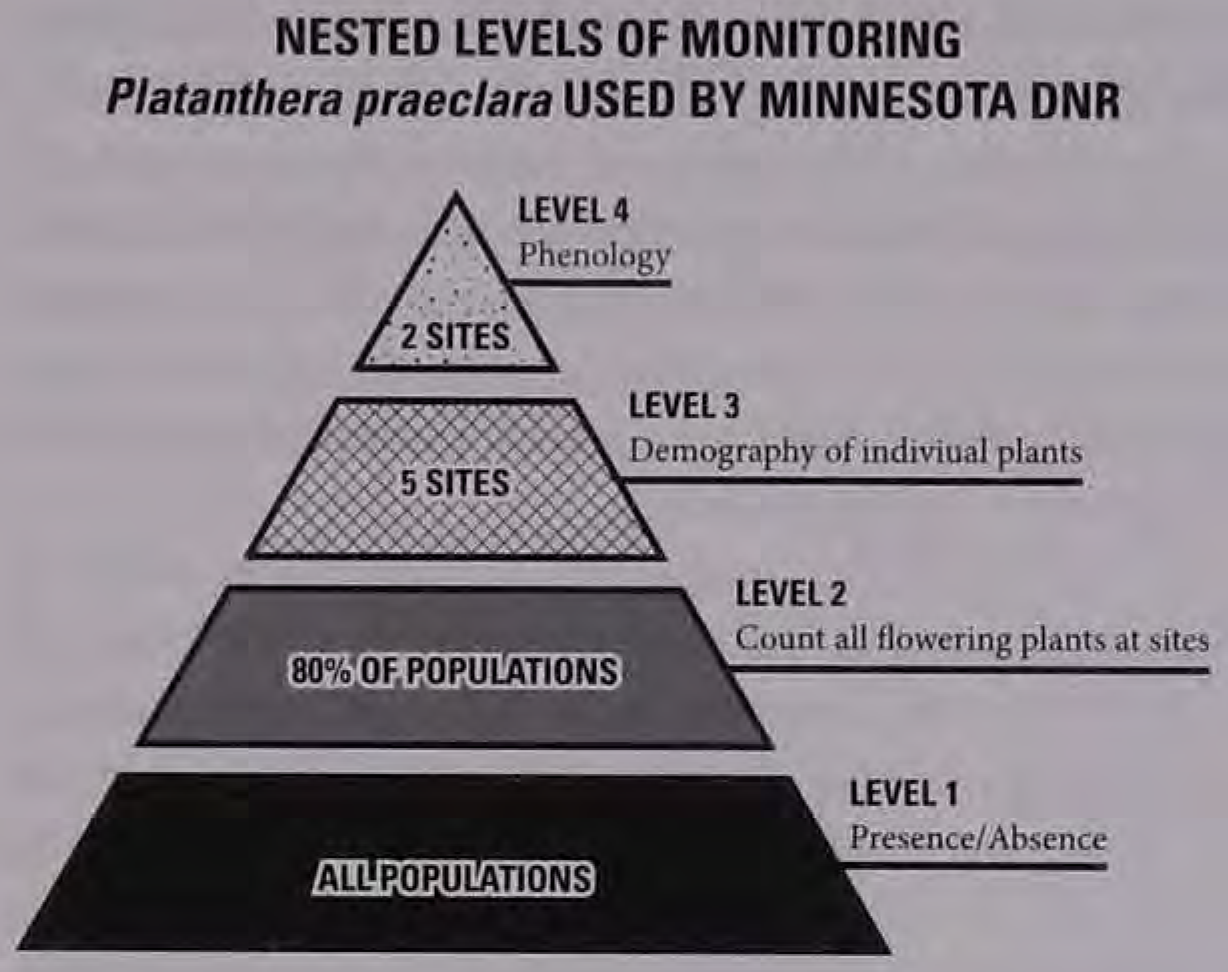

Figure 1. The intensity of monitoring increases upwards in the diagram, whereas geographic dispersion increases downwards.

DNR uses teams of volunteer citizen scientists to systematically sweep orchid habitat at each site, counting all individuals in bud, flower, or fruit. We monitor both public and private lands where we can obtain landowner permission. DNR crews monitor in Rock, Polk, Pennington, and Kittson Counties. TNC seasonal staff monitors sites in Clay and Norman Counties and TNC lands in Polk County. We also maintain data from annual counts by the National Park Service at Pipestone National Monument and the U.S. Fish and Wildlife Service at Glacial Ridge National Wildlife Refuge. Three substantive privately owned sites are not counted as part of level 2 monitoring out of respect for landowner rights.

Level 3 tracks the fate of individuals in monitoring transects at sites across the latitudinal gradient from southwestern to northwestern Minnesota (Figure 2). In 1985, the Minnesota Natural Heritage Program established permanent demographic monitoring plots at Blue Mounds State Park in southwestern Minnesota and at three Polk County interbeach populations: Burnham WMA, Pembina Trail Preserve, and Crookston Prairie SNA. In 1995, DNR Wildlife Area managers added sites in Pennington and Kittson Counties, extending monitoring to the northernmost population in the United States (Sather 1997). The Minnesota Chapter of The Nature Conservancy and the United States Geological Survey monitor two additional sites by slightly different methods, but collecting the same basic demographic data.

As was the case in the original monitoring at Sheyenne National Grasslands (Sieg and Bjugstad 1994, Sieg and King 1995), transect placement and length were arbitrarily established to lie in areas that are permanently protected and maximize sample size. Because the objective of the study was to learn more about the life history of the species, not to characterize populations, we treat the transects as our sampling universe. The proportion of the entire population represented by the transect(s) in given sites varies. As an indication of the range of likelihood that trends in the plots reflect those in the occurrences, it is instructive to compare the percent of sitewide all-time high counts with all-time high counts of flowering plants in the plots (Table 1).

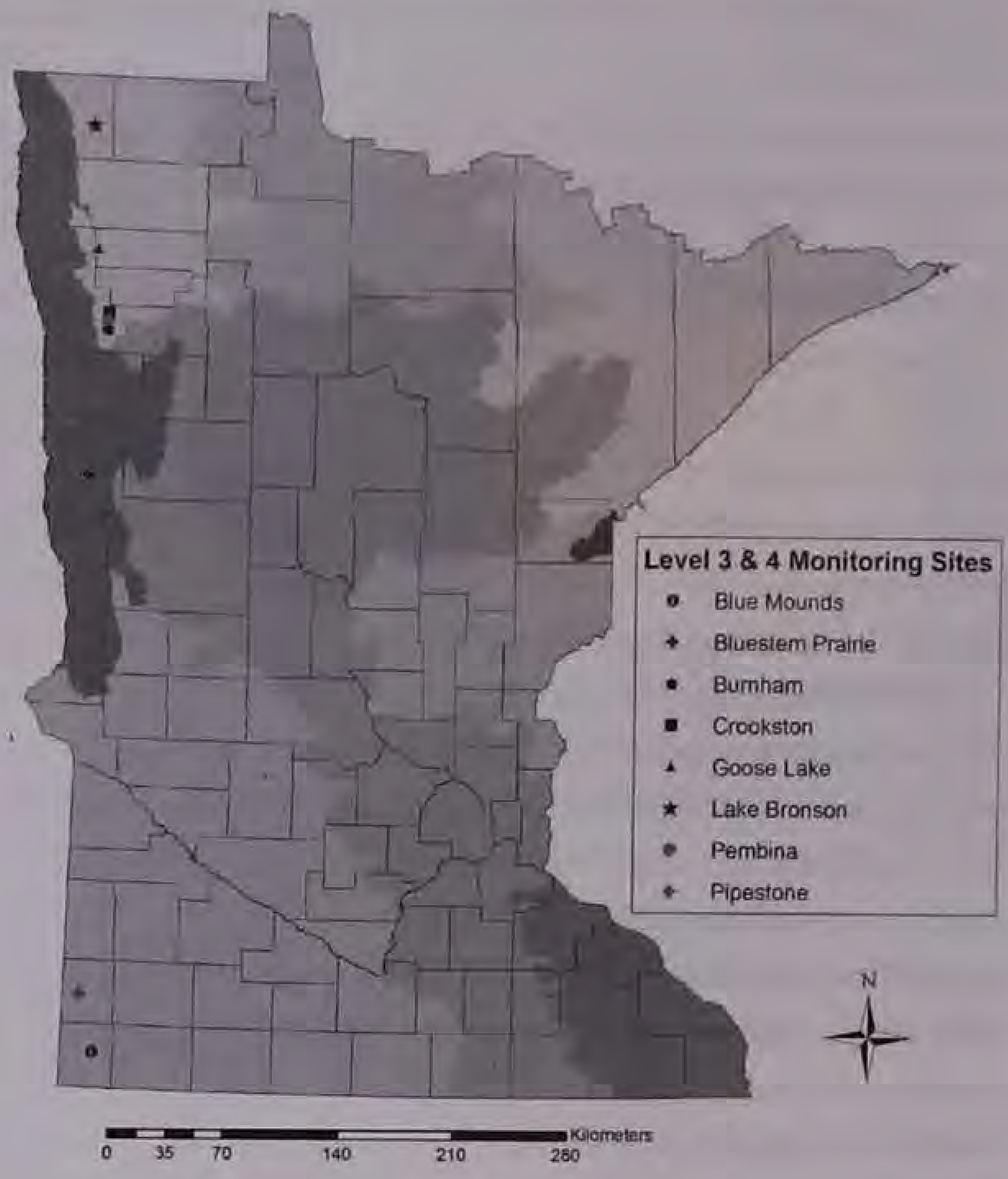

Figure 2. Minnesota's demographic monitoring sites span the species' latitudinal range in the state from Blue Mounds State Park in Ecoregion 251B near the Iowa and South Dakota border to Lake Bronson State Scientific and Natural Area in Ecoregion 223Na near the Manitoba border. Populations noted with a plus sign are monitored by USGS (Pipestone National Monument) and TNC (Bluestem Prairie, Ecoregion 251a).

Demographic monitoring infrastructure and procedures are similar to those described by Smith (1986) for prairie bush clover (*Lespedeza leptostachya*). Plot corners are permanently marked in the field by a 6-8-inch length of three-quarter-inch conduit driven into the ground. A portable 2 x 2 meter sampling frame calibrated at centimeter intervals is anchored at these corners and used to determine x/y coordinates of each observed plant to the nearest cm. Coordinates of all returning and newly observed plants are recorded each year and locations of all absent plants are checked for signs of emergence and browse or early senescence.

Field data are recorded on Mylar field forms that enable us to view a map of the location of plants in previous years. Plants are assigned a plot number and a plant number and matched to their identity by x/y coordinates and relative geographic position within the plot. For purposes of identifying whether an individual is new or a returning plant, plant numbers are carried forward from year to year.

For each plant, we record the number of leaves and height to the tallest part of the plant. Plants are defined as vegetative if they exhibit 1-3 leaves and, in the latter case, no evidence of attempt to flower. Plants are defined as "flowering" if they exhibit bud, flower, or pod development at the date of demographic sampling. The number of reproductive units (flowers, buds, or pods) is recorded for each flowering plant. Unusual traits such as aborted or arrested flowers or four-leaf plants with no raceme are reported in a comments column on field data sheets. At Polk County Inter-

Table 1. Summary of MN DNR demographic monitoring sites. Table 1 summarizes attributes of the monitoring transects. Highest counts refer to the entire population in which the monitoring transect occurs. Total plants ever monitored refers only to plants within the demographic transect.

SITE	DURATION OF MONITORING	NUMBER OF PLOTS	PROPORTION OF HIGHEST FLOWERING COUNT MONITORED	TOTAL PLANTS EVER MONITORED	LATITUDE (DM, NAD 83)	LONGITUDE (DM, NAD 83)
Blue Mounds State Park	1985-2010	8 in one transect	13	291	43° 42'	96°12'
Burnham Wildlife Management Area	1985-2010	11 in one transect	7	217	47° 38'	96°21'
Pembina Trail Triangle	1985-2006	17 in one transect	1	87	47 40'	96°21'
Crookston Prairie SNA	1985-2010	9 in one transect	20	151	47° 43'	96°21'
Goose Lake State Trust	1994-2009	10 in one transect	26	63	48 05'	96°28'
Lake Bronson SNA	1994-2010	20 in 2 transects	6	90 +27	48° 44'	96°33'

beach monitoring sites, we have recorded to the nearest mm the width of all single-leaved plants less than 1 cm wide in their first observed year since 1996 (Sather 1997).

Level 4 monitoring comprises phenological observations that frame the period when census and demographic monitoring occur. When it is logistically feasible, we engage citizen scientists to observe the condition of emerging orchids and development of pods in the Polk County Interbeach and Blue Mounds State Park populations. Locating emerging plants requires a permanent frame of reference. At sites that are occasionally burned, plot markers are essential. To expedite monitoring and assure implementation, we use the most easily accessible populations that provide sufficient infrastructure. Spring phenological observations in Polk County are facilitated by permanent plots that are part of an ongoing experimental management study at Pembina Trail Preserve (Kiefer 2004), where permanently marked 100-square-meter blocks allow us to monitor 40 plants, 10 in each of 4 treatments. At Blue Mounds State Park, we use the demographic monitoring transect to locate emerging plants. The number of plants observed in a given year depends on the number that emerge in the transect. Observers have tracked emergence and pre-anthesis development at Pembina in 2003-2005, 2007-2008, and 2010. At Blue Mounds State Park, our observer tracked spring development from 2001 through 2007. On each visit, phenology observers record the height of all visible orchids, a qualitative assessment of bud development prior to anthesis, and number of reproductive parts (buds, flowers, or fruits) observed on each date. At Blue Mounds State Park, we have also recorded the height and developmental stage of smooth brome prior to the species' flowering. Tracking pod development is easier than observing emergence, because flowering plants marked with flags indicating the number of flowers at anthesis can be tracked through the season.

Census and demographic data are maintained in Excel spreadsheets. We use separate rows for sites, and columns for years, in the census spreadsheet. We maintain each demographic monitoring site in a separate spreadsheet with rows for individual plants and series of repeating columns for each variable in each monitoring year. Phenological data are similarly arrayed across a single season, with separate spreadsheets for each site and season. Unless they have been reported in phenological observations, any plants that emerge and senesce before demographic monitoring are reported as absent in the demographic data set for a given year. Missing data are noted in comments fields in all spreadsheets, allowing us to use blank cells for negative data. This practice enables us easily to use summary functions in Excel that would otherwise calculate zeros into counts and averages. Excel spreadsheets enable us to generate summary graphs of population structure at monitoring sites over a period of years, to detect trends and develop hypotheses for statistical analysis and to compare trends across years or ecoregions; they are also exportable to commonly used statistical packages.

Demographic data enable us to determine rates of fecundity, dormancy, mortality, and age at first flowering. By direct visual examination of the spreadsheet, we assign a lifestate to each plant based on a combination of presence, morphology, and aboveground history. Life states are defined morphologically because the actual chronological age of most individuals in a data set is usually not known. Previously observed plants not seen in a given year are treated as dormant if they reappear in subsequent years. Quintana-Ascencio and Menges (2004, Quintana-Ascensio et al. 2005) conducted preliminary analyses of the first ten years' of MNDNR demographic data from eight transects at seven sites (including one site monitored by The Nature Conservancy). They broke plants into categories that reflected aboveground appearance in any given year: dead or dormant, alive, not yet sampled, and missing (a misinterpretation of dead or dormant data). They used binary logistic regression to compare populations for alive status in three years. Since then, we have annually summarized data for the U.S. Fish and Wildlife Service using summary statistics available in Excel.

Nomenclature for herbaceous species in this paper follows Flora of North America (1993+) for published volumes and Gleason and Cronquist (1991) for taxa not yet published in Flora of North America. For woody species, nomenclature follows Smith (2008).

RESULTS

STATEWIDE CENSUS

Statewide, flowering plants peaked at over 10,000 in 1994. Average counts from 2006 to 2009 are 15% of counts from the four high years between 1993 and 1996. Despite the apparent crash in the late 1990s, 2010 counts exceed the 18-year mean in all three ecoregions. All populations reached high numbers in the early 1990s, but since that time high and low counts have not been synchronous between ecoregions. Behavior of individual populations may be a response to local hydrology or site management. Figure 3 depicts trends in the number of flowering plants by ecoregion for 20 consistently counted Minnesota populations between 1993 and 2010.

LIFE-STATE TRANSITIONS AND PUTATIVE SEEDLINGS

Five lifestates are easily recognized in the field: straps (single-leaved plants with a width less than a centimeter and typical lengths no longer than 18 cm in their first observed year), single-leaved vegetative plants, two-leaved vegetative plants, three-leaved plants exhibiting no sign of inflorescence development, and reproductive plants (more than three leaves). On rare occasions, plants with four leaves and no inflorescence development have been observed. These are lumped into the mature nonflowering three-leaf lifestate. Sieg and King (1995) report multiple ramets from one genet. Our field observations suggest only two putative instances of this phenomenon. We treated these individuals as separate plants in our database.

TOTAL NUMBER OF FLOWERING *P. praeclara* PLANTS COUNTED AT 20 SITES FOR 18 YEARS

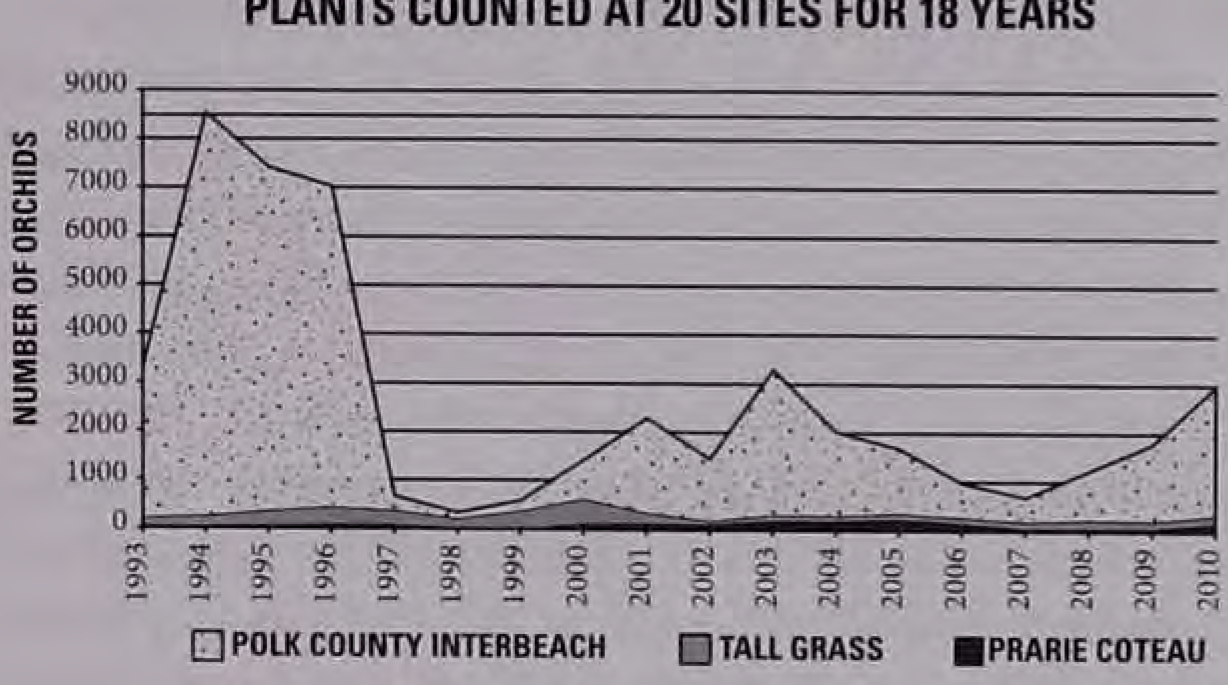

Figure 3. Census results by Ecoregion.

One- to two-leaf transitions and reversions are very common among established plants. One- to three-leaf transitions are infrequent. Most three-leaved plants move back and forth between two leaves, three leaves, and flowering. Vegetative plants seldom exhibit more than three leaves unless they are aborted plants that would have flowered. Plants can move back and forth between one or two leaves and flowering without an intervening year in a higher stage.

We treat strap-leaved plants as putative seedlings because upon excavation they exhibit a protocorm and they are morphologically compatible with known leaf-bearing seedlings raised in vitro (Sharma 2002, Sharma et al 2003). This interpretation makes no assumptions about the duration of underground development prior to first observation. In analyses of our first ten years' data, Quintana-Ascencio and Menges (2004) found that the relationship between width and height for strap-leaved plants at three Polk county sites is positive and linear (r^2=0.34). Height varied significantly among years (ANOVA, $P < 0.05$) but the widths did not.

In the 14 years since we began differentiating them from other single-leaved plants at Burnham and Crookston, straps emerged in half the years at both sites. A total of 41 straps emerged at Burnham and 31 at Crookston. In both cases, half-lives of strap cohorts ranged from one to three years. Good recruitment years were not synchronized between the two sites. Whereas 13 straps were recruited into the Crookston population in 1998 with none in 2002, Burnham's highest recruitment year (16 plants) was in 2002.

Straps are the only lifestate from which plants move unidirectionally. They move from narrow to wider single-leaved plants. Maximum survivorship of straps at Crookston was eleven years and at Burnham nine. Figure 4 depicts the fate of the 1997 cohort of straps at Crookston Prairie.

FATE OF THE 1997 STRAP COHORT AT CROOKSTON PRAIRIE

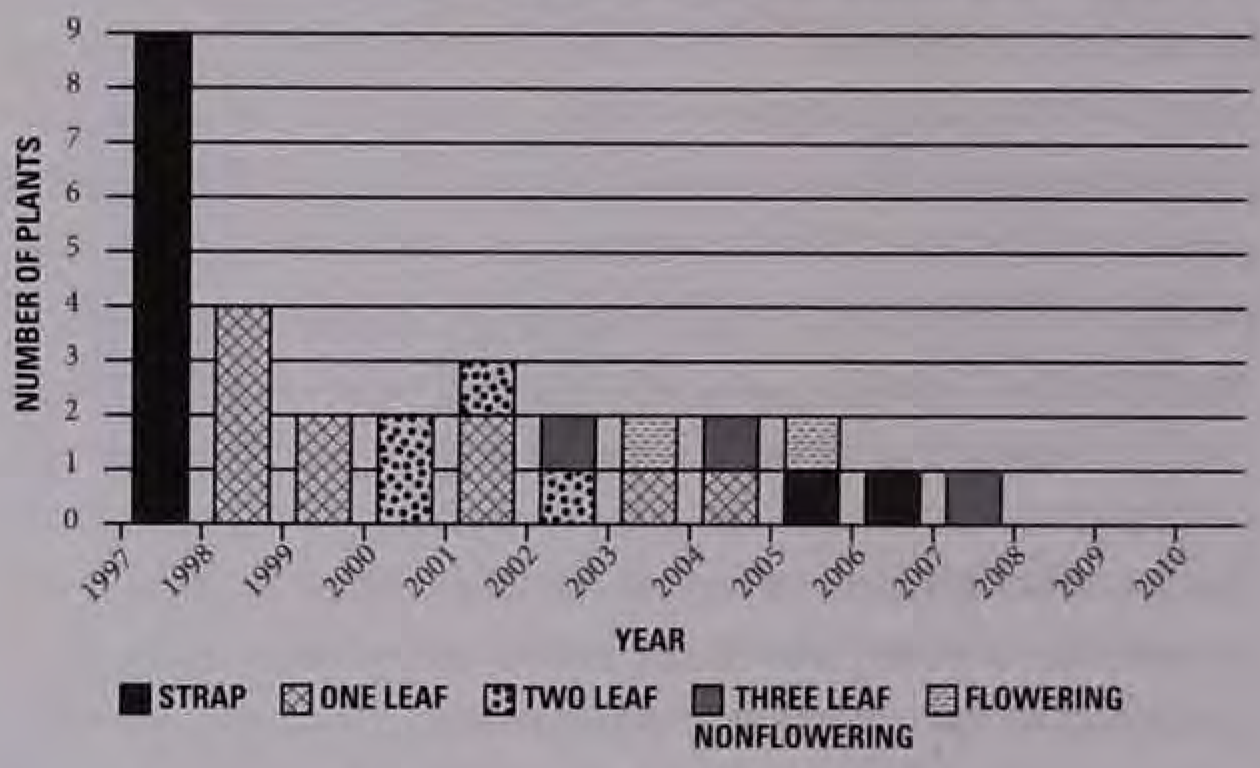

Figure 4. The subsequent lifestates of nine plants that emerged as straps at Crookston Prairie in 1997. The higher number of observed plants in 2001 than in the two preceding years reflects dormancy.

All returning members of the 1997 cohort passed through a larger single-leaved stage before further development. One 1997 Crookston recruit flowered twice (2003 and 2005). Two members of the 1998 cohort also flowered in 2005. Of 72 strap plants that emerged at the two sites in successive cohorts since we first began to differentiate them in 1996, only these three individuals attained flowering stage. None of the straps that emerged at Burnham ever flowered. Survivorship of strap plants is low. At both sites approximately 35% of ever-emerged straps failed to return for a second aboveground year.

DORMANCY AND SURVIVORSHIP IN ESTABLISHED PLANTS

Dormancy is typically defined as the failure of the rootstalk of an herbaceous perennial to produce aboveground parts in a given year (Shefferson et al 2005, Lesica and Steele 1994). In a preliminary analysis of our first ten years' data, Quintana-Ascencio et al. (2004)

report dormancy that lasted as long as eight years, with more than half of all dormancy episodes as short as one year. Rates of dormancy can be quite high. Across the full 25 years of the study at Burnham WMA, 57% of plants that ever flowered and 70% of plants that lived more than three years but never flowered exhibited at least one episode of dormancy.

Survivorship differs between lifestates. Preliminary analysis of our first 20 years' data suggests that flowering plants have higher survivorship and flower more frequently than plants that have never flowered (Quintana-Ascencio and Menges 2005). Strap-leaved plants at Burnham and Crookston seldom survived more than 3 years and persisted no more than 10 years.

Table 2 summarizes survivorship of flowering and nonflowering plants at Burnham WMA. At this site, a total of 224 plants were observed at least once over the 25-year period, with a high of 84 plants observed in a single year. Sixty-five of these plants (26%) flowered at least once, whereas the numbers of nonflowering plants observed for only one year (80 plants) and more than one year (79 plants) were nearly comparable. Average survivorship across all 224 plants was 4.2 years. Plants that flowered tended to persist longer than those that did not. Average life of flowering plants was 9.6 years, whereas that of nonflowering plants was 6.6 years.

Table 2. Survivorship of all plants that were ever observed at Burnham WMA, including both straps and established plants in the nonflowering category.

	FLOWERING PLANTS (N=65)	NON-FLOWERING PLANTS (N=159)
AVERAGE LIFE	9.6 years	6.6
AVERAGE LENGTH OF DORMANCY	1.6 years	2.2 years
AVERAGE NUMBER DORMANCY EPISODES	1.9	1.4

Although flowering plants exhibited a higher average number of dormancy episodes (1.9) than nonflowering plants (1.4), the duration of dormancy was higher (2.2 years) for nonflowering plants than for flowering plants (1.6 years). One flowering individual that experienced four single-year dormancy episodes survived for 25 years, whereas the maximum life of a flowering individual with no dormancy was 14 years. Maximum survivorship for nonflowering plants was 17 years with dormancy.

PHENOLOGY

We initiated phenological monitoring in response to observations of arrested plants and years when no flowering plants appeared in our level 2 and level 3 monitoring. Our objective was to determine whether pre-anthesis growth coincides with the timing of late frosts or fires that might intercept floral development.

Because we were initially unaware of how early the plants come up, in all but two instances the first date of reported emergence coincides with the observer's first date of observation. Earliest dates of emergence at Pembina Trail were April 1, 2010, and April 13, 2005. In both cases orchids emerged in standing water. In 2004, all plants had emerged before the first observation on June 4. Early emergence in 2010 was accompanied by flowering the last week in June, whereas typical Minnesota flowering dates are between the July 4 and the third week of the month.

Standing water in the spring is typical in Polk County Interbeach sites, making it difficult to observe emerging plants without knowledge of their exact location. The late first date of observation in 2004 was the result of bureaucratic constraints, not environmental conditions.

ATTRITION OF REPRODUCTIVE EFFORT

Analysis of phenological data from Blue Mounds State Park is in process. At this site we have evidence that orchids can emerge and become senescent before the normal date of annual census and demographic monitoring. In addition to early senescence that appears to be a response to summer drought, post-anthesis observations at several sites suggest that mid-August calculations of fecundity may be premature because of other damaging events that occur between pod formation and dehiscence.

In 2006, we counted 252 flowers on 34 plants at peak anthesis at Crookston Prairie SNA and revisited them on August 21 and September 8 to determine the rate of fruit set and timing of dehiscence. Sixty-six percent of flowers formed capsules, but an early frost the last week of August wilted 63% of the capsules, only one of which went on to dehisce. By early September only 30 fruits (16% of the

Table 3. Emergence dates for six non-consecutive years of observation at Pembina Trail Preserve in northwestern Minnesota.

YEAR	FIRST DATE OF OBSERVATION	FIRST DATE ORCHIDS WERE OBSERVED	% EMERGED	AVERAGE HEIGHT IN *CM* ON FIRST DATE OF DOCUMENTED EMERGENCE	% OF PLANTS THAT LATER FLOWERED PRESENT ON FIRST EMERGENCE DATE	DATE OF 100% EMERGENCE
2003	23-Apr	26-May	26%	10	0% (of 14)	17-May
2004	4-Jun	4-Jun	100%	11	100% (of 10)	31-May
2005	13-Apr	13-Apr	40%	1.9	50% (of 12)	17-May
2007	26-Apr	26-Apr	25%	1.1	0% (of 5)	12-Jun
2008	17-May	17-May	40%	3.7	50% (of 2)	2-Jun
2010	1-Apr	1-Apr	28%	1.2	25% (of 12)	28-May

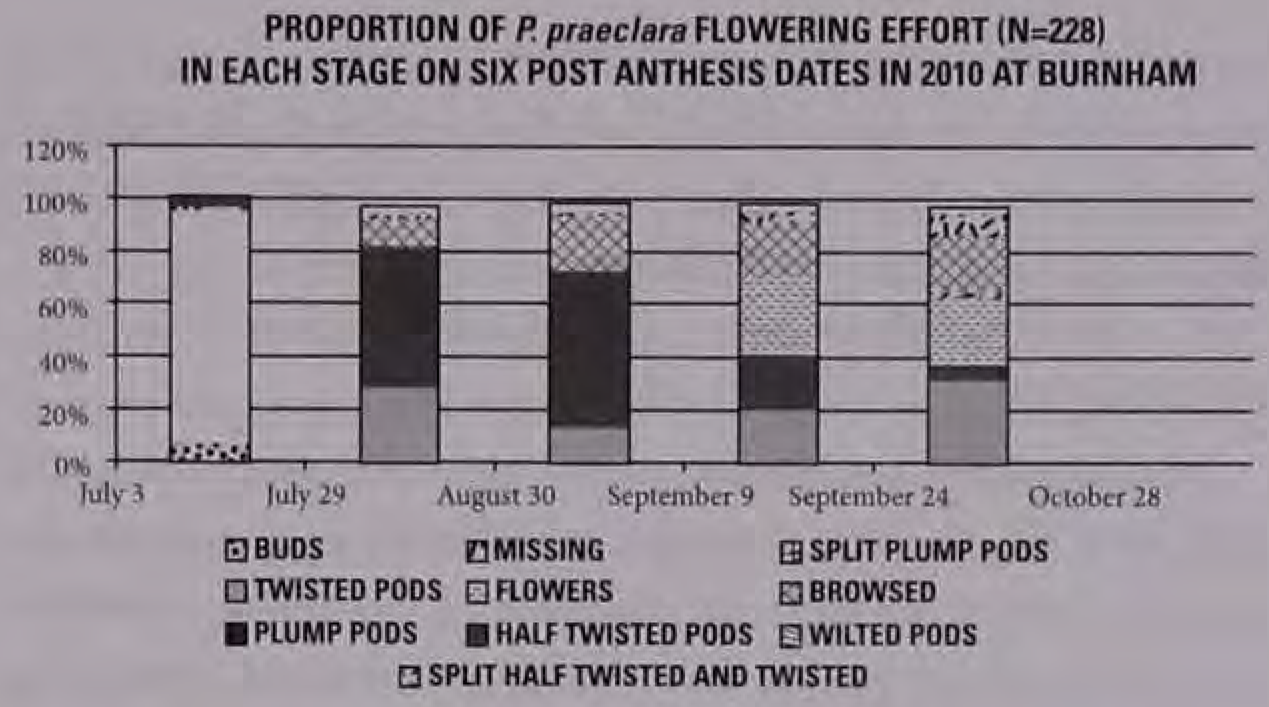

Figure 5. The shaded bars illustrate the proportion of the original 228 flowers in each stage of development between late anthesis and a killing frost. Slight variations below 100% may be the result of observer error.

original reproductive effort) remained to shed seed.

We documented a similar rate of loss at Burnham WMA in 2010, where we tracked postanthesis pod formation and maturation of 228 flowers on 24 plants from late anthesis on July 3 through the end of October. We categorized pods as "plump," "twisted," and "half plump/half twisted," all of which have been documented in previous studies to produce viable seed (Alexander 2006). July 3 plants averaged 8.3 flowers per plant, 92% of which formed capsules by the end of the month. By the end of August 21% of capsules had been lost to herbivory and by October an additional 17% had wilted or disappeared (Figure 5). At the time of hard frost, plump pods had dehisced but some twisted pods remained on the plants.

DISCUSSION

Lesica and Steele (1994) discuss implications of dormancy for monitoring studies. They suggest that surveys in periods of stress may not locate plants that are present in dormant state, studies that involve re-randomization of plots in order to estimate density and confidence intervals may be difficult to interpret, and the necessary duration of demographic studies will need to be adjusted to compensate for the duration of expected dormancy.

Because of our long period of demographic observation, we have been able to observe plants enter and emerge from multiple periods of dormancy. The longer duration of our study may account for differences between our observations and the shorter lives and lower return rates reported by Sieg and King (1995).

Periods of extended dormancy may also help explain the well-recognized tendency for the locus of flowering plants to shift geographically within populations from time to time and the resurgence of orchid populations after near-disappearance or periods of low numbers. Variations in demographic characteristics of populations between sites and years exhibited in preliminary analyses of our data (Quintana-Ascencio et al. 2004) suggest that it is unwise to draw conclusions from short-term studies or to extrapolate between sites. Studies of 3-5 years are insufficient to frame development of recruits to flowering plants and periods of dormancy that last for three or more years.

Complementary phenological, demographic, and census data illustrate the variability of population responses to management across sites or years on the same dates. The early emergence observed in some years in our phenological observations helps explain anecdotal evidence in our census and demography data that links late spring burns with the absence of flowering in the year of the burn. Differences in emergence dates and midsummer responses to management accentuate the need to understand site-specific physical conditions such as soil temperature, soil moisture, and depth of standing water throughout the year. Willson et al. (2006) present a useful model for relating the orchid's phenological year to climate data. Our data suggest that the actual mid April through late May phenological condition of *P. praeclara* (and hence its susceptibility to fire or frost damage) may vary greatly from site to site and year to year.

Previous studies that have investigated fruit set and seed production suggest that numbers are highly variable. Most observers tally flowers at anthesis and capsules within a month of flowering. In an ongoing study of management at Pembina Trail Preserve, Kiefer (2004) found average capsule to flower ratios of 8% to 20% between 1999 and 2004. In 2003 fruit set rates at Pembina Trail averaged 11% (Kiefer 2004). As part of a hand-pollination study conducted that year, Vitt and Theiss (2003) found viability of only 16% among naturally pollinated flowers (n=49). In a study on the Sheyenne National Grassland (North Dakota), Alexander (2006) reported an average of 9,825 seeds per capsule, 80% of which were viable. Our observed high capsule attrition rates at Crookston Prairie in 2006 and Burnham WMA in 2010 suggest that pod counts determined within a month of flowering may severely overestimate fecundity in some years.

Most monitoring and demographic studies are repeated at regular intervals of a year or longer between observations. Lesica and Steele (1994) suggest the need to extend the length of demographic studies to frame anticipated dormancy by at least two years, and suggest that surveys in periods of stress may not locate plants that are present in dormant state. Episodes of early senescence documented by our phenological observations illustrate the limitations of single observations in a given season. Because our phenology study is nested within the long-term demographic monitoring infrastructure, we may be able to compare true dormancy (Shefferson et al. 2005, Lesica and Steele 1994) with early senescence.

The combination of phantom emergence and capsule attrition argues strongly for monitoring protocols that enhance annual data collection at peak anthesis with early- and late-season observations. Because the demographic and phenological characteristics of populations vary with physiographic settings, orchid studies in given physiographic settings may not be easily extrapolated to other geographic areas. Our data suggest that the species' recovery strategy should include range-wide replications of whole-season monitoring at sentinel sites to fully understand trends and responses to environmental conditions and management.

ACKNOWLEDGMENTS

Minnesota DNR botanist Welby R. Smith initiated and implemented the first few years of the demographic monitoring that anchors the nested protocol. Volunteer database manager Rolf Dahle has managed the demographic data for over two decades. Then Regional Ecologist Janet Boe and volunteer Alice Sather sustained demographic monitoring in northwestern Minnesota throughout much of the 1990s. Judith Beckman, Donna Spaeth, and Rhett Johnson collected phenology data at Blue Mounds State Park and the Polk County Interbeach populations. Annual census of flowering plants would have been impossible without collaboration with the Minnesota Chapter of The Nature Conservancy under the guidance of Brian Winter and the assistance of over 70 DNR volunteer citizen scientists. We also want to acknowledge staff of the National Park Service at Pipestone National Monument and of the US Fish and Wildlife Service for sharing their annual counts with us. Funding for the majority of the work was supported by a series of discretionary cooperative agreements and Section 6 grants from the US Fish and Wildlife Service under the guidance of Charles Kjos and Phil Delphey of the Twin Cities Field Office. We wish to thank Karen Johnson, Caroline Sieg, Jyotsna Sharma, and other colleagues from neighboring states, Manitoba, and the federal Recovery Team for valuable discussions of the orchid and its biology.

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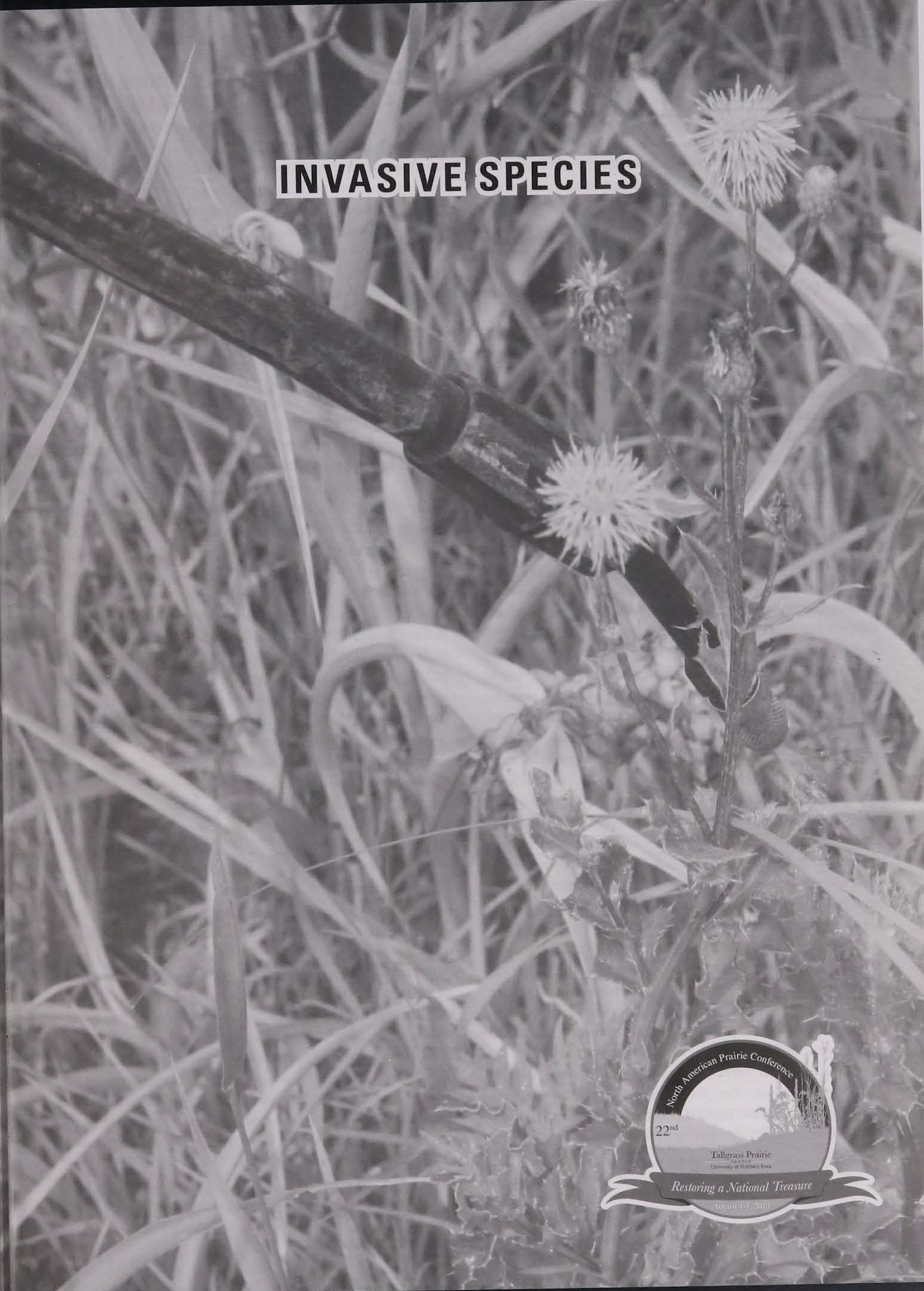
INVASIVE SPECIES



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Restoring a National Treasure
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AN ADAPTIVE APPROACH TO INVASIVE PLANT MANAGEMENT ON U.S. FISH AND WILDLIFE SERVICE-OWNED NATIVE PRAIRIES IN THE PRAIRIE POTHOLE REGION: DECISION SUPPORT UNDER UNCERTAINTY

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Abstract: Much of the native prairie managed by the U.S. Fish and Wildlife Service (Service) in the Prairie Pothole Region (PPR) of the northern Great Plains is extensively invaded by the introduced cool-season grasses smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*). Management to suppress these invasive plants has had poor to inconsistent success. The central challenge to managers is selecting appropriate management actions in the face of biological and environmental uncertainties. In partnership with the Service, the U.S. Geological Survey is developing an adaptive decision support framework to assist managers in selecting management actions under uncertainty and maximizing learning from management outcomes. The framework is built around practical constraints faced by refuge managers and includes identification of the management objective and strategies, analysis of uncertainty and construction of competing decision models, monitoring, and mechanisms for model feedback and decision selection. Nineteen Service field stations, spanning four states of the PPR, are participating in the project. They share a common management objective, available management strategies, and biological uncertainties. While the scope is broad, the project interfaces with individual land managers who provide refuge-specific information and receive updated decision guidance that incorporates understanding gained from the collective experience of all cooperators. We describe the technical components of this approach, how the components integrate and inform each other, how data feedback from individual cooperators serves to reduce uncertainty across the whole region, and how a successful adaptive management project is coordinated and maintained on a large scale.

Key Words/Search Terms: smooth brome, *Bromus inermis*, Kentucky bluegrass, *Poa pratensis*, native prairie, adaptive management, decision support, uncertainty, utility, learning, National Wildlife Refuge System, Prairie Pothole Region

INTRODUCTION

The extent of native prairie in North America has greatly declined from presettlement conditions. Native mixed-grass prairie has declined 30%–99% and native tallgrass prairie has declined more than 95% (Samson et al. 2004), primarily due to agricultural conversion. In the fragments of native prairie that remain, historic disturbances, such as grazing by native ungulates and frequent fire, have largely been excluded (Murphy and Grant 2005).

More than 100,000 ha of native prairie remnants are found in the Prairie Pothole Region (PPR) within the collection of National Wildlife Refuge System (NWRS) lands of the U.S. Fish and Wildlife Service (Service), which is charged with managing this large public land base. Given the decline of this ecosystem throughout the PPR, these refuge lands have become increasingly important conservation reservoirs for native prairie. Unfortunately, recent surveys of Service prairies revealed that these remaining fragments of native prairie are afflicted by a widespread invasion of two exotic cool-season grasses, smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*) (Grant et al. 2009). These invasions of Service-owned prairies are believed to stem in part from a common management history (circa 1935-85) of long-term rest and little or no defoliation by natural processes (e.g. grazing or fire) that historically shaped native vegetation communities (Grant et al. 2009).

Refuges are presently attempting to manage for native prairie and against these invasive grasses by reintroducing various forms of disturbance, including prescribed fire, grazing, and haying; however, results to date have been poor to inconsistent. Prairies differ by geographic location, tract size, degree of invasion, soils, etc., making their management an inherently complex undertaking. Managers face considerable uncertainties and operational constraints as they make decisions about the lands under their care (Smiley 2008). Success can be further hindered by a lack of coordinated effort among refuges in addressing prairie management. Refuges enjoy a high degree of autonomy, which can be an inducement for each station to act on its own, using different tactics to meet different objectives (Moore et al. 2011). Additionally, while monitoring has a long tradition in the NWRS, it has been less common for monitoring to be focused in a way that informs managers about the resource consequences of specific actions they take (Nichols and Williams 2006, Moore et al. 2011). A traditional go-it-alone approach to prairie management can therefore make it difficult to make sense of piecemeal outcomes that may be anecdotal, inconclusive, or contradictory.

To tackle this problem, scientists from the U.S. Geological Survey Northern Prairie and Patuxent Wildlife Research Centers are partnering with Service biologists and managers to develop an adaptive management-based system for making decisions about prairie management. This system will coordinate local efforts, recognize uncertainties that make management difficult, assist managers with making transparent and scientifically based management decisions given these uncertainties, and maximize the learning potential from management outcomes to reduce these uncertainties, thereby improving decision making and management through time. The framework of the adaptive management decision support system is built around the practical constraints of the refuges. The project interfaces with individual land managers who provide refuge-specific information and receive annual decision guidance that incorporates understanding gained from the collective experience of all cooperators. That is, individual cooperators learn from the dispersed efforts of all cooperators, as information feedback from each serves to reduce uncertainty across the whole region.

STUDY AREA

This project focuses on Service NWRS lands within the PPR of the northern Great Plains (Figure 1; Appendix A). Within the PPR, 19 refuge complexes and wetland management districts (hereafter referred to as refuges, stations, or cooperators) contributed 120 management units to the project. Management units are parcels that receive a single management treatment at any one time over its entire extent; average unit size was 35 hectares (range 3.5-241 ha). These units span the boundaries of two Service regions (3 and 6) and four states (North Dakota, South Dakota, Minnesota, and Montana).

ADAPTIVE MANAGEMENT-BASED DECISION SUPPORT SYSTEM

Adaptive management is an approach to recurrent decision making laid upon a foundation of predictive modeling, monitoring, and knowledge updating. Management decisions are chosen to pursue specifically identified management objectives, with the choice of best decision conditional on the present state of the managed system, and what is currently understood about behavior of the system. Adaptive management provides a formal framework for the improvement of management performance through the incremental reduction of uncertainty, an outcome achieved through repeated assessment of decision models against observed system response (Williams 1997, Kendall 2001, Moore and Conroy 2006, McCarthy and Possingham 2007).

The adaptive management framework consists of two stages: a setup phase, which is carried out only once or at infrequent intervals, and an iterative phase, which constitutes the recurrent steps of the annual decision-making process (Williams et al. 2007; Figure 2).

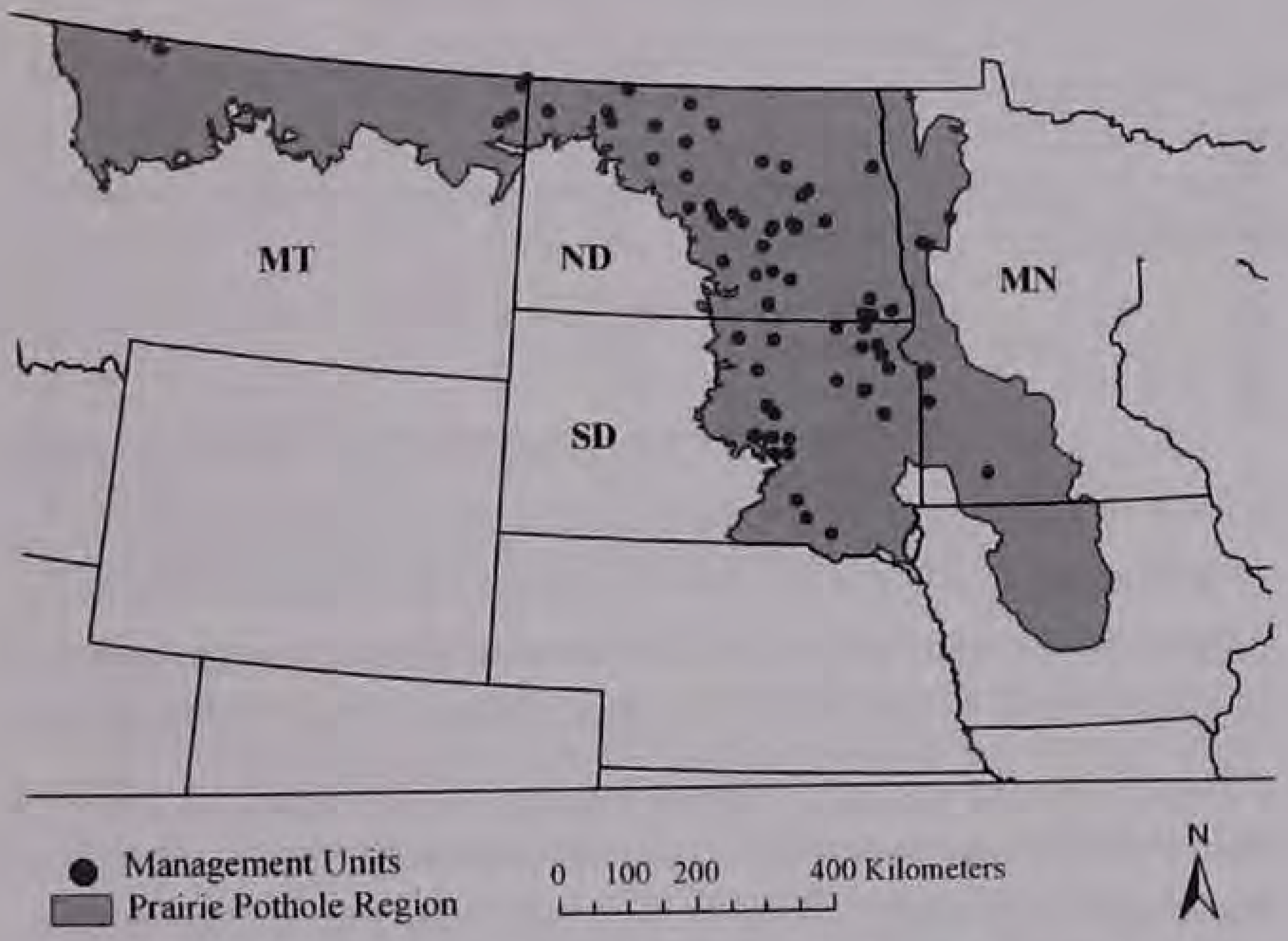


Figure 1. The project is focused on U.S. Fish and Wildlife Service National Wildlife Refuge System lands within the Prairie Pothole Region of the northern Great Plains. Service cooperators from nineteen different stations have enrolled in the project, resulting in 120 management units that span the boundaries of two Service regions (3 & 6) and four states (North Dakota, South Dakota, Minnesota, and Montana).

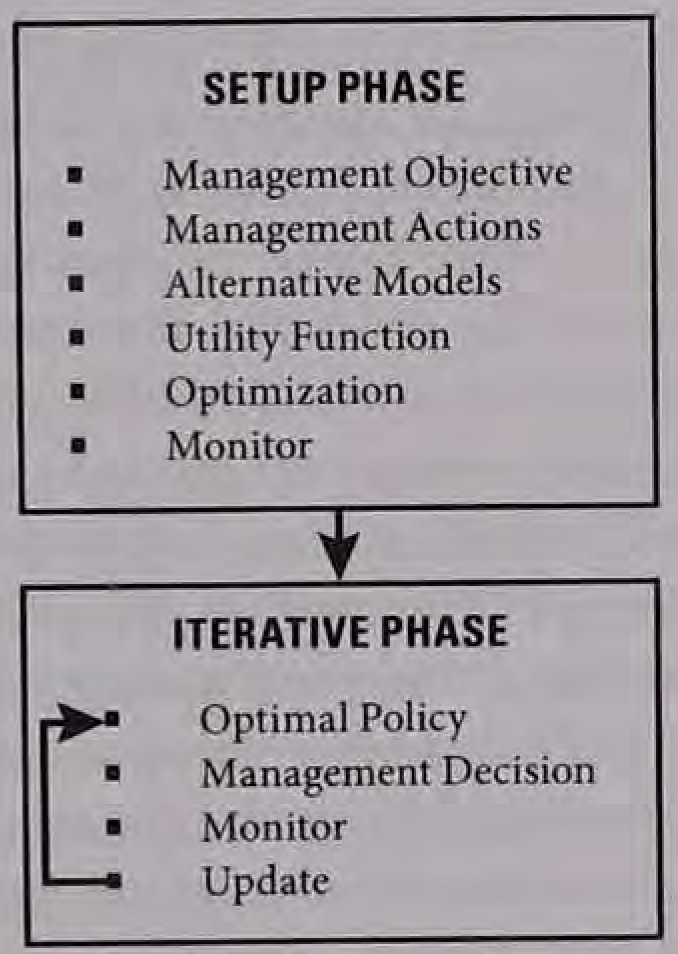


Figure 2. The adaptive management framework consists of two stages: the setup phase consists of six components and is carried out only once, while the iterative phase consists of four components and constitutes the recurrent steps of the annual decision-making process.

There are six components of the setup phase: (1) define the management objective, (2) establish the potential management actions, (3) identify uncertainties and develop alternative models, (4) determine the utility function, (5) compute the optimal decision table, and (6) develop and implement a monitoring protocol. The iterative phase consists of four components: (1) identify the optimal policy, (2) make and implement a management decision, (3) monitor the outcome, and (4) assess the outcome relative to model predictions and update model weights. The focus of this paper is to describe our decision-support system within the structure of these adaptive management framework elements; the framework we describe here is a work in progress.

ADAPTIVE MANAGEMENT FRAMEWORK: SETUP PHASE

MANAGEMENT OBJECTIVE

Under adaptive management, the selection of decisions is driven by an explicit, measurable management objective (Williams et al. 2002). The objective statement must (1) be quantifiable and measureable in the field, (2) define a quantity that can be generated as output from a decision model, and (3) balance tradeoffs among multiple objectives. One of our first tasks was to hold an initial, facilitated problem-scoping workshop to define the management objective. The workshop was held in July 2008 and was attended by 25 Service personnel (managers, biologists, project leaders) representing 19 different refuges from across the PPR of Service regions 3 and 6. During the workshop, participants considered various management goals and constraints and developed a consensus management-objective statement: *increase the composition of native grasses and forbs on native sod while minimizing cost.*

MANAGEMENT ALTERNATIVES

We next defined the menu of admissible decision alternatives that managers can use to pursue the management objective. Management decisions are supplied as input to decision models, and different decisions should yield different expected outcomes under the models. Management of grasslands is characterized by considerable uncontrolled natural variability; thus, it is important that differences in outcome among management alternatives be large and distinct if management is to be informative. A decision set containing a few, coarse-grained alternatives is more likely to provide rapid gains in learning than one containing a large number of subtly distinguished options. Therefore, construction of the set of decision alternatives was guided by actions likely to generate the greatest diversity in outcomes, and by logistical and political feasibility.

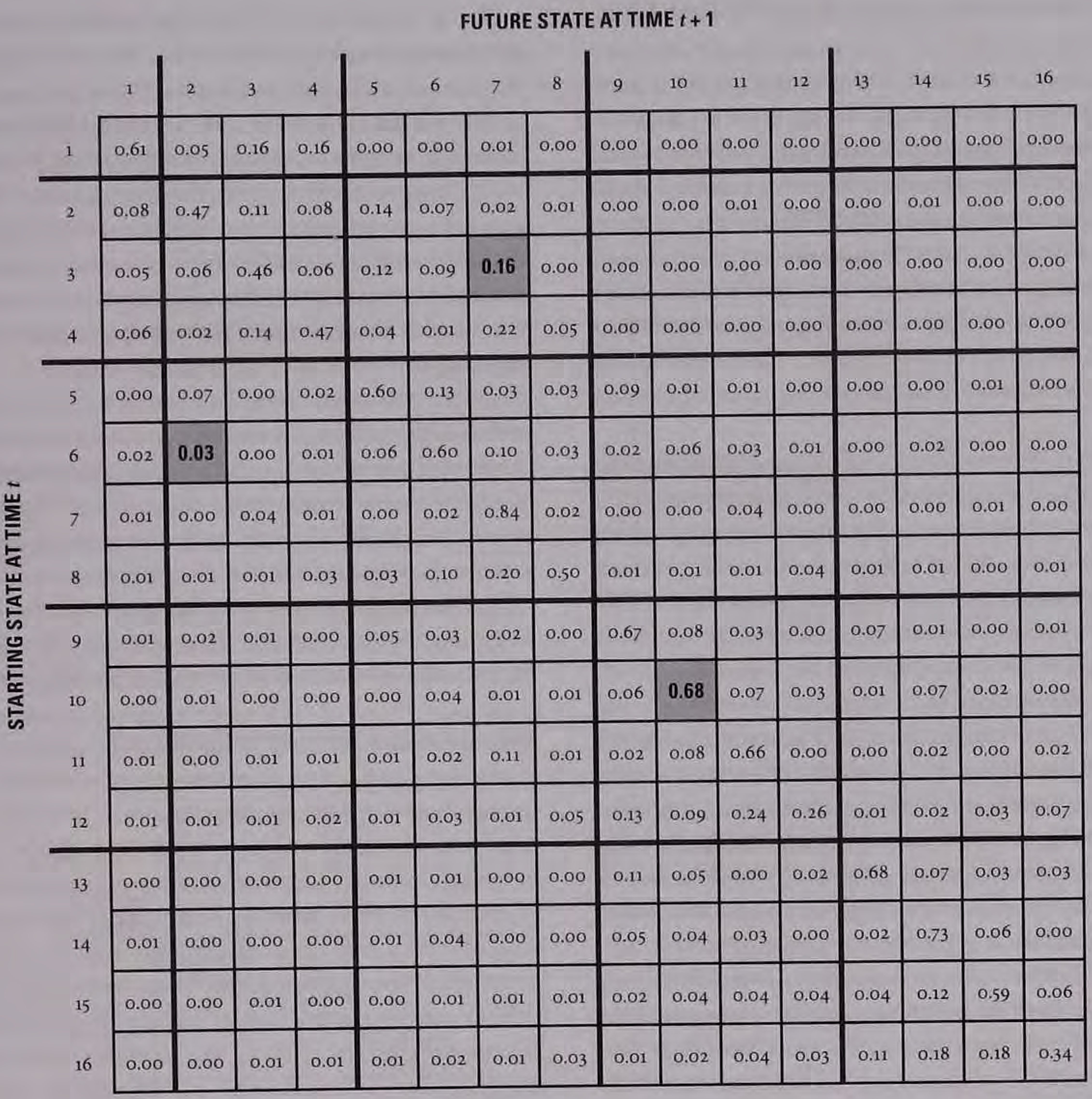
During the initial workshop, we elicited ideas from the participants about treatment options, identified constraints in their use, and narrowed the management alternatives to a manageable number to facilitate learning. The cooperators outlined five alternative management actions: *rest, hay, graze, burn, and burn/graze combination.* Each of the five management alternatives was generally defined with broad sideboards for timing, repetition, and intensity of application; within these sideboards, specific implementation of the action was left to the discretion of the manager. In each management year (defined as September 1-August 31), for each management unit, a manager selects one management action from this menu to apply to the unit.

UNCERTAINTY AND ALTERNATIVE MODELS DESCRIBING THE SYSTEM

We define the state of the biological system on each management unit at a particular time by two characteristics: the amount of cover of native grasses and forbs and the type of invasive grass that is dominant. We recognize five discrete states of native prairie cover: greater than 95%, 80%-95%, 50%-80%, 20%-50%, and less than 20%. Within each of the latter four states of native prairie cover, where some degree of invasion occurs, we recognize the dominant invasive as smooth brome, Kentucky bluegrass, or something other than either of these two invasive grasses. We also recognize smooth brome/Kentucky bluegrass codominance when native prairie cover is less than 80%. The five states of native prairie cover in combination with the dominant invasive results in 16 discrete possible states of the system (Figure 3).

	DOMINANT INVASIVE			
NATIVE PRAIRIE	SB	SB|KB	KB	OT
> 95%	1			
80 - 95%	2		3	4
50 - 80%	5	6	7	8
20 - 50%	9	10	11	12
< 20%	13	14	15	16

Figure 3. The composition of each management unit is categorized into one of 16 discrete states, depending on its amount of native grasses and forbs (>95%, 80%-95%, 50%-80%, 20%-50%, <20%) and its dominant invasive (smooth brome [SB], smooth brome/Kentucky bluegrass codominant [SB|KB], Kentucky bluegrass [KB], and other [OT]). We do not recognize codominant invasion status when native prairie cover is greater than 80%, and we do not recognize the dominant invasive when native prairie cover is greater than 95%. We define dominance as follows: smooth brome dominant if SB/(SB + KB) >= 0.67; Kentucky bluegrass dominant if KB/(SB + KB) >= 0.67; and other dominant if OT/(SB + KB + OT) >= 0.67. If none of these conditions are met, then a unit is categorized as codominant smooth brome and Kentucky bluegrass.



FUTURE STATE AT TIME *t* + 1

STARTING STATE AT TIME *t*

	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16
1	0.61	0.05	0.16	0.16	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2	0.08	0.47	0.11	0.08	0.14	0.07	0.02	0.01	0.00	0.00	0.01	0.00	0.00	0.01	0.00	0.00
3	0.05	0.06	0.46	0.06	0.12	0.09	**0.16**	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4	0.06	0.02	0.14	0.47	0.04	0.01	0.22	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
5	0.00	0.07	0.00	0.02	0.60	0.13	0.03	0.03	0.09	0.01	0.01	0.00	0.00	0.00	0.01	0.00
6	0.02	**0.03**	0.00	0.01	0.06	0.60	0.10	0.03	0.02	0.06	0.03	0.01	0.00	0.02	0.00	0.00
7	0.01	0.00	0.04	0.01	0.00	0.02	0.84	0.02	0.00	0.00	0.04	0.00	0.00	0.00	0.01	0.00
8	0.01	0.01	0.01	0.03	0.03	0.10	0.20	0.50	0.01	0.01	0.01	0.04	0.01	0.01	0.00	0.01
9	0.01	0.02	0.01	0.00	0.05	0.03	0.02	0.00	0.67	0.08	0.03	0.00	0.07	0.01	0.00	0.01
10	0.00	0.01	0.00	0.00	0.00	0.04	0.01	0.01	0.06	**0.68**	0.07	0.03	0.01	0.07	0.02	0.00
11	0.01	0.00	0.01	0.01	0.01	0.02	0.11	0.01	0.02	0.08	0.66	0.00	0.00	0.02	0.00	0.02
12	0.01	0.01	0.01	0.02	0.01	0.03	0.01	0.05	0.13	0.09	0.24	0.26	0.01	0.02	0.03	0.07
13	0.00	0.00	0.00	0.00	0.01	0.01	0.00	0.00	0.11	0.05	0.00	0.02	0.68	0.07	0.03	0.03
14	0.01	0.00	0.00	0.00	0.01	0.04	0.00	0.00	0.05	0.04	0.03	0.00	0.02	0.73	0.06	0.00
15	0.00	0.00	0.01	0.00	0.00	0.01	0.01	0.01	0.02	0.04	0.04	0.04	0.04	0.12	0.59	0.06
16	0.00	0.00	0.01	0.01	0.01	0.02	0.01	0.03	0.01	0.02	0.04	0.03	0.11	0.18	0.18	0.34

Figure 4. A 16 x 16 transition matrix describes the probability of a unit transitioning from any of the 16 resource states at time t (vertical axis) into some other state at time t + 1 (horizontal axis), after a particular management action is applied. For example, the matrix depicted represents transition probabilities (provisional; subject to further analysis) under the management action Rest. Probabilities within each of these 256 cells describe how likely each of the transitions is to occur under the management Rest. For example, a management unit starting in state 3 (80%-95% NP, KB dominant), has a 16% chance of degrading to the lower state 7 (50%-80% NP, KB dominant) under rest management. A unit starting in state 6 (50%-80% NP, SB|KB codominant) has a 3% chance of improving to state 2 (80%-95% NP, SB dominant), while a unit starting in state 10 (20%-50% NP, SB|KB codominant) has a 68% chance of remaining in that state.

STATE TRANSITION PROBABILITY MODEL

We use a state transition probability model to describe how we think the biological system behaves in response to management (Figure 4). A 16 x 16 transition matrix describes the probability of transitioning from state *x* at time *t* to state *y* at time *t*+1, given a particular management action. Because management actions are likely to influence transition probabilities from one state to another, a complete model consists of five matrices, one for each alternative management action. Given the current state of the unit and the management action applied, the model provides a probabilistic prediction of the state of the unit after applying the management action. Work is ongoing to estimate baseline transition probabilities for each matrix via a meta-analysis of data from several long-term studies. We estimated average transition probabilities across studies, and we placed vague prior probability distributions on the transition matrices to provide inference support for the many parameters where we had sparse data. To complete a prototype of our decision framework, we estimated provisional transition probabilities for mixed-grass prairies and tallgrass prairies separately, resulting in two empirically derived models; final estimates will be a focus of forthcoming work.

REPRESENTING UNCERTAINTY THROUGH ALTERNATIVE MODELS

Sole reliance on any one particular model implies that the behavior of the system is well understood and that responses to management decisions are predictable with a degree of accuracy. This is not the case in prairie management, where the wide range of site characteristics, current conditions, and variability in treatment execution makes the outcome of any treatment difficult to predict. Under adaptive management, uncertainties about response to management actions are cast in the form of plausible, competing decision models. Each model in the set represents an alternative hypothesis about how the system behaves.

Building alternative models proceeded from an initial synthesis of information about grassland restoration efforts, including expert opinions elicited from participants at the initial scoping workshop. By asking "What makes decision making difficult in grassland management?", we were able to identify and classify different areas of uncertainty in decision making. More targeted inquiry of the Service biologists serving on the project's Science Team allowed us to identify general areas of agreement about the behavior of the system, as well as to distinguish four major sources of uncertainty: (1) the effect of haying on native prairie, smooth brome, and Kentucky bluegrass, (2) the effectiveness of burning in suppressing smooth brome, (3) the effectiveness of grazing in suppressing Kentucky bluegrass, and (4) the existence of a threshold of native prairie composition, below which there is no benefit gained by active management.

We constructed four alternative models—four different notions of how we think the system may behave—to represent the four major sources of uncertainty. We formulated these four models by directed modification of the baseline, empirically derived transition probabilities. Model 1 serves as a reference point, expressing several baseline statements about system behavior, as follows: natural mimics of disturbance (i.e., graze, burn, and burn/graze combination) are equally effective at increasing native prairie, haying is equivalent to rest, and graze and burn are differentially effective against specific invasive species—grazing is more effective than burning against smooth brome and burning is more effective than grazing against Kentucky bluegrass. Models 2-4 differ from Model 1 in ways that isolate identified areas of uncertainty. Model 2 focuses on the first uncertainty, and states that while haying is less effective than the natural mimics of disturbance at increasing native prairie, it is more effective than rest. Model 3 targets the second and third uncertainties, and proposes that burning is not effective against smooth brome and grazing is not effective against Kentucky bluegrass. Model 4 aims at the fourth uncertainty and introduces the existence of a threshold (< 20% native prairie cover) below which active management is no better than rest. We have two sets of these four alternative models—one set for mixed-grass prairies and one set for tallgrass prairies—based on the same four concepts but derived from different initial transition probabilities, depending on the prairie type.

The mere existence of multiple models speaks to our uncertainty about the behavior of the system; however, we further quantify this uncertainty by assigning a weight to each model that connotes our current belief in each model as the best representation of system behavior. Model weights are an important element of adaptive management because they are a quantitative expression of current understanding about the system (also referred to as the belief state). Model weights determine the influence of each model on the decision at each point; models with greater weight exert more influence on the selection of a management action. However, model weights continually change through time in response to decisions made, as feedback from the monitoring data informs us about how well or how poorly each model performs as a predictor of management effect. The influence of each model changes through time as our understanding about the system changes. At the outset of decision making, where uncertainty is greatest, it may be reasonable to assign each model equal initial weight. Thus, we assigned equal initial weights of 0.25 to each of our four competing models.

IRREDUCIBLE FORMS OF UNCERTAINTY

The sources of uncertainty described above, around which our alternative models were created, are considered *structural uncertainty*—the type of uncertainty that adaptive management is intended to address and reduce. Three other sources of uncertainty exist, however, and include: (1) *environmental stochasticity*, unexpected outcomes brought about by chance events (e.g., unanticipated differences in treatment efficacy due to temporal and spatial variation in precipitation), (2) *partial controllability*, the inability to carry out an action as intended (e.g., an incomplete burn), and (3) *partial observability*, the inability to see or measure the system accurately (e.g., sampling variability in monitoring) (Nichols et al. 1995, Williams 1997). These sources of uncertainty are themselves irreducible; nevertheless, because they can have an impact upon decision making, they must be addressed and implicitly or explicitly accounted for in the predictive models. In our decision framework, environmental stochasticity is reflected through the probabilities contained in the state transition models; that is, because of the effects of the random environment, the transition from a given state into the same or some other state is not known with certainty, but only probabilistically. Partial controllability will be accounted for in a model component that makes a probabilistic determination of which action is carried out given which action was indicated as "best"; we will elicit from our cooperators information that will help parameterize this model component. Finally, we plan to account for partial observability in the updating step of our framework; inaccuracy in measuring the resource should result in reduced learning from management actions.

UTILITY FUNCTION

The utility function describes what we want from the system through management. It combines both the resource and cost aspects of the management objective by balancing the *value* of having native prairie with the *cost* of achieving it. The utility function is a subjective expression of the value system (i.e., importance of having native prairie, undesirability of invasive species, willingness to direct resources to address either) of the cooperators. As such, parameterizing the utility function is a process that stands completely apart from the process that expresses our beliefs about the science of the system (i.e., construction of the model set). Utility is the annual measure of what the manager receives from the system in return for what he/she invests; therefore, it is reasonable to describe sound management as the sequence of decisions over many years that makes accumulated utility as large as possible.

We distinguish three main characteristics that cooperators value: (1) high cover of native prairie, (2) increasing the amount of native prairie cover, and (3) gaining more native prairie cover for less investment. We have constructed the utility function to recognize these three values by accounting for three corresponding elements: (1) the future native prairie state resulting from an action, (2) the starting native prairie state before applying an action (comparison between elements 1 and 2 allow us to distinguish between improvements and degradations in prairie state), and (3) the management action that was taken to prompt the transition between the starting and future states. Because subjective preferences are hard to draw out and evaluate, and because different stakeholders will have different perspectives of how they value these three elements, quantifying the utility function will require expertise to elicit and resolve these values. While the actual quantification of the values may vary, the utility function will follow the structure outlined above (Figure 5).

		FUTURE STATE				
		> 95%	80 - 95%	50 - 80%	20 - 50%	< 20%
STARTING STATE	> 95%	1.0	**0.40**	0.03	0.006	0.001
	80 - 95%	1.0	**0.80**	0.13	0.009	0.003
	50 - 80%	1.0	**0.90**	0.35	0.02	0.006
	20 - 50%	1.0	**0.95**	0.58	0.10	0.008
	< 20%	1.0	**0.98**	0.74	0.23	0.01

Figure 5. This is an example of how we quantify the values cooperators place on having high cover of native prairie (NP) and gaining more NP. These values are expressed in a two-dimensional utility matrix indexed by the NP cover before (starting state) and after (future state) a management action. We assign values of utility to each possible transition between the starting and future states, where utility is expressed with a value ranging between 0 and 1. These values represent cooperator satisfaction with each outcome (0 is the least and 1 is the most satisfied). Values along the diagonal represent cooperator satisfaction with staying in a given state; satisfaction is greatest with the highest NP cover and declines with lower NP cover. Cooperators also value making improvements from a lower to a higher NP state and disfavor degrading from a higher to a lower NP state. Values in the matrix beneath and above the diagonal represent transitions where NP was gained and lost, respectively. Given a future NP state of 80%-95% (bolded values), satisfaction is relatively high (0.8) when that condition was maintained from its starting state, is greater (0.90) when that condition was an improvement from a lower state of 50%-80% NP, and is greatly reduced (0.4) when that condition resulted from a degradation from >95% NP. The third element of the utility function, cost, is not shown here. The complete utility function, encompassing all three aspects of the cooperators' value system, consists of five utility matrices like the one depicted here—one for each management action; each matrix contains the same internal relationship among values, but the utilities are discounted according to the relative cost of management actions, which are ranked from most expensive (burn/graze) to least expensive (rest).

OPTIMIZATION

Optimization is the search for best management actions through a process that integrates the model, which describes how we think the system works, and the utility function, which describes our values. Dynamic programming is a form of optimization for decisions and the resulting rewards (utility values) that occur through time (Dreyfus and Law 1977). We use adaptive stochastic dynamic programming (ASDP; Lubow 1995, 1997), which accounts for current and future expected rewards, future dynamics of system state and knowledge gain, and the degree of management control (partial controllability). The procedure determines the trajectory of decisions through time that will maximize expected cumulative utility, thereby achieving the management objective. The end product of the optimization is a large table that contains every possible combination of resource state (i.e., 1-16) and belief state (i.e., weights assigned to the four alternative models), and identifies the optimal management decision for each combination (Table 1).

The optimal decision table generated by ASDP provides a best decision for the current condition of the resource and for the degree of confidence (model weights) we currently place on each of the four alternative models. The current condition of the resource is management-unit specific and ascertained annually via a standardized monitoring program (see "Monitoring" below). The current understanding of the system, indicated by the weights assigned to each model, is specific to prairie type (i.e., mixed or tall) and is determined annually via an updating procedure (see "Compare and Update" below). Because we have two sets of alternative models, one for mixed-grass prairies and one for tallgrass prairies, we obtain two optimal decision tables, one for each prairie type.

MONITORING

The monitoring protocol is designed to provide data for three purposes: (1) determining current system state (i.e., prairie composition) on each management unit, (2) evaluating progress toward the management objective, and (3) assessing predictive performance of the alternative models. We adopted a protocol that employs a modified belt-transect sampling method (Grant et al. 2004) and was familiar to many

Table 1. Excerpt from an optimal decision table produced from the adaptive stochastic dynamic optimization. The full table contains all possible combinations of native prairie state (i.e., 1-16) and belief state (i.e., weights assigned to the four alternative models), and identifies the optimal management decision for each combination. We discretized model weights by 0.125, making 165 possible combinations of the four model weights (three of which are shown in the excerpted portion of the table). Combining 165 belief states with 16 resource states results in 2,640 possible combinations. As an example, if our current understanding of system behavior is perfect, with 100% of confidence on Model 1, then the optimal decision for a management unit in state 3 would be to Burn. However, if our current understanding of system behavior is imperfect, with 37.5 % of confidence in Model 1, 37.5% in Model 2, 0% in Model 3, and 25% in Model 4, then the optimal decision for the same resource conditions would be to Burn/Graze.

STATE	MODEL 1	MODEL 2	MODEL 3	MODEL 4	OPTIMAL DECISION
1	1	0	0	0	Hay
2	1	0	0	0	Graze
3	1	0	0	0	Burn
4	1	0	0	0	Graze
⋮	⋮	⋮	⋮	⋮	⋮
16	1	0	0	0	Graze
1	0.625	0.125	0.125	0.125	Rest
2	0.625	0.125	0.125	0.125	Graze
3	0.625	0.125	0.125	0.125	Burn
4	0.625	0.125	0.125	0.125	Burn/Graze
⋮	⋮	⋮	⋮	⋮	⋮
16	0.625	0.125	0.125	0.125	Graze
1	0.375	0.375	0	0.25	Rest
2	0.375	0.375	0	0.25	Graze
3	0.375	0.375	0	0.25	Burn/Graze
4	0.375	0.375	0	0.25	Burn/Graze
⋮	⋮	⋮	⋮	⋮	⋮
16	0.375	0.375	0	0.25	Graze

of the refuges across the Dakotas. A primary consideration when deciding upon the monitoring effort was that it be sustainable by the refuge personnel who are charged with carrying it out each year. For this reason, only necessary attributes that inform the models are measured. In addition to being sustainable, the monitoring protocol reliably conveys prairie composition, is flexible for use in both mixed- and tallgrass prairie, is quick and efficient, and is robust to multiple observers. Along with monitoring prairie composition, refuge managers are responsible for keeping detailed descriptions of the management activities they carry out on each management unit each year (e.g., burn intensity, stocking rate, timing of application, etc.) so that over time a fuller picture of management practices emerges, facilitating future study of native prairie response to management.

A centralized database was developed to standardize, organize, and maintain the vegetation monitoring data and the management activity data collected by project cooperators. Vegetation monitoring occurs annually during the growing season (between June and August).

ADAPTIVE MANAGEMENT FRAMEWORK: ITERATIVE PHASE

LOOK UP THE OPTIMAL DECISION POLICY

Given the current state of the system and the current understanding of the system, identifying the current best management decision is a matter of looking up the combination (i.e., system state and belief state) in the appropriate (i.e., mixed- or tallgrass prairie) optimal decision table (e.g., Table 1). Given complete uncertainty at the outset of decision making, 0.25 is a reasonable initial assignment of weight to each model. The decisions in the optimal decision table that correspond to this current level of understanding constitute the current optimal decision policy (Figure 6). Following monitoring, which informs cooperators about the current composition of native prairie on their sites, we identify the recommended management actions for each unit with respect to its system state and its prairie type (mixed or tall) by consulting the current optimal decision policy. By 31 August of each year, we provide individual cooperators with a recommended management action for each of their management units for the upcoming management year (September 1-August 31).

In future iterations of the decision cycle, the current state of the system will be ascertained by the annual monitoring program (see "Monitor" below) and the current understanding of the system (i.e. weights on each alternative model) will be determined by the annual updating procedure (see "Compare and Update" below).

MAKE AND IMPLEMENT A MANAGEMENT DECISION

Upon receiving the management recommendations for their units, managers consider the recommendation, along with other relevant information (e.g., funding constraints; access to a burn crew, cattle, or haying cooperator; fuel load; weather conditions), and decide what management action to implement on each unit that year. The management action is carried out at some point during the management year (September 1-August 31).

MONITOR

During the period of the growing season when both cool-season and warm-season grasses are visible (June-August), refuge personnel carry out the annual monitoring protocol. Cooperators individually enter their vegetation and management data in the standardized database and trans-

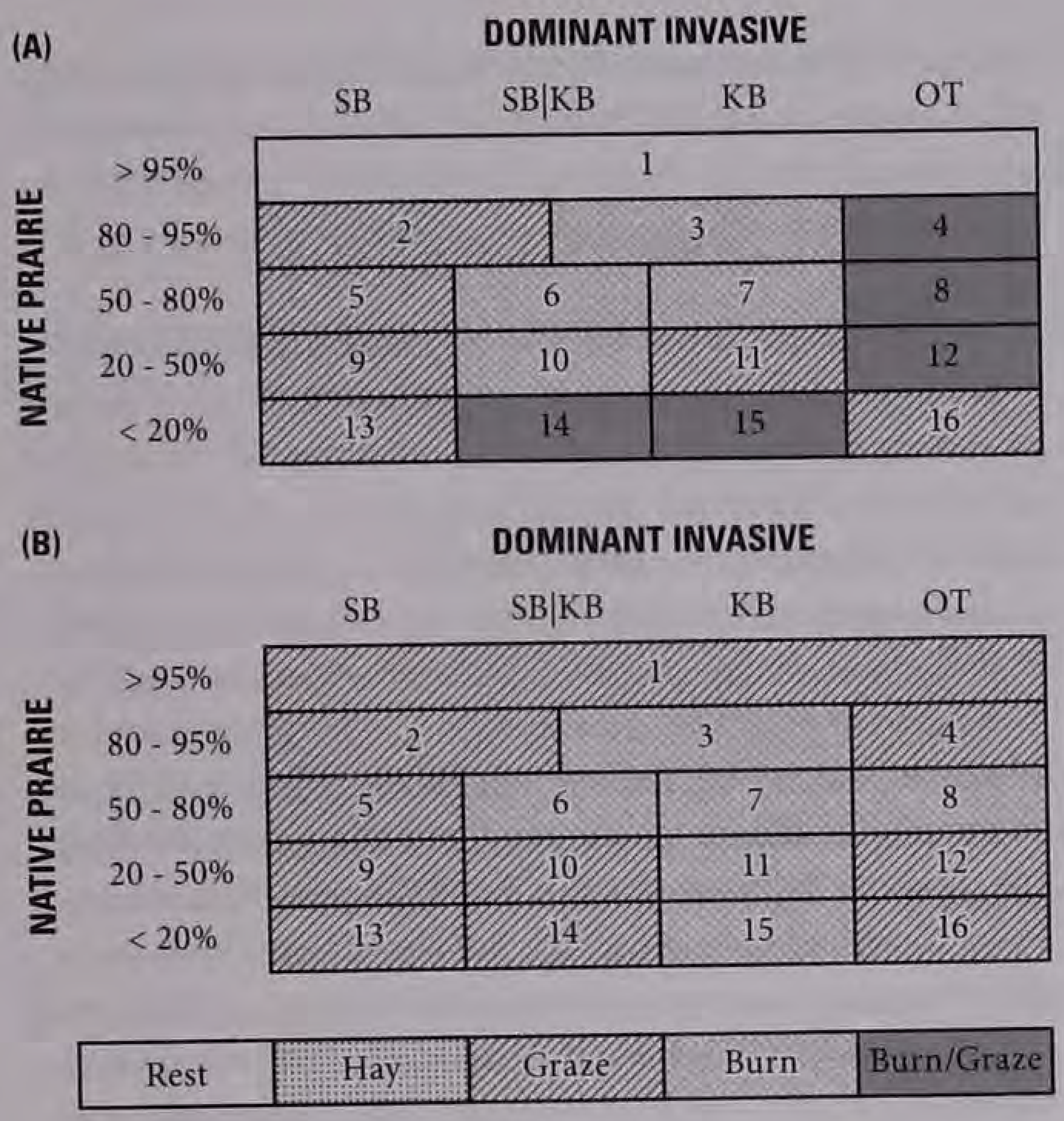


Figure 6. Optimal decision policies for (a) mixed- and (b) tallgrass management units resulting from the adaptive stochastic dynamic optimization. These policies are identified by extracting from the large optimal decision tables the optimal actions for each state that pertain to complete uncertainty among the four alternative models (i.e., 0.25 weight on each model). These two policies are used to generate the recommended management actions per unit with respect to its system state (i.e., 1-16) and its prairie type (i.e., mixed or tall).

mit them to a centralized site for integration by 25 August of each year, a deadline necessitated by the management decision cycle.

COMPARE AND UPDATE

Prediction and monitoring are the keys to reducing uncertainty. After managers select their management decisions, we use the models to make predictions about the outcome (i.e., prairie composition state) of the chosen management action on each management unit. After the management action is carried out and the resulting prairie state is observed through the annual monitoring program, we compare the predicted outcomes of our models to the observed outcome. By computing the likelihood of the observed response under each alternative model and applying Bayes' rule, we update our model weights (Johnson and Williams 1999). The comparison of each model's prediction to the actual outcome allows us to distinguish models that are better representations of system behavior than others. To the extent that one model's prediction makes a better match to the observation than does another model, that model will inherit a greater share of the weight at the expense of the other models; in other words, we are reducing uncertainty among models. With this updating of knowledge, the iterative cycle is complete, and the new model weights become the starting point for the next cycle of the iterative phase.

THE ITERATIVE CYCLE: PUTTING IT ALL TOGETHER

With the updated model weights in hand, we return to the first step of the iterative cycle and look up the new decision policy corresponding to the newly realized model weights. Because the optimal decision policy is influenced by the model weights, for the next decision cycle, those models that have garnered more weight will assert more influence on the resulting policy and thus on the next decision recommendation.

We repeat the cycle of deciding on a management action, predicting the response, monitoring the outcome, comparing the predicted and observed outcomes, and updating the model weights (Figure 7). Through time we hope to see an accumulation of weight on one of the models, thus distinguishing it as a better model. The shift in model weights that occurs through this iterative cycle embodies our learning and reduction in uncertainty about system behavior. The change in the optimal decision policy that results from the adjusted relative influence of the decision models exemplifies how our management adapts based on this learning. It is this change in understanding of how the system works and this updating of model influence on subsequent decisions that makes the management *adaptive*. Because decisions are based on improved understanding of the system, the result is management decisions of better quality than when we started.

FUTURE WORK

We have presented in this paper the framework for the native prairie adaptive management project. We completed this first prototype of the framework in August 2010, at which time we successfully completed our first full implementation of the adaptive management cycle. While the general framework will remain the same, we will be revisiting the specifics of several of the components (e.g., representation of the system state, estimation of the baseline model transition probabilities, construction of alternative models, elicitation of values for the utility function, consideration of partial controllability, and evaluation of the updating method). The final framework will be complete by August 2011, at which time the Service will assume operational control of the iterative steps of the framework.

BENEFITS AND TRADEOFFS

We see some clear benefits from applying this adaptive framework. First, refuge managers maintain flexibility of management at the scale of the individual field station. The management practices and associated implementation tasks are familiar to managers, and managers are not locked into carrying out specific actions; in other words, the framework provides *decision support*, not decision mandates. Second, because multiple individual stations are coordinating annually under this framework, learning is enhanced through spa-

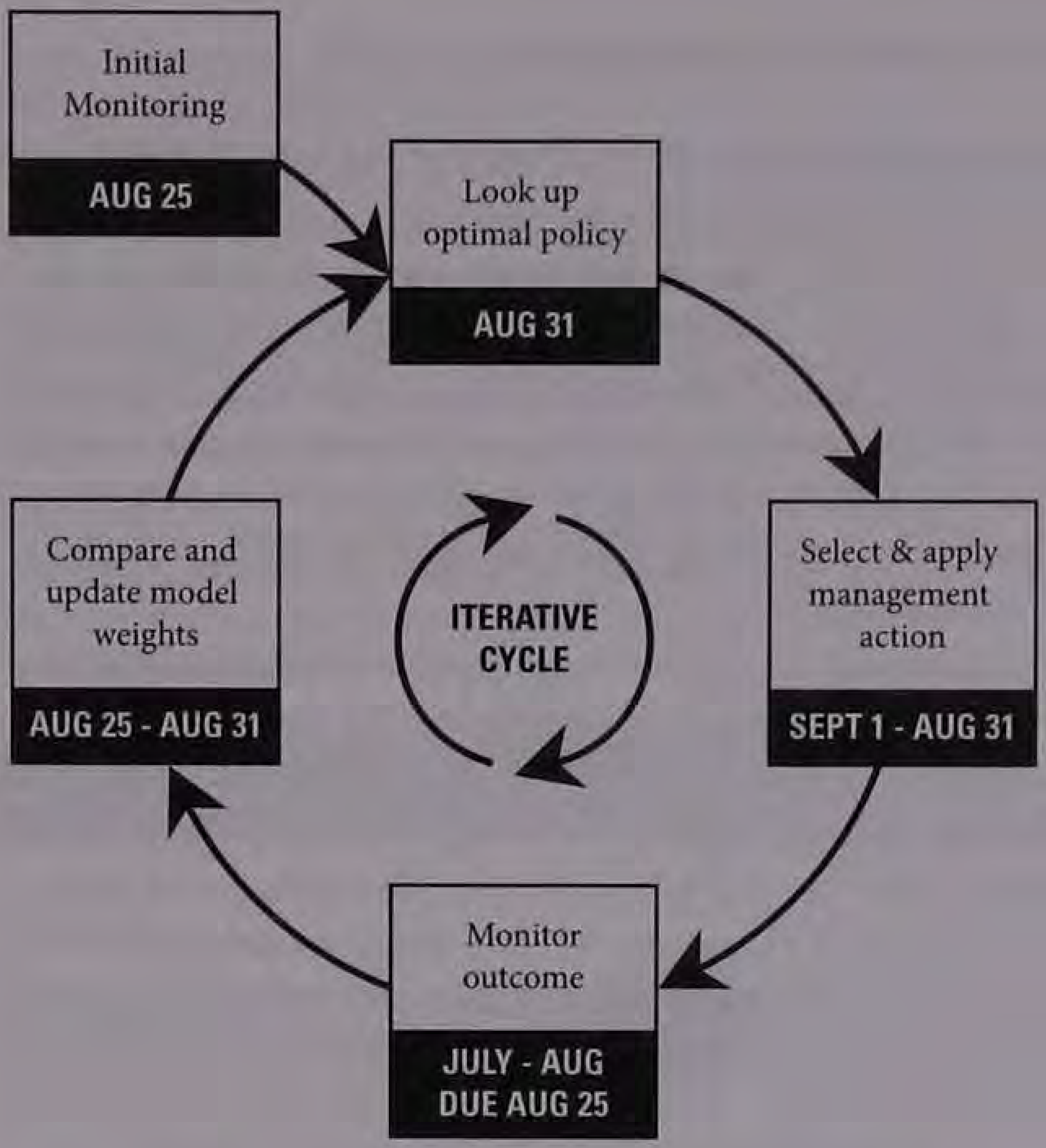


Figure 7. After initial monitoring of all management units during the setup phase, we know the current state of each of the management units and move into the iterative cycle of the adaptive management framework. Here we enter into an annual cycle that includes (1) identifying the optimal policy and generating recommended management actions for each unit with respect to its current state (August 31); (2) selecting a management action for each unit and then applying that action (September 1-August 31); (3) monitoring the units for their new state after management has been applied (July-August), and entering and transmitting the data (August 25); and (4) comparing the predicted outcomes of each alternative model to the observed outcomes from the monitoring data and updating the confidence weights on each model. With the updated model weights in hand, we return to the first step of the iterative cycle and look up the new decision policy that corresponds to the newly realized model weights. Because the optimal decision policy is influenced by the model weights, when it comes to making the next decision, models that have garnered more weight will assert more influence on the resulting policy and thus on the recommended next decision.

tial and temporal replication. Third, the gain in knowledge is directed back to improve management at both the local and the system-wide scales. Each station benefits from the collective gain in understanding achieved by all stations.

As in any effort over a large and heterogeneous system, there are tradeoffs. The flexibility exercised by individual stations and the large geographic scale make for a noisy system, which means that learning is slower compared with a controlled experiment that has strict protocols and dictated actions. But, if the framework is adhered to, learning will occur.

It is beyond the scope of this paper to address the process of consultation and negotiation with cooperators on the many difficult aspects of this problem. Discussions concerning the choice of the annual time step, the state structure, the date which demarcates successive management years, and the model set were thoughtfully considered and negotiated. The science team and resource managers conducted these interactions with the understanding that the inherent complexity of the system had to be simplified to make the problem manageable, and that the desire to represent the complexity had to be balanced with the need to be parsimonious.

ATTRIBUTES OF SUCCESSFUL IMPLEMENTATION

We agree with Moore et al. (2011) that there are three elements at the heart of a successful, integrated, large-scale adaptive management effort: components, collaboration, and commitment. The first, *components*, has been the focus of this paper and includes all the steps of the setup and iterative phases of the adaptive management framework (i.e., management objective, management action alternatives, alternative models, utility function, optimization, monitoring program, decision selection, and assessing and updating). The second is *collaboration* that is well-structured and broad. Team members should include people who are knowledgeable about management issues, operational procedures, and refuge capabilities and constraints; skilled in coordination, communication, organization, elicitation, and facilitation; and have expertise in decision structuring and modeling. Regular communication among members of the project team and between the team and cooperators, as well as a common understanding of roles and responsibilities among team members, are requisite for successful collaboration. Adaptive management is a challenging undertaking, especially in environments that operate in short-term budgetary and priority-setting horizons; thus, long-term *commitment* to the process at the station, coordinator, and administrative levels is vital to project success. All three elements are essential to successfully develop, implement, and reap the benefits of a large-scale adaptive management project.

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and conclusions in this article do not necessarily represent the views of the U.S. Fish and Wildlife Service. Use of trade, product, or firm names does not imply endorsement by the U.S. government.

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Appendix A. List of U.S. Fish and Wildlife Service project cooperators. National Wildlife Refuge (NWR). Wetland Management District (WMD).

COMPLEX	NO. OF UNITS	GRASS TYPE	STATE	USFWS REGION
Arrowwood NWR Complex	16	Mixed	ND	6
Audubon NWR Complex	11	Mixed	ND	6
Benton Lake WMD	2	Mixed	MT	6
Big Stone NWR	1	Tall	MN	3
Detroit Lakes WMD	3	Tall	MN	3
Devils Lake WMD	2 1	Mixed Tall	ND	6
Huron WMD	10	Mixed	SD	6
Kulm WMD	10	Mixed	ND	6
Lake Andes NWR	3	Mixed	SD	6
Long Lake WMD	5	Mixed	ND	6
Lostwood NWR Complex	3	Mixed	ND	6
Madison WMD	3	Tall	SD	6
Medicine Lake NWR	6	Mixed	MT	6
Morris WMD	9	Tall	MN	3
Sand Lake NWR Complex	3 2	Mixed Tall	SD	6
Souris River Basin NWR Complex	10	Mixed	ND	6
Tewaukon WMD	7	Tall	ND	6
Waubay NWR Complex	11	Tall	SD	6
Windom WMD	2	Tall	MN	3

MONITORING THE ANNUAL INCREASE OF GARLIC MUSTARD IN EASTERN NEBRASKA

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Abstract: Introduced from Europe to North America in the mid-1800s, garlic mustard (*Alliaria petiolata*) has demonstrated an effectiveness at invading the understory of many wooded sites. The increase in garlic mustard was monitored at the Boyer Chute National Wildlife Refuge located in eastern Nebraska in wooded plots adjacent to existing prairie and prairie restoration projects. Permanent transect locations were established and line-intercept data collected each summer from 2008 to 2010. Data indicate a rapid increase in the number of transects intersecting garlic mustard as well as a dramatic increase in frequency each year. These studies emphasize the rate of colonization as a significant challenge when attempting to control garlic mustard.

INTRODUCTION

Garlic mustard has steadily invaded the understory of North American forests, but the dynamics of the long-term invasion by this species have not been thoroughly studied (Lankau 2009). Invasion by garlic mustard has been correlated with changes in soil bacteria composition (Burke 2010), with inhibition of arbuscular mycorrhizal fungi (Barto 2010) and with complex alteration of plant communities (Pardini 2009). The purpose of this study was to monitor and quantitatively measure the invasion of garlic mustard into forested areas of eastern Nebraska.

MATERIALS AND METHODS

STUDY SITE

Data was collected at the Boyer Chute National Wildlife Refuge, and special use permits were obtained from the U.S. Fish and Wildlife service for all activities. Three heavily wooded study sites were selected on an island formed by a remnant of the Boyer River which connects with the north and south portions of a bend in the Missouri River (Figure 1). The refuge consists of at least 3,121 acres and is currently expanding to a planned 9,912 acres with holdings in both Iowa and Nebraska. The study site is in the ten-year or less floodplain of the Missouri River (990-995 feet above sea level). Historically the area was heavily utilized for agriculture, but has been modified greatly since refuge establishment in 1999 by the Army Corps of Engineers and the Fish and Wildlife Service (Boyer Chute Restoration Project).

MAPPING

Map coordinates for the perimeter of the study area were obtained using a hand-held GPS unit (Garmin, eTrexH) with WAAS enabled. The resulting GIS data was plotted using QtiPlot graphical software. Geometric distortion of data was corrected by correlation of latitude and longitude to surface distances measured on site (0.052±0.003 minutes / 100m north-south; 0.077±0.003 minutes / 100m east-west).

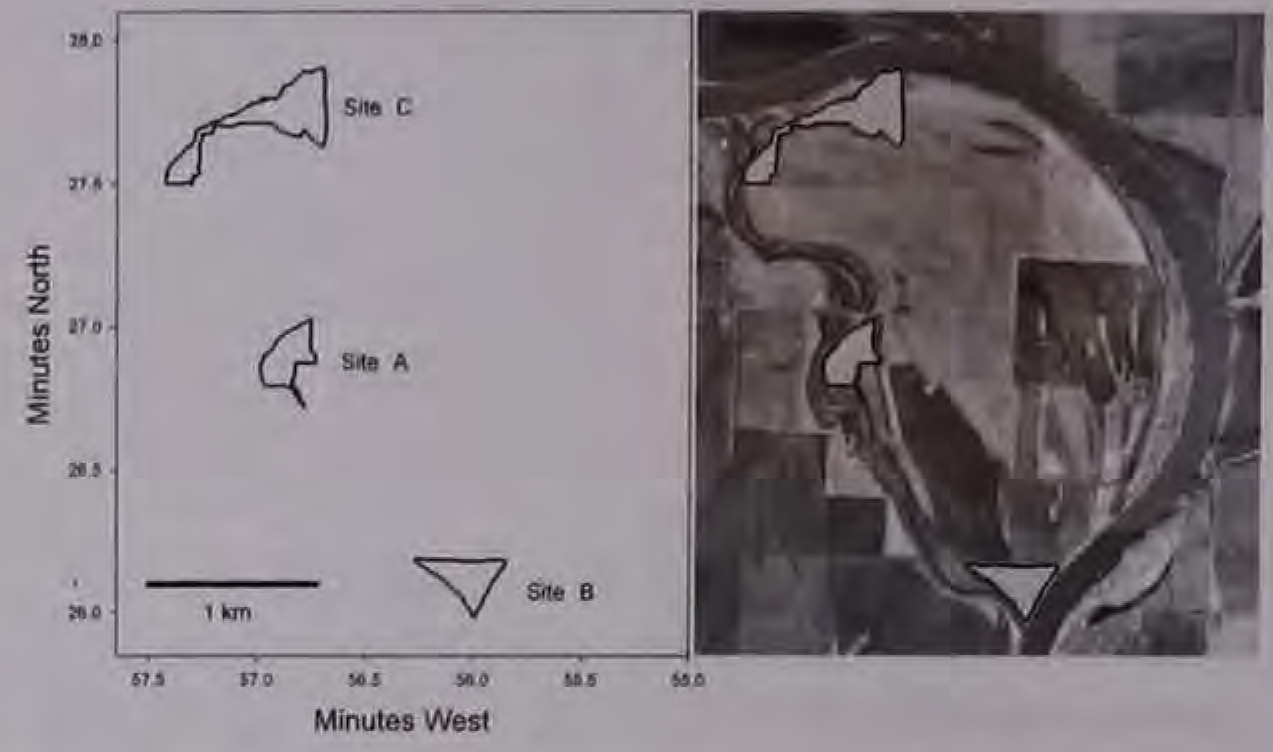

Figure 1. Boyer Chute study sites. Boundaries of study sites were mapped by walking the perimeter and determining GPS coordinates every 100 to 200 meters. Correlation of GPS coordinates to physical distances was accomplished with on-site measurements as described in Materials and Methods. Aerial photograph of the study site, composed of an island bordered on the north and east by a bend of the Missouri River and the chute of the Boyer River on the west and south.

VEGETATION MEASUREMENTS

Twenty-seven permanent sample sites were established in 2007, and data were collected at those sites in the summers of 2008, 2009, and 2010 in late June and July. Sites were located from previously established GPS coordinates, and point intercept data were collected. Data for each site consisted of a single 20-meter point intercept transect with intercepts measured every meter to a height of 1.5 m utilizing standard methodology (Bonham 1989).

DATA ANALYSIS

Descriptive and mathematical significance of data were determined using the OpenOffice spreadsheet software and OOoStat Statistics Macros version 0.5. Data were analyzed by one-way ANOVA with statistical significance set at $p<0.05$.

RESULTS AND DISCUSSION

Garlic mustard increased in geographic distribution and frequency in the wooded areas of Boyer Chute National Wildlife Refuge between 2008 and 2010. Sampling locations containing garlic mustard increased from 3 to 9 to 12 locations in 2008, 2009, and 2010, respectively (Figure 2). Garlic mustard was intersected at a frequency of 47, 109, and 225 intercepts / 1000 samples in 2008, 2009, and 2010, respectively (Figure 3). Because there was no apparent Corps of Engineers activity between 2005 and 2009, the movement of garlic mustard was likely not due to construction projects on the refuge. The increase in garlic mustard was significant ($p<0.05$) as evaluated by one-way ANOVA and can be crudely estimated as a doubling of frequency each year. For comparison, white snakeroot was intersected 133, 113, and 81/1000 samples in 2008, 2009, and 2010, respectively, but the difference between years was not statistically significant despite a strong downward trend in frequency (Figure 3). Given the observed rate of garlic mustard increase, it is anticipated that garlic mustard will continue to increase at the study site.

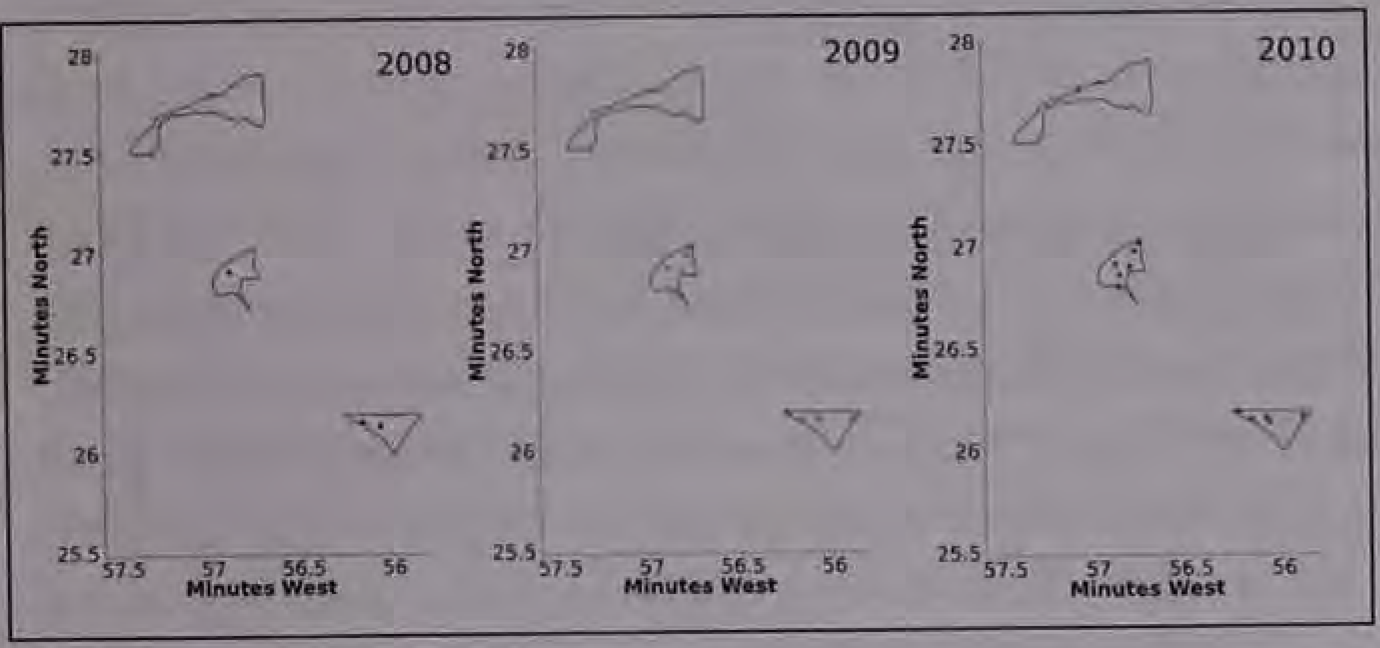

Figure 2. Location of sample sites containing garlic mustard. Map of study sites illustrating the location of transects intersecting garlic mustard in the summers of 2008, 2009, and 2010.

While long-term data will be required for modeling, some gross predictions can be made for the movement of garlic mustard within the study site. As seen in Figure 2, garlic mustard has on average appeared at 3-6 new locations per year, which would imply that 100% of the 27 permanent transects would possess at least one garlic mustard intersection in 2-4 years, which would correspond to sometime between the years 2012 and 2014. Approximating the change in garlic mustard frequency by line intercepts is more problematic and difficult to anticipate with sparse data, but the maximum garlic mustard frequency in a single transect which was judged to be maximally infested with garlic mustard in 2010 was measured at 34 intersections over 20 point intercept measurements, which would correspond to approximately 1,700 garlic mustard intersections per 1,000 measurements. If a crude estimate of a doubling in garlic mustard frequency per season is accepted from the data presented in Figure 3, then the maximum frequency of garlic mustard will be reached on the study site in approximately four years, or the summer of 2014, which is in reasonable agreement with the prediction that all permanent transects could contain measurable garlic mustard between 2012 and 2014. While these predictions are based on limited data, only three seasons, they do emphasize the rapidity in which garlic mustard is likely to invade the area. Plans for the future of this investigation are to continue to monitor the study site for garlic mustard in an effort to produce robust models of garlic mustard invasion.

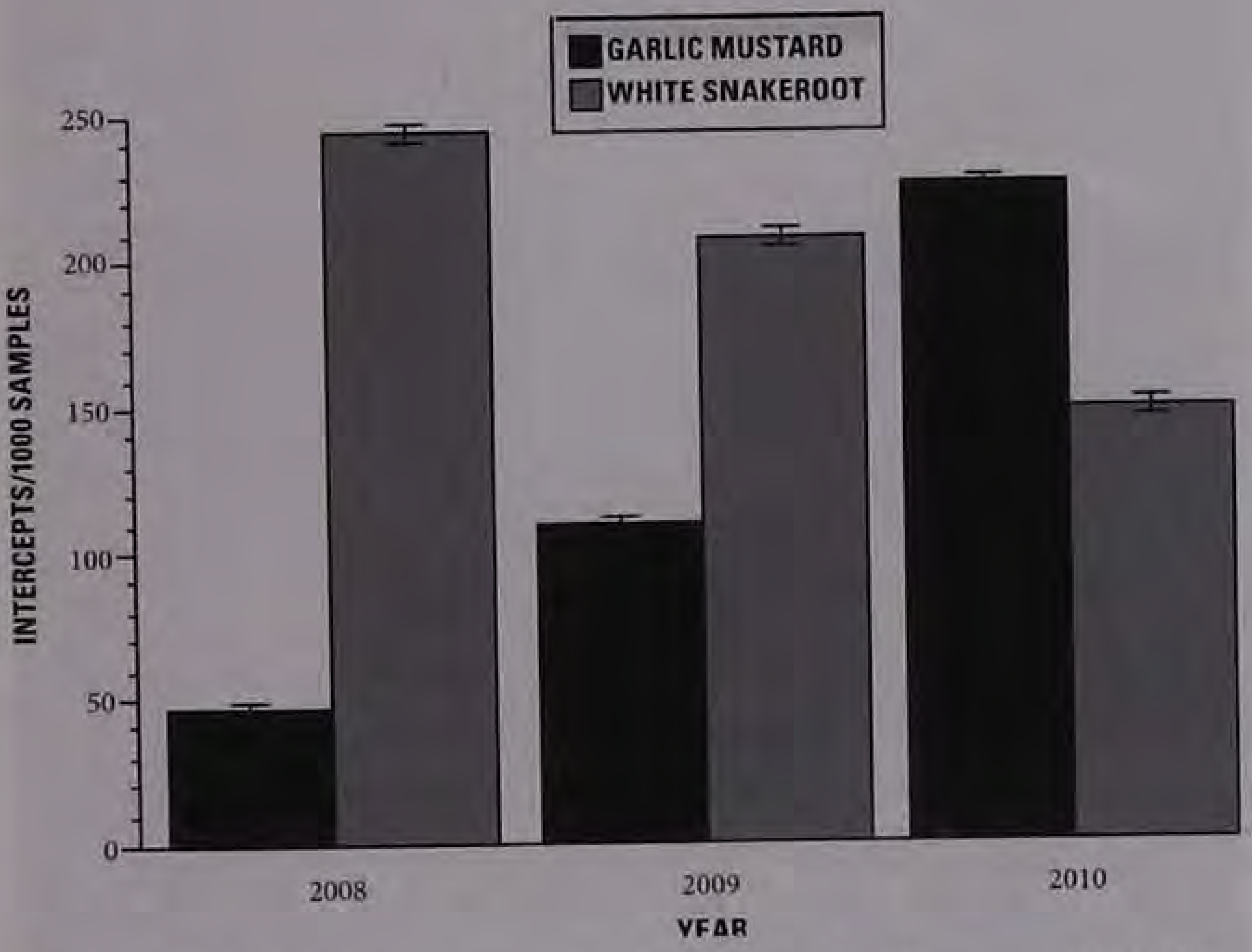

Figure 3. Garlic mustard frequency. Frequency of garlic mustard (black) and white snakeroot (gray) during summers of 2008 to 2010. Data were normalized to 1000 point intercept measurements, and bars represent the mean of intercept data collected for the indicated year. Error bars span one standard deviation. Data evaluated by one-way ANOVA. Garlic mustard significantly different between groups ($p<0.05$), while white snakeroot failed to demonstrate statistical significance ($p<0.60$).

ACKNOWLEDGMENTS

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JAPANESE RASPBERRY (*Rubus parvifolius* L.): AN INVASIVE SPECIES THREAT IN SAVANNA AND PRAIRIE

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Abstract: Japanese raspberry (*Rubus parvifolius* L.) is native to eastern Asia and Australia and has naturalized in several locations in Illinois, Iowa, Massachusetts, Missouri, and Ohio. This species was introduced in North America for food and erosion control, but it appears to be becoming a serious invasive species threat in savannas and prairies. It was found in a former commercial game-hunting farm on Walnut Creek National Wildlife Refuge (now Neal Smith National Wildlife Refuge) in 1991, and was identified as Japanese raspberry in 1995. Japanese raspberry grows vigorously and spreads via rooting from low-arching-to-prostrate canes that are up to 300 cm long, and its seeds can be dispersed by birds. In summer, primocanes are green to purplish green, though they turn reddish brown in winter. It has small pink flowers and bright red fruits. It thrives in shade in a remnant savanna on the refuge, forming rapidly expanding near-monoculture populations. Efforts to control it with herbicide treatment since its discovery have been unsuccessful, but also somewhat sporadic. A second population in a roadside within a mile of the refuge demonstrates its ability to thrive in full sun. County dredging of ditches for drainage improvement may be serving as a vector for its expansion.

Keywords / Search Terms: bramble, naturalization, roadsides, trailing raspberry

JAPANESE RASPBERRY AN INVASIVE SPECIES

In 1991, I (Pauline Drobney) and a colleague, Scott Bryant, were contracted to develop a flora of the proposed Walnut Creek National Wildlife Refuge, now Neal Smith National Wildlife Refuge (Neal Smith NWR). An unusual bramble was found growing within the dripline of a large oak in a degraded savanna in Jasper County, Iowa, being used as a game farm. The plant was densely matted and weakly prickled, and it virtually excluded other vegetation. Initial attempts to positively identify it to species failed. In June of 1995, my coauthor, Mark Widrlechner, identified the species as *Rubus parvifolius*, the Japanese, or trailing, raspberry (Widrlechner 1998).

Difficulty in identification was understandable, given that this species was absent in nearly all North American botanical manuals. Widrlechner noted that the refuge had the dubious distinction of having the largest population he had seen to date.

Japanese raspberry was found in Iowa in 1988 in a roadside bordering the North Central Regional Plant Introduction Station (NCRPIS) farm, southwest of Ames. Widrlechner supposed it had persisted as a result of a past cultivation project, but began to notice it in various other locations in the Ames area, with one population in an Ames woodland spread over about one-half acre. He began an investigation to discover the path of its introduction to Iowa.

A search of all accession records at the NCRPIS failed to yield any evidence of its cultivation there, strongly suggesting that its appearance in Iowa likely predated the founding of the NCRPIS in 1948 (Widrlechner and Rabeler 1991). Notably, Japanese raspberry had apparently escaped cultivation much earlier in Massachusetts (Rich 1908; Knowlton and Deane 1918). It had been grown at Arnold Arboretum at least since 1915, escaped cultivation, and in 1948 was vouchered as growing on a slope there (Palmer 48417, BH). In 1929, seeds of *R. parvifolius* were collected in Japan by P. H. Dorsett and W. J. Morse of the USDA's Division of Plant Exploration and Introduction (Widrlechner and Rabeler 1991). Because of its large red fruits, coupled with the discovery that it was disease resistant (Williams and Darrow 1940) in Maryland and North Carolina, the species became attractive to breeders interested in fruit production (Widrlechner and Rabeler 1991).

Figure 1. Closeup of Japanese raspberry flowers. Photo by Brian Riley, June 1, 2008.

Its establishment in Iowa, however, was most likely related to a project developed by the USDA's Soil Conservation Service (SCS) to find ways to control erosion on sloping land. A Federal Erosion Nursery and Hill Culture Research Station were established by the SCS in the 1930s on the land that was to become the NCRPIS farm in 1948. An SCS accession book and typewritten inventories for an "introduction area" and other reports from this project indicate that cane fruits, including various species of *Rubus*, were being cultivated and studied in Ames during the 1930s and 1940s for their potential to control erosion. A raspberry breeding program at Iowa State College (now University) also reported growing hybrids involving *R. parvifolius* (Maney 1945). However, there are no specific records indicating that *R. parvifolius* was being grown in the field, either by the SCS or Iowa State College (Widrlechner and Rabeler 1991), leaving its origins in Iowa enshrouded in mystery.

In any case, by the 1960s, this species was being distributed for conservation purposes within Iowa by the Iowa Department of Natural Resources State Forest Nursery in Ames (J. Grebasch, personal communication). These distributions may have led to the occurrence of *R. parvifolius* at the savanna at Neal Smith NWR. In July 2010, I learned that this population had originated in about 1967 or 1968, when propagules were obtained from another shooting preserve near New Sharon, Iowa, because of its interesting and tasty berry (S. DeBruin, personal comm.). For two years after planting, the developing patch was tended, but the owners lost interest in weeding and the patch was abandoned. Steve DeBruin, who grew up on the site, believed that it had been planted in a sunny location, but that when weeding ceased, it became overgrown with grass and, eventually, with trees. It is interesting to note that despite early cultivation and apparent success in growth, the patch did not necessarily persist in the original location, as the observation of it in 1991 was under the dripline of a large, old oak, and thus not in the open sunny position where it began.

Figure 2. Closeup of Japanese raspberry fruits. Photo by Pauline Drobney.

Japanese raspberry is native to Japan, Korea, China, and southeastern Australia (Naruhashi 1987). When in fruit, this species is obviously a raspberry (subgenus Idaeobatus) with clusters of drupelets that can easily be removed from the central receptacle (or torus), forming a juicy red thimble. Compound leaves, whitened beneath, are born on trailing primocanes (first-year stems) that root at the tips and nodes. Small rose-pink flowers bloom on floricanes (second-year stems) from late May to early June in Iowa, with fruit set in late June into July. These characteristics (see Figures 1 and 2) and its distinctive tangled, mounding growth habit make it easily distinguishable from Iowa's native *Rubus* species. (Both *R. parvifolius* and Iowa's native species are described and keyed in Widrlechner (1998). However, its tangled, mounding habit does resemble a different nonnative bramble occasionally found in Iowa, *Rubus caesius* L., the European dewberry (Widrlechner and Wagner 1998). Since 1998, *R. caesius* has been noted at an increasing number of sites in central Iowa, where it may pose another invasive threat, but this is not the focus of our current presentation.

Another rose-pink-blossomed, red-fruited bramble, native to the Midwest (but not Iowa), is R. odoratus, the purple-flowering raspberry, a member of the subgenus *Anaplobatus*. But it would be difficult to confuse it with *R. parvifolius*, since it has upright, unarmed canes, simple leaves, and large showy flowers.

In addition to its occurrence at the NCRPIS and Neal Smith NWR, *R. parvifolius* was first collected in Iowa in 1954 in Cherokee County and later from other sites in Page, Story, and Taylor Counties, as reported by Widrlechner (1998). More recently, it has been collected at Black Hawk Point Wildlife Management Area in Allamakee County (M. J. *Leoschke* 2659, 12 Jun 2007 ISC), along Tunnel Mill Road in Hamilton County (J. D. Thompson s.n., 27 Jun 2002, ISC), at Harmon Lake Wildlife Management Area in Winnebago County (M. J. *Leoschke* 2897, 9 Jun 2010, ISC), at Dewey's Pasture Wildlife Management Area in Palo Alto County (M. J. *Leoschke* 2928, 26 Jul 2010, ISC), and at Elk Creek Marsh State Wildlife Management Area in Worth County (C. Hall s.n., 16 Oct 2001, ISC). In addition to herbarium vouchers, in 2009 Widrlechner received two photographs of this species taken on parkland near Summerset State Park in northern Warren County. Within the Midwest, it has also been found as an escape from cultivation in Illinois (Widrlechner and Rabeler 1991), Missouri, and Ohio.

After discovering the plant at Walnut Creek NWR in 1991, and becoming Refuge Biologist a year later, I periodically observed its status for several years, noting that the population was expanding and seemingly excluding native herbaceous species. Upon learning its identity and status as an exotic species from Widrlechner, I became concerned about its invasive potential.

In fall of 2001, refuge pesticide records indicate that the staff treated this population using a backpack pump and glyphosate, noting that summer treatments at concentrations lower than 7% were ineffective, as was 1% to 2% triclopyr. Five treatment sets (treatment was not always accomplished on a single day) occurred in June, July, August, and early September with treatment intervals ranging from approximately two to five weeks. Treatment was considered successful if green leaves turned brown and seemingly lifeless, and the plant was considered dead.

Revisitation of the Japanese raspberry site in 2006 confirmed a population near the original location, and a second, clearly separate population was discovered within 100 m to the north. However, small isolated patches on the refuge of *Lespedeza cuneata* (Dum.-Cours.) G. Don, commonly known as sericea lespedeza were increasing in size and being found in several new locations at this time. Treatment had not been effective and a rapid and immediate response was required by refuge staff. With this shift of invasive-species control priorities, Japanese raspberry was left unchecked.

In 2006, I observed a cane of Japanese raspberry growing in a recently dredged roadside within 0.1 mile of the former game farm. A robust, dense mat of this species was noted in the same roadside approximately 0.5 mile south of the dredged area, and also existed in terraces within the adjacent crop field. The source of the newly establishing cane in the roadside is uncertain, but it is possible that it was actually a seedling resulting from bird dispersal, or that the cane was transported by dredging equipment. It may have been already present in the ditch before dredging took place, although this cane did not seem to be well established. In any case, the potential danger of mechanical transport is apparent. Concern about this danger is amplified because soil taken from dredged roadsides in the local area, including a portion of roadside immediately adjacent to the obviously infested area, was stockpiled for use as fill for projects where rural bridges are being replaced by large culverts. If Japanese raspberry was present, soil movement in the manner described could rapidly spread it to other sites.

Table 1. Area of Japanese raspberry patches in 2010.

PATCH NUMBER	1	2	3	4	5	6	7	8	9
AREA M^2	48	706	705	96	155	340	179	283	23

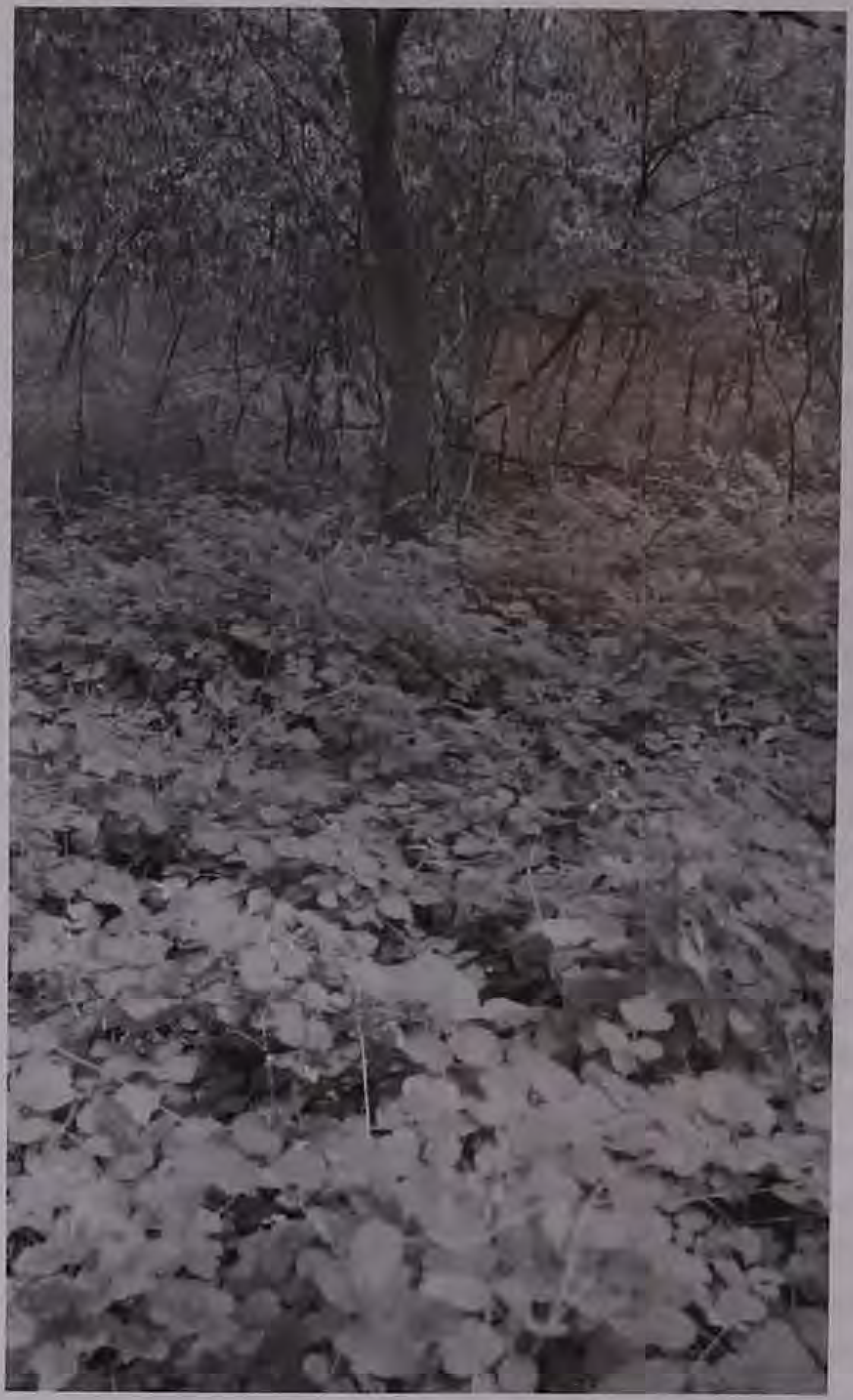

Figure 3. Japanese raspberry in a savanna at Neal Smith National Wildlife Refuge in 2010. Photo by Pauline Drobney.

In early July 2010, the savanna site was revisited, and nine Japanese raspberry patches, five of which are in relatively close proximity, provided evidence of substantial expansion of Japanese raspberry (Figure 3).

Using measuring tapes and extrapolation for odd shapes, current coverage by *R. parvifolius* at Neal Smith NWR is roughly estimated at 2,535 square meters. This is a large increase from an estimated 254 square meters known coverage of Japanese raspberry in 1992. In 2010, the largest patch was thriving in a portion of the savanna where all trees had been removed, and it had expanded into an adjacent cornfield. This area had been treated with 7% Garlon 4 (triclopyr) combined with 2% methylated seed oil applied with a backpack sprayer in June 2010. Treatment of the remaining area had not yet been accomplished, impeded by downed trees and excessive rain.

Where the triclopyr application was most effective, canes appeared black with little to no living foliage. Where the Japanese raspberry was growing in full sun, untreated canes were producing large red berries. At the transition between treated and untreated areas, leaf cupping and yellowing typical of herbicide treatment was evident, though the minimally damaged canes will probably recover, and fruits on these canes sometimes remained bright red and hydrated. Treatment of a cane did not guarantee death of untreated portions, likely because of rooted nodes on the canes within the untreated area. Abundant rain in 2010 may have contributed to an easier recovery than might be expected in times of normal or below average precipitation.

Plant vigor, as well as fruit and flower production of live portions of the population that extended into the woodland, declined with reductions in light. The dense mounds of canes, however, still impeded or excluded growth of native herbaceous vegetation. In areas where visual estimates of woody canopy coverage were 80% to 95%, canes continued to be present, but were much thinner and no flowers or fruits were observed. In these areas, native herbaceous plants were present but also sparse. Widrlechner observed a similar phenomenon when revisiting a woodland population of *R. parvifolius* in Ames, which had been quite vigorous in the late 1980s when the tree canopy was rather open, but which declined considerably as the canopy closed.

The roadside population was also expanding, though it was not yet a mat, and had abundant herbaceous growth of other species interspersed with the canes. Interestingly, by August of 2010, half of the dense population of Japanese raspberry in the roadside had been sprayed by a local landowner targeting small trees. Though the chemical used for this treatment is unknown, the treated portions appeared dead, but canes immediately adjacent to the treatment continued to support live foliage.

Herbicide treatments with glyphosate and triclopyr severely damaged Japanese raspberry, and, with post treatment vigilance for new seedlings and/or plants regenerating from existing crowns or canes, such treatments could control or eliminate these invasive plants. However, "successful" herbicide applications are likely to damage any adjacent or interspersed desirable plants. Although fire management has not been tested as a control for this species, frequent fire treatments in the presence of a sufficient fuel matrix reduce populations of some woodland *Rubus* species, in my experience, and could likewise be useful in controlling *R. parvifolius*. Experimentation is needed to test this possibility. Additionally, if this invasive is found in the context of a savanna restoration project, managers may be wise to consider postponing tree-thinning until Japanese raspberry is controlled or eliminated, as it grows more vigorously in higher light levels. Alternatively, an aggressive program of thinning to promote development of a fuel matrix, careful herbicide treatment of infested areas, and repeated annual burns may also be effective, though again, these ideas are untested.

In conclusion, the presence of *R. parvifolius* has been documented in ten counties in Iowa, and in three other midwestern states. It is likely that additional populations are undetected due to misidentification and lack of awareness. It is a species that spreads rapidly and can tolerate 80% or more shade, though it is more vigorous and healthier at higher light levels, thus posing a greater threat to sparse prairies and open savannas. Plants reproduce vegetatively and produce attractive fruits. Populations can be spread both by land-management equipment and wildlife. In our observations, Japanese raspberry reduces diversity and density, at least in native, open woodlands and roadsides and, if left unchecked, can be difficult to control. More work and study of this species is necessary to discover treatments for effective control while preserving native vegetation on infested sites. We believe that educational programs and websites related to midwestern invasive species should add information about Japanese raspberry to help limit the expansion of this invasive species and educate both land managers and gardeners about its invasive potential.

JAPANESE RASPBERRY (*Rubus parvifolius*)
MANAGEMENT WATCH-OUTS
PLANT CHARACTERISTICS

- decumbent, mounding growth habit
- weakly prickly canes
- small rose-pink flowers
- large, bright red fruits (size of commercial raspberries),
- obovate to broadly sub-rhombic central leaflets with obtuse tips
- compound leaves, whitened beneath

HABITAT CHARACTERISTICS

- full sun, partial sun to moderate shade
- semi-xeric to wet-mesic sites (not a wetland plant)

THREAT

- formed near-monoculture excluding other plant species under favorable conditions
- seed dispersed by birds
- ready rooting and spread by nodes on primocanes
- nodes and seed dispersed by equipment

CONTROL

- Current information is scant.
- Effective herbicide treatments need to be tested and documented.
- Some *Rubus* species are reduced with frequent fire, but control of Japanese raspberry using fire is untested.
- Mechanical control is untested but likely to be difficult or ineffective, due to ease of recolonization via crown persistence and nodal rooting.

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SEARCH, SMELL, AND SPRAY:
A STRATEGY FOR MONITORING AND CONTROLLING *Lespedeza cuneata*

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The findings and conclusions in this article are those of the author and do not necessarily represent the views of the U.S. Fish and Wildelife Service.

Abstract: Neal Smith National Wildlife Refuge is a prairie reconstruction project, initiated in 1991, which includes 1,376 ha (3,400 acres) of diverse prairie plantings. Invasive sericea lespedeza (*Lespedeza cuneata*) has been found in these plantings. This species poses a serious threat because it is adapted to fire, quickly forms a deep root system, and spreads to form dense patches. In 2006 an extensive effort was begun to locate and eradicate sericea lespedeza on the refuge. The infestation was at a low density but widespread throughout many of the plantings, probably transferred by harvesting seed from areas containing sericea lespedeza. Monitoring has been conducted annually by walking through plantings after burning and marking sericea lespedeza plants and patches using Global Positioning System (GPS). As a control method, sericea lespedeza is spot-sprayed with a 2% solution of triclopyr. Locations are revisited to treat new seedlings and plants not killed by previous treatments. Locating small, scattered plants is a challenge because growth of sericea lespedeza coincides with that of native prairie plants, making it difficult to see among prairie vegetation. Sericea lespedeza is most visible in fall when it stays green after warm-season grasses begin to cure; however, by this time viable seed is already present. In 2010 specially trained dogs were deployed on the refuge to locate sericea lespedeza by scent. Dogs are able to detect plants that humans might not see, allowing earlier detection of young plants and new infestations. Dogs are also able to detect plants from greater distances, making search efforts faster.

Key Words/Search Terms: sericea lespedeza, invasive, detection dog

INTRODUCTION

Perhaps the most challenging part of prairie restoration and reconstruction is managing invasive species. At Neal Smith National Wildlife Refuge (NWR) a particularly problematic invasive plant, sericea lespedeza (*Lespedeza cuneata*), has been found in native prairie reconstructions and is being managed using a variety of techniques. Neal Smith NWR is a large-scale ecological restoration project located in central Iowa and initiated in 1991. Much of the refuge was formerly cropland. Beginning in 1992, fields were retired from crops and planted with local ecotype seed originating from local remnants. To date 1,376 ha (3,400 acres) have been planted with native prairie plants.

In the early stages of the reconstruction work, sericea lespedeza was recognized as a potential high-priority invasive species if it were to be found on the refuge. Sericea lespedeza has several traits that make it invasive in prairie systems. It is a warm-season perennial with a growing season that roughly coincides with native warm-season grasses. In central Iowa, it usually emerges in late May, blooms August through October, and begins seed production in early October. Sericea lespedeza remains green until a hard frost, usually after warm-season grasses have senesced. This characteristic makes it easier to locate visually in the fall, although by this time it may already be producing seed. Sericea lespedeza is a nitrogen-fixing legume, which allows it to thrive in many soil types. It has deep roots, making it difficult to hand-pull effectively. It has high tannin content and is not palatable to most livestock. Sericea lespedeza is known to exhibit allelopathic characteristics (Kalburtji and Mosjidis 1993), inhibiting the growth of other plants. It can self-pollinate, and the seeds may remain viable for 20 years or more, although this contention has not been verified by research (Munger 2004). Fire, which may be used to control some invasive plants, scarifies the seed, resulting in a flush of new seedlings. In addition, sericea lespedeza has few insect or disease pests in North America. The seeds are known to be ingested by birds or mammals, but transporting of seed through haying or other equipment is thought to be the primary method of dispersal.

In the late 1990s, sericea lespedeza plants were seen in two units of the refuge but could not be found when staff returned to search for them. It was thought that they had been killed through trampling by bison in one of the units. The refuge practice of machine-harvesting reconstructed prairie on the refuge for seed to plant in newly retired crop fields apparently resulted in the unintended harvest and planting of sericea lespedeza seed. In October 2003, sericea lespedeza was found in an area being harvested and reported to the refuge biologist. It was sprayed within 24 hours using triclopyr amine. The plants were blooming and no fruits were found. In the fall of 2004, patches of sericea lespedeza were found in an area that had been planted the previous fall. This reconstruction was sprayed with clopyralid, an herbicide that is somewhat selective for legumes and composites. Patches found after seed development were clipped by hand and removed from the reconstruction. Also in fall 2004, staff and volunteers searched other reconstructions planted from fall 2003 through spring 2004 using the same seed mix, and were not able to find any additional sericea lespedeza. In the following two years, more plants were found in areas where sericea lespedeza had been treated with herbicide, as well as in additional reconstructions. In 2006 staff recognized that planting seed that was recovered from machine harvest of infected fields probably contributed to the spread of sericea lespedeza on the refuge. As a result, in 2007 refuge staff stopped the machine harvest of seed on-refuge, and destroyed machine-harvested seed in storage. A plan was developed to clean vehicles and equipment transferred between units of the refuge to avoid transporting seed.

Several management options for controlling sericea lespedeza were considered. Hand-pulling was not viewed as a viable option because the deep roots make pulling ineffective. There are no known biocontrols, and they are not likely to be developed because sericea lespedeza is grown as a crop in some areas of the United States. Burning does not kill mature plants, although in some situations it may control seedlings. Since fire scarifies the seeds, it can be used to exhaust the seed bank more quickly. Mowing or grazing may reduce plant height and seed production, but does not kill plants, and mowing has been demonstrated to increase stem density and canopy cover of sericea lespedeza (Brandon et al. 2004). The only known effective method of killing sericea lespedeza is herbicide treatment. Because the population at Neal Smith NWR is believed to be a low-density and diffuse infestation, it was decided that spot spraying with herbicide would likely be the best method of treatment.

METHODS AND MATERIALS

BURNING AND USE OF HERBICIDES

In 2008 refuge staff began a more extensive program to search for sericea lespedeza. Refuge management units were prioritized based on the planting history (units planted with seed harvested from a planting known to have sericea lespedeza) or units known to have some sericea lespedeza plants. These units were burned to make walking and seeing plants easier, and to express the seed bank. Five summer interns and some volunteers searched for the plants by systematically walking 5-10 meters apart, in a pattern covering the entire field, while visually scanning the vegetation. Whenever a sericea lespedeza plant or patch was found, its location was marked using GPS and the plant was sprayed with herbicide. Most of the units searched were found to contain sericea lespedeza plants.

Herbicide selection was based on a current literature search and personal communication with land managers experienced in treatment of sericea lespedeza. The herbicide selected for use was a 2% solution of triclopyr ester (Koger et al. 2002, Menard p.c. 2006, Munger 2004). Fall searching was effective at locating plants not seen earlier in the year because the green foliage of actively growing sericea lespedeza was in striking contrast to the tans and browns of most late-season prairie species. Plants found after seed production began were clipped, bagged, and removed from the unit after marking their locations using GPS. These plants were treated the following growing season, thus requiring a two-year work commitment. This method was repeated in 2009.

The 2% triclopyr ester solution appears to be effective at killing sericea lespedeza, with most plants appearing dead within a week of spraying. However, triclopyr kills most forbs, so it must be applied carefully to avoid harming native species. The year after plant treatment, especially after a burn, seedlings typically emerge, so each location needs to be revisited each year. New plants and any plants that had survived or been missed by previous treatments were sprayed. We found that some patches had spread to a large enough area that backpack sprayers were not efficient, so a spray tank mounted on an all-terrain vehicle was necessary to treat these patches. This allowed the spraying of a larger area, although it was still difficult to locate and kill every plant in a patch.

WORKING DOGS AND LOCATING INVASIVE SPECIES

One of the crucial steps in controlling invasive species in the early stages of invasion is locating isolated plants. As one approaches total control of sericea lespedeza in a unit, the ability to detect isolated plants becomes increasingly difficult. Small plants are often obscured by larger vegetation, making it nearly impossible to detect all plants. However, detection of these plants is crucial in order to complete the treatment cycle. In an attempt to increase the detection rate for these small, isolated plants, refuge staff consulted with Working Dogs for Conservation, a nonprofit organization based in Montana. This organization has successfully used dogs to identify and detect invasive plants, as well as other biological targets (Cablk and Heaton 2006, Goodwin et al. 2010, Smith et al. 2003).

In fall 2009 the Friends of the Prairie Learning Center agreed to fund the deployment of Working Dogs for Conservation at the refuge. Because the dogs had already been shown to be effective in locating low densities of invasive species (Goodwin et al. 2010), the intention of the deployment was not to conduct a scientific investigation on the effectiveness of using dogs for this purpose, but to apply the method over a broad area. It was also the first deployment of these dogs in a tallgrass prairie environment. In fall 2009 refuge staff selected dog-training areas known to have low densities of sericea lespedeza. In these areas, known sericea lespedeza plants were not sprayed, but were clipped and bagged in fall to remove any viable seed. In other areas of the refuge sericea lespedeza plants to be used for training were either propagated by stem cuttings or were dug from the field as small plants. These were grown in a greenhouse in pots. Training sites were burned in spring 2010, prior to the arrival of the dogs. This made the areas easier for dogs to walk through and to detect scents that can be impeded in taller vegetation.

Two handlers and three dogs from Working Dogs for Conservation arrived 28 May 2010. They began training immediately, using greenhouse plants and cuttings from newly emerged sericea lespedeza plants in the field. Dogs were also taken to areas where the locations of sericea lespedeza plants were known to the handlers. As a final test, the dog/handler teams searched designated training sites for a double-blind test. The teams were able to locate more sericea lespedeza plants than had been found the previous year using visual identification by experienced observers (see Table 1). Some plants may have been seedlings that emerged after the spring burn, but may also have been plants that were missed during the fall search. Only two known plants from the previous year were not found by the dogs, and one of these could not be located when refuge personnel searched the area.

With the successful ten-day training period completed, the dog/handler teams were sent to search selected units. One of the dogs completed training but was removed from the program because of health issues. The selected search units had either not been searched at all or had been searched and evaluated as having only a low density of sericea lespedeza plants. Each handler walked a systematic grid perpendicular to the wind, with the dog moving left and right across the handler's transect within 10 meters of the handler. Search transects were 7-35 meters apart, based on wind speed, vegetation height, and the likelihood of small individual plants being present in more recent plantings. When plants were located, they were marked with a flag and the location recorded using GPS. Data on detections were recorded, including time and date, whether a single plant or a patch was detected, distance that the dog changed behavior, and whether the plant was detected by the handler or the dog. The two teams searched 371 ha (917 acres) in eight days. The handlers walked 155 km, tracked using GPS units. They spent 63 hours searching and located 34 individual plants and 27 patches of sericea lespedeza. The dogs were able to detect sericea lespedeza plants from up to 40 m away, with a detection distance of 1-15 m (mean 5.5 m) for individual plants and 1-40 m (mean 12.9 m) for patches of plants. Although the handlers were also searching for plants as they walked, the dogs detected 23 of the 27 patches and 23 of the 34 single plants.

Table 1. Sericea lespedeza plants found by dogs in training sites (Hurt and Whitelaw 2010).

NAME OF DOG	NO. PLANTS KNOWN TO BE IN AREA	NO. PLANTS DOG TEAM FOUND IN AREA
Tsavo	1	1
Tia	3	6
Wicket	2	4
TOTAL	**6**	**11**

DISCUSSION AND CONCLUSIONS

Overall, the use of dogs to locate sericea lespedeza plants was considered to be a success. Specially trained detection dogs were able to locate small plants from greater distances than refuge personnel could locate them. They were able to locate plants obscured by vegetation that would have been overlooked by human observers. The smallest plant located was only three inches tall and was not found by the handler until repeated alerts by the dog, after which the handler located the plant by examining the area on hands and knees. On several occasions dogs detected sericea lespedeza plants that were not noticed by the handler walking within two meters of the plant. Detection of young plants and individual plants before the patches spread provided for earlier treatment, or early detection and rapid response. The ability to search a larger area in a shorter period of time meant that treatment could be applied earlier in the season when it was most effective and less herbicide was needed to cover the plants. Another added benefit was in the public relations arena—a local television station did a story on the dogs, raising awareness of the importance of invasive species control.

There were some limitations to using this technique to locate invasive species: one of the greatest was the expense. If the nonprofit Friends of the Prairie Learning Center had not funded the program, it would not have been possible. Another limitation was weather. Dogs were best able to detect scents on breezy days without precipitation. Central Iowa had higher than normal precipitation in June, so many days the dog/handler teams had to work between rain showers, and days off had to be rearranged because of weather. As the season progressed, vegetation grew quickly, and by the time the teams completed work on 19 June, much of the vegetation was too tall to locate plants effectively. Dogs need to have their noses above the vegetation to detect scents over it, and it is difficult for a dog to hold its head high as it runs over large areas for several hours each day. It was not possible to start the season any earlier because the sericea lespedeza did not emerge until late May.

The cost to have two dog/handler teams could be prohibitively high, but there are other factors to consider. Dogs are able to search for multiple targets, so if they were trained to search for another rare plant (invasive or native), they could be performing double duty. Dogs are most effective in searching for low-density targets, and could be used to search for rare plants, burrows, scat, wildlife species, etc. Since these dogs have now been trained to search for sericea lespedeza, training time will be greatly reduced in future years, making their deployment more cost-effective. The ability to find and spray sericea lespedeza plants now while they are still at low densities will save the refuge time and expense in the long term.

Through the past several years, refuge staff has learned some valuable lessons regarding control of sericea lespedeza that may apply to other invasive species and other locations. One of these is the value of preventing invasive species from being introduced and established. Had this species been located earlier and prevented from becoming established, the refuge would not be in the position it is in today. Early detection and rapid response are necessary to control invasive species before they spread. Land managers should be prepared for invasive species by knowing which new invasive species are likely to have an impact upon natural areas and by considering all the possible ways they can be carried to sites, including by hikers, bikers, and hunters, as well as by equipment exchange for management such as prescribed fire. They should also be prepared to act quickly to recognize and control new invasive species. A plan of action to react to new threats will help in the long term. An integrated approach should be considered, including various methods of detecting and treating new invasive species. In this situation, a combination of prescribed fire and chemical control has been effective in controlling and reducing sericea lespedeza. Refuge personnel using GPS units and detection dog teams have combined to locate and monitor sericea lespedeza. The deployment of dogs to search for invasive species is a viable option for locating low-level infestations of invasive plants, which is critical in gaining control of invasive species.

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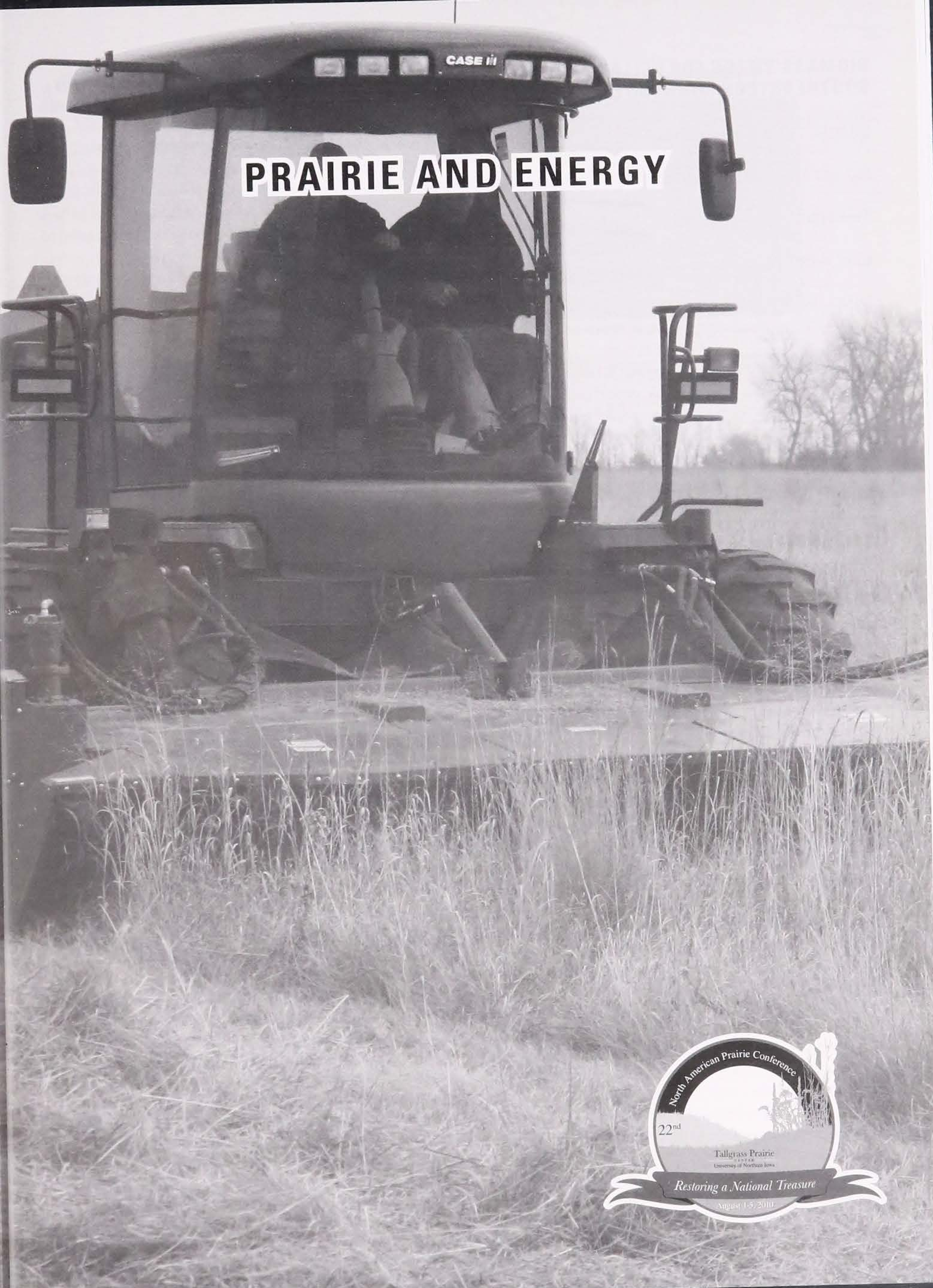
PRAIRIE AND ENERGY



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BIOMASS YIELDS FOR PELLET PRODUCTION FROM RECONSTRUCTED PRAIRIE PATCHES IN SOUTHEASTERN MINNESOTA

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Abstract: The utilization of marginal farmland for prairie reconstruction may offer tangible opportunities for developing renewable energy systems while restoring habitat and ecological services on agricultural landscapes. Concurrently, pellet production from prairie biomass may be a promising approach to sustainability in this region of the Upper Midwest. This work aims at verifying the feasibility of a pellet production industry in Winona County, Minnesota, that could become an incentive for landowners to convert marginal farmland back into natural prairie habitat while developing sustainable energy systems for local farms and neighboring human communities. Our work is an ongoing effort on a farm in southeastern Minnesota where marginal farmland was converted into native prairie in 2007. Biomass yields from corn stover and prairie patches planted with mixed grasses or grasses with forbs were measured in order to verify initial data that Tilman and his collaborators collected in 2006. A single-way ANOVA employed to analyze our data for 2009 indicated that there was a significant difference between yields for the three types of biomass. Scheffe's *post hoc* test was also employed to further verify which cropping system was different when compared to the other two. Our data indicated that prairie plots with mixtures of grasses and those with a mixture of grasses and forbs can be valuable for biomass production, thus supporting the yield data measured in 2008.

Key Words / Search Terms: biomass, pellet, prairie reconstruction, renewable energy, sustainability

INTRODUCTION

For several decades, modern farmers have been supportive of a cultural, technological, and global agricultural paradigm that threatens to end the era of small-scale farming in the U.S. (Rigby and Caceres, 2001). Increasing costs of food production attributed to the recent recession in the global economy and the fluctuating affordability of fossil fuels (Borsari et al., 2009) have exacerbated the agricultural crisis a step further. In order to counteract the unpredictable availability and price of fossil fuels, farmers in the U.S. have been encouraged to grow more corn to produce ethanol, which emerged about a decade ago as a promising renewable-energy crop. However, this conversion has led to a significant rise in corn prices, and consequently food costs, affecting a large segment of consumers and the whole agriculture industry (Leibtag 2008). More negative consequences of this approach have been responsible for an extirpation of prairies. Nonetheless, many farmers have been planting corn year after year, which inevitably has led to more soil erosion, worsening soil quality (Reicosky et al. 1995) and the accelerated demise of natural habitats like prairies.

Corn cultivation in the U.S. grew from 31.9 million hectares (79 million acres) in 2002 to 37.6 million hectares (93 million acres) in 2007 (Michael Osterholm, personal communication, July 19, 2007), accompanied by a loss of interest among landowners for land conservation projects and programs. Soil erosion remains a major threat to the viability of agricultural systems despite the estimate of soil loss by Pimentel et al. (1995) to be near 10 million hectares per year, worldwide. Enthusiasm for ethanol from corn has led also to an increasing use of marginal farmland for growing this crop, which exacerbates soil loss a step further, since marginal land includes areas where the soil is prone to significant erosion and fertility is limited (Brady and Weil 2002). Consequently, these forces have caused many small-scale farmers to abandon agriculture and search for other means of employment.

However the public desire and need for sustainability in modern agricultural systems is rapidly increasing. This emerging trend is persuading the farming industry to rethink many agricultural practices and to encourage innovative small-scale ventures that meet the terms of sustainability as defined by Smyth and Dumanski (1993)

Sustainable land management combines technologies, policies, and activities aimed at integrating socio-economic principles with environmental concerns, so as to simultaneously: maintain and enhance production (productivity), reduce the level of production risk (security), protect the potential of natural resources and prevent degradation of soil and water quality (protection), be economically viable (viability) [and], [Be] socially acceptable (acceptability).

Many farmers in southeastern Minnesota are able to subsidize heating bills during the winter by burning corn stover. However, the most significant drawback to this method is that fields with 40% or more of the corn stover removed may lose as much as 560 kg/ha of soil per year (Comis and Perry 2009). With the same equipment and methods used for harvesting and burning corn stover, native prairie mixtures can be pelletized and employed as a feasible and renewable source of energy. This approach seems much more sustainable than corn stover for several reasons. The use of prairie grasses for burning allows corn stover to be left in the field, which decreases soil erosion rates significantly (Montgomery 2007). Furthermore, these types of low-input, high-diversity native prairie grass mixtures have been producing 51% more usable energy per hectare on degraded land than ethanol from corn produced from fertile soils (Tilman et al., 2006).

The extensive root systems of prairie perennials grown in polycultures improve water and soil quality, while adding to the overall ecological diversity of the farm (Tilman et al. 2006). This effort enhances a more stable and reliable ecosystem in which agricultural land is better preserved, while the operating costs of production are reduced (Kintisch 2008). The extensive root systems of native prairie grasses also increase the amount of organic carbon in the soil, and sequester atmospheric CO_2, which is believed to contribute to global climate change (Omonode and Vyn 2006). Reestablishing prairie also enables a farm to sequester as much carbon as it would produce with other fossil fuel-based machinery (Tilman et al. 2006). Thus, this type of carbon neutrality may empower farming systems as they attempt to wean themselves from a heavy reliance on nonrenewable oil (Rigby and Caceres 2001).

The purpose of this study was to measure and the yields of different kinds of biomass derived from native prairie systems and compare them to those of corn stover. Higher yields would be expected from plots with more biodiversity, as described by Tilman and his collaborators in 2006. A comparison of yields allowed us to speculate about the feasibility of a prairie pellet production industry in southeastern Minnesota as a competitive form of sustainable energy. Despite its limitations, this study could demonstrate innovative economic venues for farmers while restoring ecological services that had vanished with large-scale modern agriculture.

MATERIALS AND METHODS

In the spring of 2007, 4.7 hectares of mixed grasses and 3.2 hectares of mixed grasses and forbs were planted on marginal farmland at Pork and Plant, which is a 16.2-hectare farm located in Elba, in southeastern Minnesota (Winona County). The native prairie grasses were drilled directly into the soil where the reconstruction of prairie patches had been planned (Figure 1).

Eight plots varying in surface area were planted, five of these with a mixture of prairie grasses, and three with grass and forb seed (Table 1).

Figure 1. Aerial photograph of Pork and Plants Farm showing the eight prairie plots planted in 2007 and their boundaries.

Plots were divided by physical markers that would allow them to be easily distinguished for future harvests. The areas were measured with a handheld GPS unit (Garmin, 2004). The plots were initially mowed in early September 2007, to discourage growth of invasive species, thus facilitating the establishment of the native plant community.

Table 1. Assigned plot numbers with hectarage for prairie mixtures of grasses and grasses and forbs.

PRAIRIE MIXTURE	PRAIRIE GRASSES					GRASSES & FORBS		
PLOT #	1	2	3	4	5	6	7	8
HECTARES	0.7	1.5	1.1	1.1	1.2	0.9	1.0	0.5

The harvest for the growing season 2009 took place in late October and early November, and it was initiated after a killing frost, in order to achieve the highest yield of biomass, as Mulkey et al. (2008) recommended for crops to be converted into pellets. The plots were mowed on October 31 and the biomass was left to lie on the field for a week. Then it was baled and weighed with scales obtained from Winona County Soil and Water Conservation office. The scales were Intercomp wheel load scales (Intercomp 2008). A skid loader was used to carry the bales onto the scales. The weight of the bale was determined by subtracting the average weight of the skid loader from the combined weight. The weight of the skid loader was taken before and after the weighing of each bale, and these two weights were averaged in order to account for any changes in the weight of the skid loader during the weighing process. The bales were then pelletized, and the pellets were later burned to provide heating for the greenhouses at the farm. Table 2 shows the species of plants originally planted in 2007 (Borsari and Onwueme 2008).

Table 2. Species planted in 2007 at Pork and Plant Farm. (Borsari and Onwueme 2008).

NATIVE GRASSES SCIENTIFIC NAME	PURE LIVE SEED, KG	NATIVE FORBS SCIENTIFIC NAME	PURE LIVE SEED, KG
Big Bluestem *Andropogon gerardii*	22.7	Long Head Coneflower *Ratibida columnifera*	0.3
Indian Grass *Sorghastrum nutans*	16.3	Maximilian Sunflower *Helianthus maximilianii*	0.3
Little Bluestem *Schizachyrium scoparium*	4.7	Partridge Pea *Chamaecrista fasciculata*	0.3
Side Oats Grama *Bouteloua curtipendula*	7.9	Black-eyed Susan *Rudbeckia hirta*	0.31
Blue Grama *Bouteloua hirsuta*	4.5	White Prairie Clover *Dalea candida*	0.42
Green Needle Grass *Stipa viridula*	9.0	Oxeye Sunflower*Heliopsis helianthoides*	0.42
Switch Grass *Panicum virgatum*	7.2	Purple Prairie Clover *Dalea purpurea*	0.25
Slender Wheatgrass *Agropyron trachycaulum*	9.0		
Virginia Wild Rye *Elymus canadensis*	9.0		
TOTAL:	90.8		2.3

During the harvest of the 2008 growing season, the fields with the grasses and forbs mixture were flooded by heavy precipitation, which prevented harvest of the biomass. In order to collect data for comparison, the adjacent DNR restored prairie was harvested and weighed (DNR Mix 08 in Figure 2); however, that prairie has not been harvested since. The corn stover was harvested from corn grown with standard agricultural fertilization (220 Kg N/Ha, 126 Kg P_2O_5/Ha, 157 Kg K_2O/Ha) for southeastern Minnesota.

RESULTS

Biomass yields were recorded for the mixed grasses and mixed grasses and forbs sections. Biomass from the corn crop was not available in 2009 because of weather conditions during the harvest period, and so the corn stover was left in the field throughout the winter. The following spring (2010), most corn biomass was lost to not warrant biomass measurement. The data from the two cropping seasons are presented in Table 3.

Table 3. Summary statistics for yields, crop species, and plot type.

PLOT	YEAR HARVESTED	MEAN YIELD (Kg/Ha)	CROP TYPE	LAND TYPE (AGRICULTURAL VS MARGINAL)	TIMING OF HARVESTS
Grasses and forbs	2008 2009	1140+780 2450+740	Polyculture	Marginal	Autumn
Prairie grasses	2008	2170+870	Polyculture	Marginal	Summer, autumn
	2009	2930+990			Autumn
Corn	2008 2009	3150+680 N/A	Corn	Agricultural	Autumn

Mean biomass yields for the three different cropping conditions and their standard error (SE) in 2008 and 2009 are shown in Figure 2.

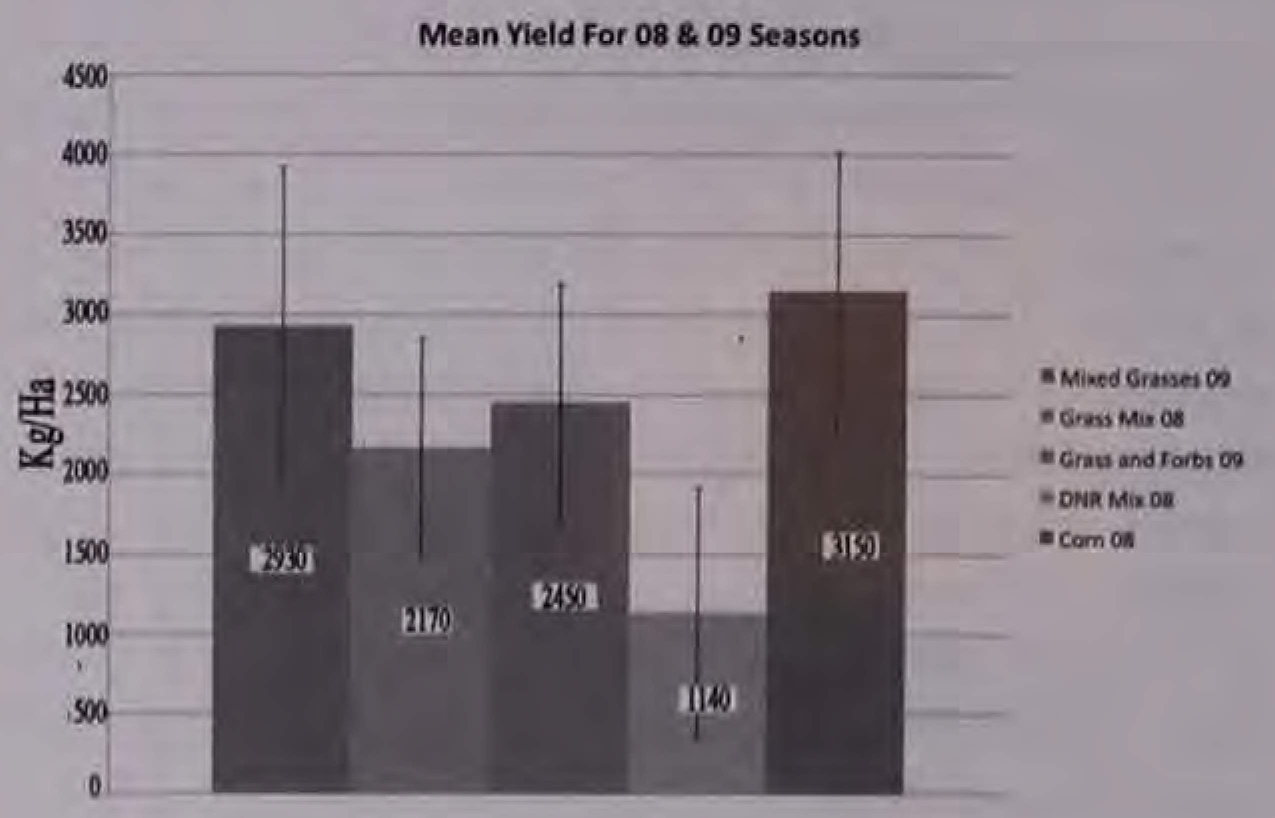

Figure 2. Comparison of yields considering the data (kg/ha) of two consecutive growing seasons (2008 and 2009), with variance (SE) for each crop system.

A single-way analysis of variance (ANOVA) suggested that the difference of yields was significant, F (2, 15) = 3.8, $p<0.05$ among corn stover, mixed grasses, and mixed grasses and forbs. Additionally, a *post hoc* test (Gravetter and Wallnau, 2000) showed that there was a significant difference ($p=0.05$) between the yields of plots with mixed grasses and plots with corn in 2008, as already presented by Borsari and his collaborators (2009). In addition, the yields between the mixed grasses and forbs and the corn from 2009 were statistically different ($p=0.05$). However, no significant difference was found between the two prairie polycultures.

DISCUSSION

The corn stover from 2009 was lost due to inclement weather. We therefore used the yields of 2008 for this crop in order to conduct comparisons at the end of 2009. The assumption that biomass for corn stover remained the same for the two seasons was based on the fact that the main goal of agriculture is to maintain or increase yield from season to season. Working with this assumption, the analysis of our data indicated that there was a significant difference between the biomass of corn and the biomass from both mixed grasses and mixed grasses and forbs. This suggests that native prairie grasses are truly capable of producing as much usable biomass as traditionally harvested corn stover, and the same can be said for the mixture of grasses and forbs.

It is also possible that, by leaving the corn stover in the fields over the winter, the rates of soil erosion might have been lowered at Porks and Plants, as Comis (2009) pointed out, although soil erosion was not considered, in our study. The biomass from both the mixed grasses and the mixed grasses and forbs was found to be significantly higher when each was compared to corn, with more biomass obtained from the mixed grasses than the mixed grasses and forbs. These results do not confirm the outcome that Tilman and his collaborators presented in 2006, indicating that the most-biodiverse plots yield maximum biomass. We think that our results may be explained by the size of our sample fields, as there was more land dedicated to growing mixed grasses than mixed grasses and forbs at our study site. A larger hectarage may have given mixed-grasses plots a biased statistical advantage over the potentially higher yields of mixed grasses and forbs, as the literature suggests (Tilman et al. 2006); however, this remains to be demonstrated. At this time, we can argue only that the decision to dedicate more hectares to mixed prairie grasses at Pork and Plants was made because grassy plant species appeared to establish more quickly on marginal land.

As the project continues and the plants become better established, it is expected that the biomass will continue to increase for both plots, with an increased growth of the grass and forbs mixture.

CONCLUSION

Aside from producing biomass used for heating purposes, prairie reconstruction on marginal land has many additional benefits because the extensive root systems of prairie perennials hold soil in place to reduce soil erosion while improving water retention (Lovell and Sullivan 2008). Strips of natural prairie grasses have also been shown to provide habitat for both pollinators and the natural enemies of crop pests. By fostering increased diversity among natural enemy species, they reduce the need for pesticides in agriculture (Bianchi et al. 2006, Sande et al. 2010, Landis et al. 2000, Patten et al. 2007). When prairie grasses are planted around agronomic crops, they keep pesticides from leaching away, and, depending on the chemical composition of the pesticide, they can metabolize it to less-toxic compounds, eventually eliminating its poisonous effects from the soil (Radkins et al. 1998).

In addition to this, restored prairies have been shown to have the highest rates of carbon sequestration compared to older prairies (Kucharik 2007), thus reducing the amount of atmospheric carbon while also enriching the soil. Also, prairies planted as buffers to agriculture land reduce chemical leaching and prevent runoff of soil, nitrogen, pesticides, and phosphorus into surrounding areas (Lovell and Sullivan, 2006). An agriculture paradigm that relies on native perennials was recently proposed by Jackson (2010) as an imperative provision to his visionary "Fifty-Year Farm Bill." With this proposal, the use of native perennial species, marks a desire to rethink the agricultural practices of the last 150 years, while reconceiving the farm as a natural habitat, in which biodiversity is a clear sign of health and sustainability (Jackson and Jackson 2002). Reconstructing prairie on agricultural land is not only progressive in thought, but beneficial to farmers, farms, and their communities.

Because this research project took place on an operational farm, there were many factors and aspects that are not fully explored within the scope of this paper. For example, there is evidence suggesting that ashes from prairie pellets are rich in cations (Ca^{2+}, Mg^{2+}, K^{+}), and these could be used as fertilizer for prairie grasses (Jariel et al. 2010). A chemical analysis of the ash produced from the combustion of prairie pellets could provide insights on the feasibility of recycling pellet byproducts. Analysis of the gases exhausted during the combustion of pellets could also allow an assessment of the chemicals released into the atmosphere. Evaluation of the cost to produce pellets from prairie biomass and the amount of heat released in a burner or stove could allow a more thorough investigation into the economics and feasibility of a prairie-pellet industry in our bioregion. Research to determine which mixture of native prairie species produces the maximum biomass would benefit other farmers interested in employing prairie pellets as a renewable source of energy.

A study measuring the ground coverage of the native plants and the percentage of nonnative plants would provide a more accurate picture of the productivity of the prairie plots.

Besides the initial costs of seed and planting, there appear to be minimal costs to growing prairie for the production of pellets. Prairie pellets produce a comparable amount of biomass to corn stover, which further encourages the conversion of agricultural fields into prairie. With the growing recognition of the tangible financial benefits and the intangible benefits that healthy ecosystems provide to our society (Daily et al. 1997), the need for responsible management of the prairie ecosystem remains critical. This work exhibits the necessary combination of sustainable agricultural practices and effective financial management, and shows a concrete example of the innovative and regional approaches that are necessary to lead American agriculture to a new, sustainable era.

It is prudent to encourage the cultivation of native, perennial polycultures over other, nonnative biofuel crops. The future condition of our agricultural system will be dependent on individual farmers who are willing and able to recognize the importance of developing agricultural systems that can produce usable energy while also preserving the integrity of the ecosystems that preceded them. The burden of the transition to a more sustainable system is dependent on our body of knowledge. It also hinges on our ability to put into practice projects that can be feasibly employed in current agriculture practices, bypassing lengthy legislative processes while adding significant benefits to the environment. To this end, we support this vision of prairie farming to be embraced by farmers in Minnesota and in other states of the U.S. Midwest.

ACKNOWLEDGMENTS

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CELLULOSE AND LIGNIN CONCENTRATIONS IN PRAIRIE AND WETLAND SPECIES: IMPLICATIONS FOR FEEDSTOCK IN CELLULOSIC ETHANOL PRODUCTION

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Abstract: Highly productive landscapes, such as tallgrass prairies and constructed wetlands, have recently been identified as potential sources of plant material for cellulosic ethanol production. Holocellulose (cellulose + hemicellulose) is a complex structural polymer found in primary and secondary cell walls of vascular plants, and concentrations can vary between species. Theoretical ethanol yields depend on the holocellulose concentration within each plant, but can also be confounded by high concentrations of lignin, which can inhibit microbial enzyme accessibility to cell-wall polysaccharides. The goal of this study was to examine the cellulose, hemicelluloses, and lignin concentrations of species commonly found in prairies and wetlands in southern Minnesota. We examined big bluestem (*Andropogon gerardii*), indian grass (*Sorghastrum mutans*), little bluestem (*Schizachyrium scoparium*), reed canarygrass (*Phalaris arundinacea*), and switchgrass (*Panicum virgatum*) from the Kasota Prairie (Kasota, MN) and cattails (*Typha angustifolia*), reed canarygrass, and bulrush (*Scirpus sp.*) in Rasmussen Woods Nature Area wetlands (Mankato, MN). Biomass from prairie plants had 10%, 7%, and 17.5% more cellulose, hemicelluloses, and lignin, respectively, than biomass from wetland species. Total theoretical holocellulosic ethanol yields (expressed per 1000 kg dry weight) were higher in prairie species averaging 150 liters, compared to wetland species that averaged 136 liters. Theoretical cellulosic and hemicellulosic ethanol yields ranged from 89 to 117 and 35 to 51 liters, respectively, and were generally higher in prairie species. Holocellulose to lignin ratios ranged from 5.9 to 12.2. Our results show there are subtle differences in lignocellulosic biomass between plants from prairies and wetlands that may influence feedstock selection for cellulosic ethanol production.

Key Words / Search Terms: biofuel, energy, hemicellulose, holocellulose, Kasota Prairie, Minnesota, Rasmussen Woods, tallgrass prairie

INTRODUCTION

Communities with high net primary productivity have recently been proposed as sources of plant material for cellulosic ethanol production. In particular, prairies (McLaughlin et al. 2002, Palmer 2006, Tilman et al. 2006) and constructed wetlands designed for phytoremediation (Suda et al. 2009, Zhang et al. 2010) have gained considerable attention due to their species composition consisting of members that are relatively high in structural polymer concentrations. These feedstocks for cellulosic ethanol are high in lignocellulose (lignin + holocellulose) content, and concentrations of these compounds may vary between species.

Holocellulose and lignin are major components of the secondary cell wall of vascular plants. Holocellulose (cellulose + hemicellulose) is comprised of long-chain glucose monomers that are somewhat recalcitrant to degradation. It has long been recognized that these sugars can be fermented to ethanol, although the process of isolation and conversion is quite labor-intensive and costly (Ding and Himmel 2008). In addition, the presence of lignin confounds the conversion of the cellulosic polysaccharides to glucose monomers because holocellulose is imbedded in this highly recalcitrant lignin matrix. Lignin is comprised of phenolic alcohols that polymerize from three monolignols to provide rigidity to the secondary cell wall (Davin et al. 2008). Because of the recalcitrant nature of the secondary cell wall, a chemical pretreatment is necessary to release structural carbohydrates from lignin before conversion to simple sugars can occur. Reducing the cost and increasing the efficiency of these pretreatments are currently the limiting factors for cellulosic ethanol production (Himmel and Picataggio 2008). Once cellulose and hemicellulose are released from the lignocellulosic matrix, these polysaccharides can be broken down into simple sugars using different methods, such as gasification, microbial fermentation, or enzymatic and chemical hydrolysis.

Interestingly, very little attention has been paid as to what types of plants are suitable feedstocks for cellulosic ethanol production outside of traditional sources, such as corn stover, switchgrass, and poplar (Somerville et al. 2010). The purpose of this study was to examine and compare concentrations of cellulose, hemicelluloses, and lignin of plant species found in naturally growing wetlands and in prairies of southern Minnesota. We hypothesized that there would be subtle differences in concentrations of these compounds that could potentially play a role in the selection of feedstocks for the production of cellulosic ethanol. In addition, understanding differences in cellulose and lignin concentrations in naturally growing wetland species may help aid in species selection for construction of wetlands for the sole purpose of phytoremediation.

MATERIALS AND METHODS

SITE DESCRIPTION

Prairie plants were collected from the Kasota Prairie (44°16' N x 93° 59' W) located 2 miles from Kasota, MN, in September 2008. The Kasota Prairie is a 90-acre Minnesota Department of Natural Resources Scientific and Natural Area on the eastern side of the Minnesota River. It contains prairie, wet meadow, oak hardwood, and lowland hardwood communities and is maintained by periodic burning. The prairie portion is mostly a mixture of grasses and dominated by big bluestem (*Andropogon gerardii*), indian grass (*Sorghastrum mutans*), little bluestem (*Schizachyrium scoparium*), switchgrass (*Panicum virgatum*), and the invasive reed canarygrass (*Phalaris arundinacea*).

Wetland plants were collected from the wetlands at Rasmussen Woods Nature Area (44°15' N x 94° 01' W) located in Mankato, MN, in September 2009. Rasmussen Woods Nature Area is a 150-acre city park and contains deciduous forest, prairie, savanna, and cattail marsh. The wetlands are formed from the seasonal Indian Creek stream that runs lengthwise through the park, separating a low-lying cattail marsh and a wooded area dominated by maple (*Acer* spp.) and basswood (*Tilia americana* L.). The presence of a deep (>3.8 m) peat layer in Indian Slough indicates that this area has supported wetland vegetation for thousands of years (EI/USI 1974). The Rasmussen Woods wetlands are dominated by willows (*Salix* sp.), cottonwood (*Populus deltoides*), cattails (*Typha angustifolia*, *T. latifolia*, and the hybrid T. x *glauca*), bulrush (*Scirpus sp.*) and invasive reed canarygrass (Mahoney and Ellstrom 2005). Cattails, reed canarygrass and bulrush are often found in monocultures in this wetland.

CONSTITUENT ANALYSIS

We randomly collected standing aboveground biomass from eight 0.25 m^2-quadrats in both the Kasota Prairie and the Rasmussen Woods wetland areas. Plants in each quadrat were counted and identified to species. Standing aboveground biomass was then cut with shears, placed into paper bags, and returned to Minnesota State University. All samples were oven dried (60°C >48 h) prior to constituent analyses. Dried samples were then hand-sorted by species. Individual plants were processed with a Wiley mill (1-mm mesh size), separated into two 0.5-g subsamples, and stored in 50-ml acrylic vials until analyzed for constituent analyses.

A fiber analyzer (model A200, Ankom Technology, Macedon, NY) was used to estimate concentrations of cellulose, hemicelluloses, and lignin of dried samples. Briefly, 0.5-g samples were placed into pre-weighed filter bags and placed into the fiber analyzer with acid detergent fiber (ADF) solution (20 g cetyl trimethylammonium bromide to 1 L 1.00N H_2SO_4) at 100°C for 60 min. Samples were rinsed with hot dH2O and acetone, dried, and placed in a drying oven (102°C) overnight. Samples were then cooled, weighed, and %ADF (cellulose + lignin) was calculated.

The second subsample for each species was used to estimate neutral detergent fiber (NDF) by placing dried plant material into pre-weighed filter bags and placed into the analyzer with NDF solution (sodium lauryl sulfate, ethylenediamine-tetraacetic disodium salt dihydrate, sodium tetraborate decahydrate, sodium phosphate dibasic, anhydrous and triethylene glycol; pH 7.0). Heat-stable bacterial alpha amylase and sodium sulfite were added to the analyzer and samples were heated to 100°C for 75 min. Samples were rinsed twice with alpha amylase solution, then rinsed in acetone and dried overnight (102°C). Samples were cooled, weighed, and %NDF (cellulose, hemicellulose + lignin) was calculated.

Acid detergent lignin (ADL) was estimated on samples used for ADF analysis. Samples were immersed in 72% H2SO4 for 3 h and agitated every 30 min. Samples were rinsed in dH2O and acetone, dried overnight (102°C), and weighed. Samples were ashed in a muffle furnace (525°C) for 3 h, cooled, and weighed. Acid detergent lignin (lignin) was then calculated. Cellulose concentrations were calculated as %ADF - %ADL, and hemicellulose concentrations were calculated as %NDF - %ADF.

THEORETICAL ETHANOL YIELDS

Theoretical ethanol yields were determined following Badger (2002), using cellulose and hemicellulose concentrations and fermentation assumptions based on 1000 kg of dried biomass. Ethanol yields from each community were calculated based on average cellulose and hemicellulose species content, species density, and dried aboveground biomass per unit area in the prairie and wetland areas sampled.

STATISTICAL ANALYSIS

The general linear model procedure was used with a one-way ANOVA to examine differences in cellulose, hemicellulose, lignin, and theoretical ethanol yields between species. The least significant difference (LSD) post-hoc test was then used to compare individual species means. A Student's t-test was used to compare prairie versus wetland means. Unless specified, species differences were considered significant at $P < 0.05$ level.

RESULTS AND DISCUSSION

CONSTITUENT ANALYSIS

Holocellulose, cellulose, and lignin concentrations were generally higher for species found growing in prairie than in wetland communities. For example, prairie plants had 9.1%, 10.0%, and 18.4% greater holocellulose, cellulose, and lignin concentrations, respectively, than those found in the wetlands (Figure 1). Hemicellulose concentrations also tended to be greater in prairies than in wetlands (P= 0.06; Figure 1). All prairie species had higher concentrations of holocellulose and cellulose than wetland plants, with the exception of reed canarygrass growing in the prairie (Figure 1, A and B). Switchgrass and big bluestem had the highest hemicellulose concentrations (25.8% and 25.7%, respectively; Figure 1, C), but they also had relatively high concentrations of lignin (9.7% and 9.2%, respectively; Figure 1, D) than other plants examined in this study (cattails being the exception). On a species basis, it appears that switchgrass and big bluestem would be good candidates for cellulosic ethanol production based on their relatively high cellulose content combined with greater hemicellulose concentrations, which has a higher conversion efficiency than that of cellulose (Badger 2002).

The cellulose to hemicellulose ratio was not different between prairie and wetland species (Figure 2, A). However, little bluestem had a higher ratio than other plant species owing to its high cellulose (Figure 1, B) and low hemicellulose (Figure 1, C) concentrations (Figure 2, A). The holocellulose to lignin ratio was higher in wetland than in prairie species (Figure 2, B). Bulrush from the wetland had the largest holocellulose to lignin ratio due to its relatively high hemicellulose content (Figure 1, B) and low lignin (Figure 1, D) concentrations (Figure 2, B). On a species basis, bulrush could potentially be a good candidate for a cellulosic ethanol feedstock.

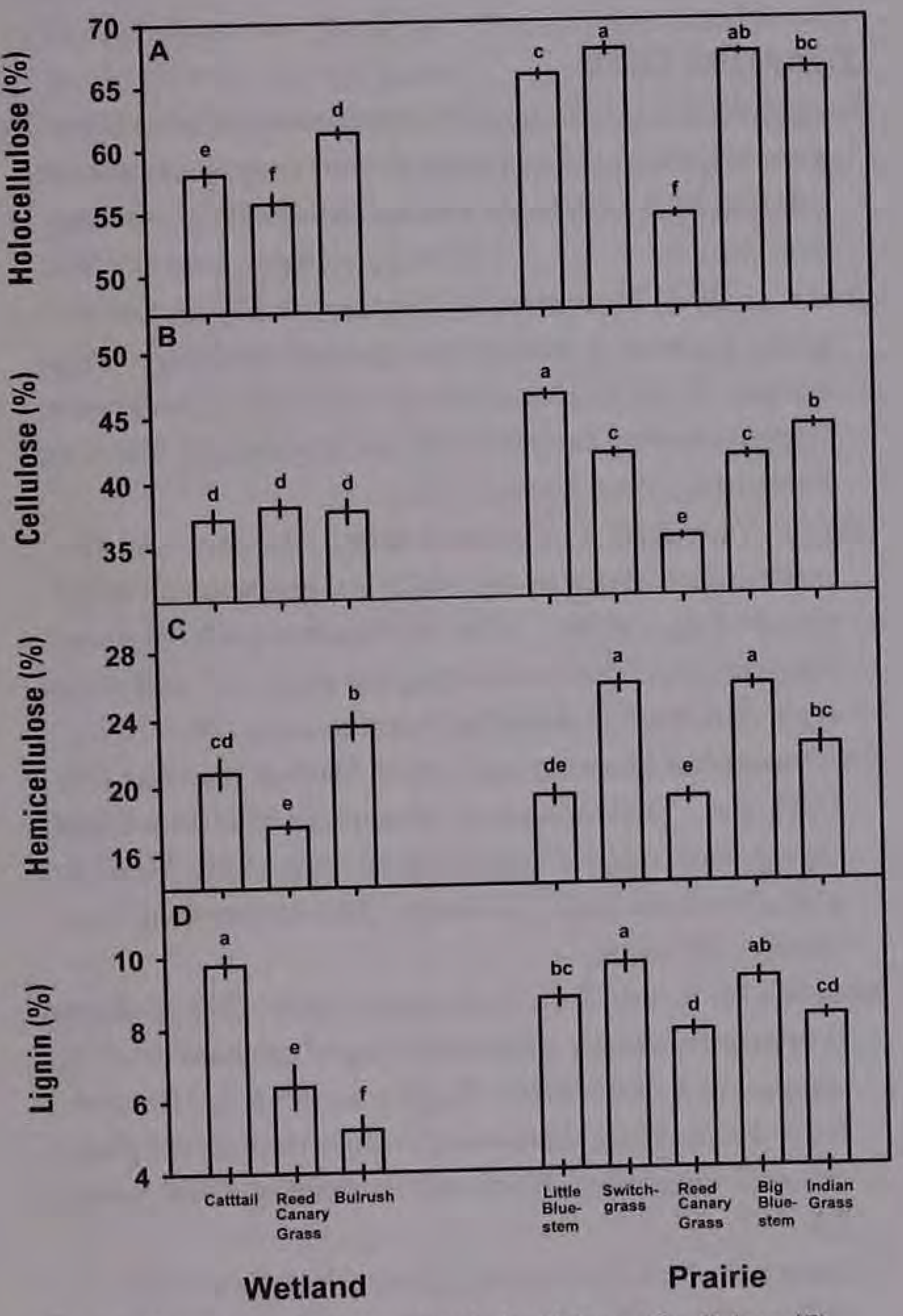

Figure 1. Mean concentrations of (A) holocellulose, (B) cellulose, (C) hemicelluloses, and (D) lignin on a dry-mass basis of wetland plants found in Rasmussen Woods Nature Area and prairie plants found in the Kasota Prairie in southern Minnesota. Values are means of individual species (n=8). Vertical error bars denote ± 1SE. Means values with the same letter designations are not significantly different (LSD comparison, P>0.05).

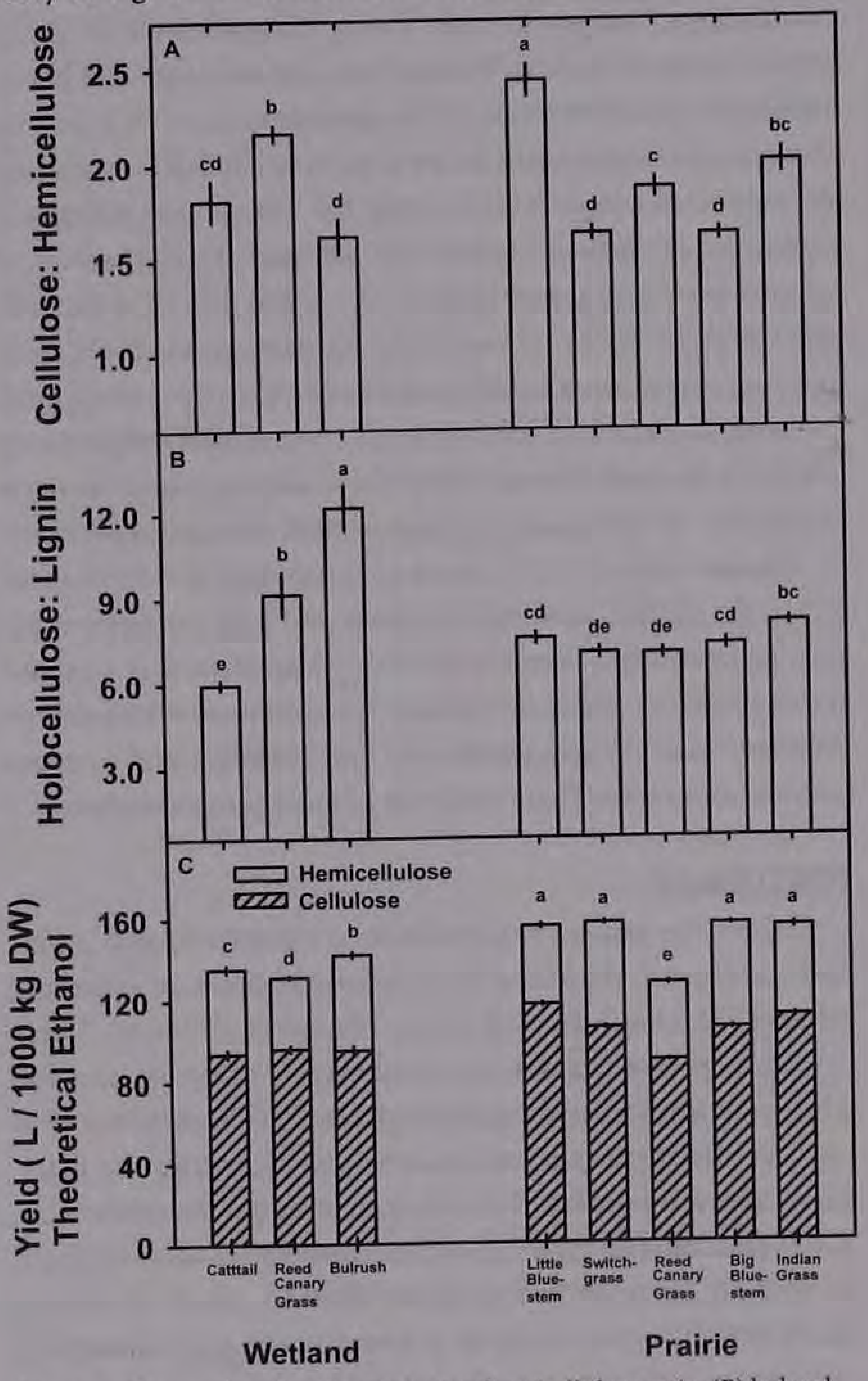

Figure 2. Mean values for (A) cellulose to hemicellulose ratio, (B) holocellulose to lignin ratio, and (C) theoretical ethanol yield for wetland plants found in Rasmussen Woods Nature Area and prairie plants found in the Kasota Prairie in southern Minnesota. Values are means of individual species (n=8). Vertical error bars denote ± 1SE. Means values with the same letter designations are not significantly different (LSD comparison, P>0.05).

THEORETICAL ETHANOL YIELDS

The total calculated theoretical ethanol yield on a species basis was higher in prairie plants than in wetland plants, averaging 150 L and 136 L per 1000 kg dried plant material, respectively (Figure 2, C). With the exception of reed canarygrass growing in the prairie community, total theoretical ethanol yields ranged from 155-157 L per 1000 kg of dried plant material among prairie plants. This was mainly due to prairie species having a higher cellulose concentration than the wetland species (Figure 1, B).

We also measured total aboveground biomass in the prairie and wetland communities we sampled for constituent analyses. In the tallgrass prairie community, total aboveground biomass production (±1SE) averaged 364 (± 46) g m^{-2}. These quadrats were dominated by prairie grasses (>75% cover), but also contained small amounts of herbaceous dicots such as scurfy pea (*Psoralidium tenuiflorum*), ragweed (*Ambrosia* sp.), sage (*Artemisia* sp.), and Aster (*Aster* sp.). These herbaceous dicots were included in total aboveground biomass calculations, but we did not assess holocellulose or lignin content of these plants. We used average cellulose and hemicelluloses concentrations of the prairie grasses to calculate theoretical ethanol yield on an area basis. Based on standing aboveground biomass and using the conversion efficiency equations of Badger (2002), we estimate total theoretical ethanol production to average 54.7 (±0.66) ml m^{-2} in the tallgrass prairie. Total aboveground biomass production averaged 935 (±114), 1408 (±238), and 642 (±175) in the cattail, reed canarygrass, and bulrush wetland communities, respectively (data not shown). Based upon these aboveground biomass numbers, we estimate total theoretical ethanol production to average 126.4 (± 1.70), 184.9 (± 0.94) and 91.6 (±2.54) ml m^{-2} in the cattail, reed canary grass and bulrush communities, respectively (data not shown). It appears that wetland species may be good candidates for cellulosic ethanol production based on species density and aboveground biomass production, rather than solely on holocellulose content.

CONCLUSION

Tall prairie grasses require little to no agricultural inputs and have been identified as potential sources of cellulosic ethanol (McLaughlin et al. 2002, Tilman et al. 2006). These carbon-negative grasses can sequester carbon in the soil while providing abundant aboveground biomass for harvesting. While there is some uncertainty regarding the feasibility of prairie grasses for cellulosic ethanol (Russelle et al. 2007), the demand for ethanol from alternative sources could potentially increase in the future (Service 2010). It is also likely that improvements in pretreatments and conversion efficiencies will increase ethanol yield from natural sources. Wetlands are highly productive communities and produce abundant amounts of aboveground biomass. However, harvesting plants from natural wetlands is likely to cause controversy due to detrimental impacts on the ecosystem and wildlife populations (Suda et al. 2009). Constructed wetlands built solely for phytoremediation might be a viable alternative, and serving dual purposes in wastewater mitigation and as a feedstock for cellulosic ethanol production. These constructed wetlands require few inputs, and annual harvests of aboveground biomass are unlikely to impact their remediation abilities. Our results suggest that wetland species commonly found in southern Minnesota could be an acceptable feedstock for cellulosic ethanol production and could provide as much ethanol, on an area basis, as those found in naturally growing prairies.

ACKNOWLEDGMENTS

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CHARACTERIZING GRASSLAND BIOMASS FOR ENERGY PRODUCTION AND HABITAT IN MINNESOTA

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Abstract: To offset fossil fuel consumption and meet national energy standards, Minnesota is investing in multiple renewable energy technologies, including biomass. Grassland biomass can contribute as a feedstock, but little is known about how repeated harvest of native plant communities will affect future yields and ecosystem functioning. This report describes ongoing research to learn how native grasslands can be managed for renewable energy while simultaneously providing wildlife habitat, water purification, carbon sequestration, and other ecosystem services. Across the western landscape of Minnesota, where significant grassland biomass is available, 60 plots were delineated on previously reconstructed prairie, each plot approximately 20 acres in size. Plots were arranged in a random block design within three general regions spanning the latitudinal gradient of Minnesota (referred to hereafter as northwestern, west-central, and southwestern regions). During the 2009 growing season, surveys documented plant species composition, biomass productivity, estimates of wildlife populations, and soil and vegetation nutrient concentrations in all plots. Following vegetation senescence in 2009, 48 plots were harvested with one of five treatments. Each treatment left a specific shape and size refuge of standing vegetation for wildlife. Production-scale harvest yields were recorded and compared to the productivity measurements gathered during the growing season. The ongoing project will investigate how repeated harvests at different frequencies will affect plant species composition, biomass yields, and other ecological processes, including wildlife habitat suitability. This information will be available for establishing best management practices and initiating a grassland bioenergy industry in the Upper Midwest.

Key Words / Search Terms: prairie bioenergy, biomass quality, grassland management, ecology

INTRODUCTION

As the global demand for energy increases, nations are investing in renewable energy systems to avoid the climatic consequences of fossil fuel use and to stimulate local and regional economies. In the Upper Midwest, many states are producing energy from local water sources, wind, and various sources of biomass. A variety of plant products can be used to produce bioenergy, and, depending on the feedstock and the method of energy conversion, some of these will be more sustainable than others (Tilman et al. 2009, Pineiro et al. 2009). Native grasslands may be able to provide a renewable energy feedstock while providing other ecosystem services such as water purification (Dijkstra et al. 2007), carbon sequestration (Follett et al. 2001, Ajwa et al. 1998), enhanced soil fertility (Dybzinski et al. 2008), and wildlife habitat (Fargione et al. 2009).

Along with other states in the region, Minnesota has state and federal programs to restore and manage grasslands. Depending on the management objective, restorations achieve varying degrees of biodiversity, which in turn influences a number of ecosystem functions. Diversity and composition vary due to initial management goals, differences in methods used to control woody encroachment and litter accumulation, and other factors. With proper management, reconstructed grasslands annually produce aboveground biomass that may be harvested for energy production (Adler et al. 2009). Because prescribed fire is becoming more difficult to implement, practitioners are seeking alternative techniques for grassland management. Mowing has been shown to enhance some grassland ecosystem characteristics (Maron and Jefferies 2001), but little is understood regarding the effects of continued biomass removal. Biomass harvesting at certain times of the year could be detrimental to some wildlife and beneficial to others. Game birds, such as ring-necked pheasants (*Phasianus colchicus*), are ecologically and economically important and could be sensitive to biomass removal at specific times (Murray and Best 2003). Plant communities have changed under mowing regimes in prairies (Williams et al. 2007), which could influence ecosystem properties including soil fertility, hydrology, biodiversity, and stability.

Land managers could have a new economic alternative to prescribed burning if researchers can identify where and when biomass removal can benefit wildlife and ecosystem services. This project is designed to understand how biomass harvesting in grasslands affects habitat quality and to evaluate biomass yields under various harvesting strategies.

We hypothesized that restored grasslands can be harvested sustainably to provide renewable energy, maintain habitat quality, and provide economic stimulus to keep conservation lands intact and encourage further reconstruction of native grasslands (Fargione et al. 2009). Here we report initial results from an ongoing project designed to investigate the effects of biomass harvesting on grassland plants and animals. One objective was to determine dry matter yields of restored grasslands in different regions of Minnesota and use chemical analysis to predict energy production with various conversion technologies. We recorded nutrient concentrations in the biomass to identify any soil nutrient depletion with repeated harvest over multiple years. Long-term monitoring is planned to identify any significant trends in nutrient flux from soil to biomass and to understand how repeated harvests under different frequencies influence future biomass yields and other ecosystem processes.

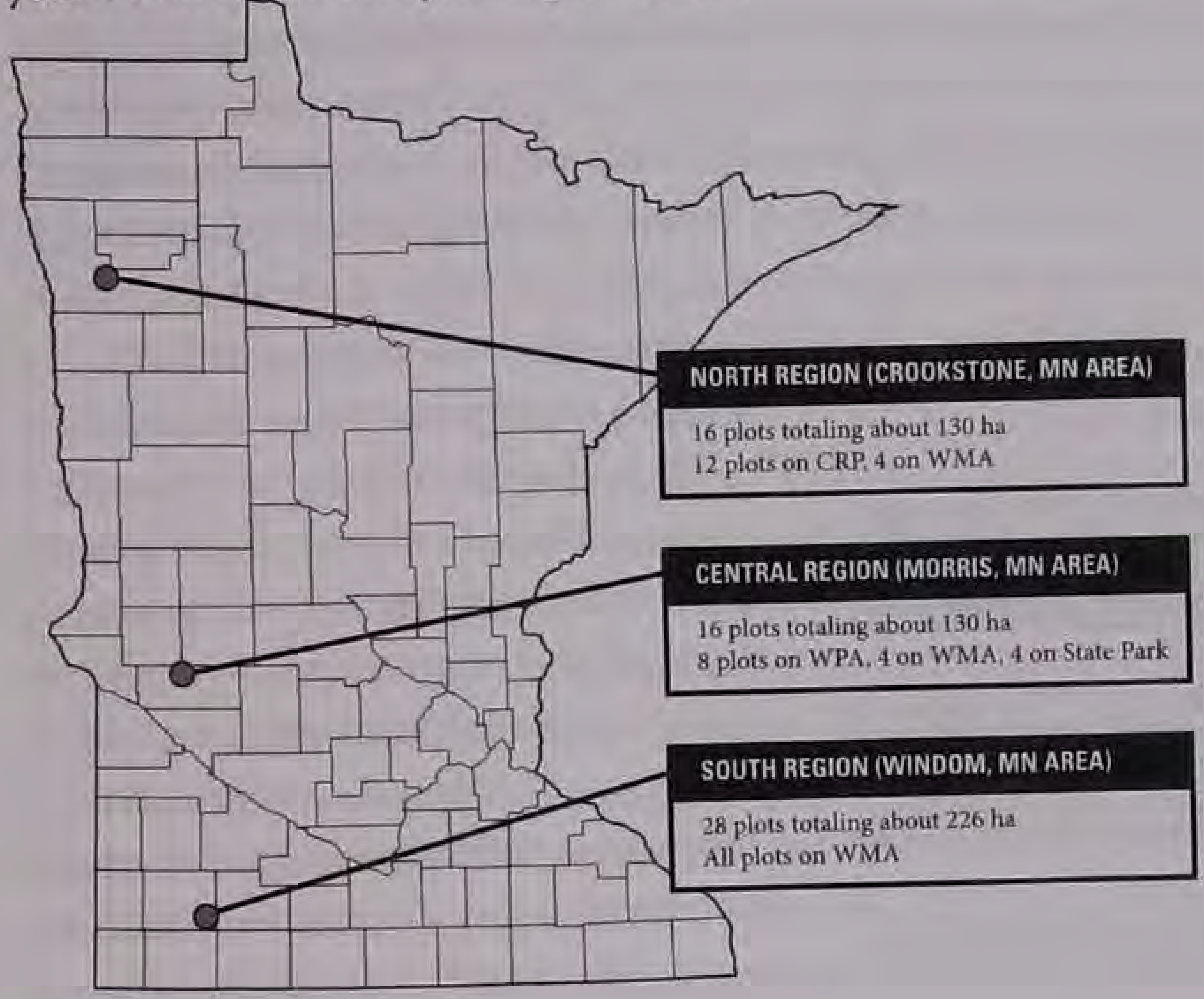

Figure 1. Three general study regions where restored prairie plots are surveyed and harvested in western Minnesota. Each plot is approximately 8.1 ha.

MATERIALS AND METHODS

A repeated block design covering 486 ha (1,200 acres), organized as 60 plots of approximately 8.1 ha (20 acres) each, was set up throughout three regions in western Minnesota (Figure 1). Research plots were chosen and delineated in 2009 on previously restored grasslands in the northwestern, west-central, and southwestern regions of Minnesota. The southwestern region has 28 plots on wildlife management areas (WMA) near Windom, Minnesota. The west-central region has 16 plots on waterfowl production areas (WPA), WMA, and Minnesota State Park land near Morris, Minnesota. The northwestern region has 16 plots on WMA and Conservation Reserve Program (CRP) land east of Crookston, Minnesota. Table 1 describes the general location of each block and the average temperature and rainfall amounts for that region. Initial plant and animal surveys were conducted on all 60 plots prior to the 2009 harvest.

HARVEST

Between late fall and early winter of 2009, 43 of 60 plots were harvested. The harvest began October 12, 2009, in the northwestern region and ended December 14, 2009, in the southwestern region. Vegetation from four plots scheduled for harvest was not cut because excessive rainfall prevented equipment access. Vegetation was cut using a self-propelled windrower with a mounted disc header that cut a swath approximately 4.6 meters wide and left a mean stubble height of 13.9 cm. Prior to baling, the biomass was raked into larger windrows and allowed to dry overnight unless conditions were exceptionally dry. Biomass was collected with a large square baler in the northwestern region. To reduce rutting, that equipment was exchanged for a large round baler to collect the biomass in the west-central and southwestern regions. Bales were counted on each plot after all vegetation was collected. Bales were weighed on transport to the storage facility using a DOT-certified scale.

VEGETATIVE SURVEYS

In each plot, two randomly chosen points were surveyed for plant composition and biomass productivity. At each point, a 0.75 meter x 5 meter grid was used to create a sampling subplot. Within the subplot, estimates of cover were recorded for each species, bare ground, and litter. Cover was determined as a percentage of the whole subplot area, with values for all species summing to 100%. Following the cover estimates, vegetation from the entire area was cut, bagged, and weighed. A subsample of approximately 500 g was removed and weighed in a separate no.57 paper bag and dried for at least 14 days at 38°C before being reweighed to estimate water content. Biomass was cut using a battery-powered hand shears to an approximate height of 2.5 cm.

SOIL SAMPLING

Eight soil cores were collected from each plot and analyzed for nitrogen, phosphorus, potassium, zinc, iron, magnesium, manganese, copper, calcium, pH, organic matter, and cation exchange capacity. Soil samples were taken at the northwest corner of each survey subplot and six additional random points generated with ArcGIS. Soil samples were taken with an Oakfield Apparatus LS soil sampler with an S-2 tip to a depth of 20 cm. All of the samples for a plot were combined, placed in a paper bag, and dried for at least 30 days at 38°C.

HARVEST BIOMASS SAMPLING

At each plot, every second bale was cored using a Forageurs Hay Probe attached to a battery-powered drill. In each plot, all cores were mixed and stored in a no.57 paper bag and weighed. The location of each bale core was recorded with an eTrex global positioning system, and the stubble height was recorded 10 meters from the bale at a randomly chosen direction. Core samples were dried for at least 14 days at 38°C and then weighed to calculate dry matter of bales in each plot.

Biomass from core samples was processed to achieve values for nutrient concentrations. Analysis produced values for nitrogen, phosphorus, potassium, sulfur, calcium, magnesium, sodium, zinc, iron, manganese, copper, and boron. Sugar analysis described concentrations for C5 and C6 sugars to produce a theoretical ethanol yield using the equation provided by the U.S. Department of Energy Calculator (available online at http://www1.eere.energy.gov/biomass/ethanol_yield_calculator.html).

Mean biomass yield was calculated by multiplying the number of bales produced on each plot by the mean bale weight (either round or square). Dry matter was calculated by subtracting the water content calculated from the core samples from the total yield value.

DATA ANALYSIS

Data were collected and processed throughout the project. Statistical analyses will compare yields, biomass nutrient values, and soil properties between years and among the harvesting regions. Data analysis will be ongoing. Preliminary data follows here.

Table 1. Average temperature and total rainfall values during 2009 growing season collected from weather stations near experimental plots in three regions of Minnesota. Precipitation and temperature data gathered from National Climatic Data Center/NOAA reports (NOAA 2010).

REGION	STATION	APRIL	MAY	JUNE	JULY	AUGUST	SEPTEMBER
		MEAN TEMPERATURE (°C)/TOTAL RAINFALL (CM)					
SOUTH-WEST	Windom	7.9/6.68	15.8/2.64	19.8/11.15	20.7/7.72	20.8/5.33	18.7/5.21
	Lakefield	6.2/4.67	13.8/3.68	18.6/13.89	18.8/16.1	18.7/5.38	17.1/4.52
	Worthington	5.9/5.84	*	*	*	19.1/11.3	17.1/4.83
WEST-CENTRAL	Morris	5.2/2.24	13.1/1.22	18.1/5.39	18.9/2.59	18.4/9.6	17.3/6.35
	Benson	6.2/2.74	13.7/0.74	18.5/6.76	19.6/3.99	19.4/9.09	18.3/1.55
	Artichoke	5.8/1.47	13.7/2.13	17.9/4.55	19.1/7.09	19.0/11.1	17.7/7.75
NORTH-WEST	Crookston	*	10.6/6.4	16.9/11.81	*	*	*
	Red Lake Falls	4.4/2.67	10.9/8.89	16.3/10.92	18.1/6.43	17.8/14.71	17.6/3.58

* indicates missing values

RESULTS

Mean dry yields in the southwest were greater with a mean of 2.65 Mg/ha compared with 1.6 and 1.35 Mg/ha for the west-central and northwestern locations respectively. Chemical analysis of the sugars revealed differences in the theoretical ethanol yield between the three locations. The theoretical yield calculator generated a value of 458.6 L/Mg for biomass from the southwest, and values of 431.9 and 427.7 L/Mg for biomass from the west-central and northwest, respectively. Mean nitrogen concentration in all the biomass samples was 7.3 g/kg, phosphorus was 0.6 g/kg, and potassium was 2.3 g/kg.

DISCUSSION

Biomass quantity and quality differed from restored grasslands in the northwestern, west-central, and southwestern regions of Minnesota, and the significance of these differences will be determined as more data are available. Present variation in these values may be a function of climatic conditions or management methods and objectives. More data will identify any significant differences in biomass quality and quantity between plots managed by state, federal, or private entities. Regional differences in biomass production are important to organizations establishing energy-processing plants that can use grassland feedstocks. These data will inform market decisions as the bioenergy industry grows, and suggest management directions for producers and land managers. With accurate estimates of the raw energy content of biomass and theoretical yields of liquid fuels, land managers will be able to quantify energy production from local grasslands. With nutrient concentration data from the biomass, energy conversion facilities can predict ash concentrations, providing estimates of indirect operating costs that would accrue from ash disposal.

These data will also help determine any ecological impacts of repeated biomass removal. In particular, the biomass nutrient concentrations will help predict potential nutrient removal from soils with repeated harvest, and soil cores will monitor nutrient and mineral concentrations to detect any depletion as a result of biomass removal.

Quantitative values of bioenergy potential from perennial grasslands will add information concerning agricultural expansion to include multifunctional practices (Jordan et al. 2007). Any resulting economic incentives for conventional farmers to implement conservation practices on agricultural land, such as planting perennial grasslands to intercept contaminated runoff, can benefit both the environment and society (Boody et al. 2005).

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PRAIRIE POLYCULTURES AND LOCAL BIOENERGY

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Abstract: While much of the current research on perennial bioenergy crops focuses on increasing yields of a few grass species grown in monocultures, other scientists and organizations, notably including the Tallgrass Prairie Center, are studying yield potential and other aspects of how prairie polycultures can fit into the emerging bioeconomy. An implicit or explicit hypothesis of such research is that harvesting of prairie vegetation for bioenergy can be compatible with environmental objectives such as soil conservation, water quality protection, and biodiversity. As farmers begin to grow perennial grasses and forbs for bioenergy production, there will be opportunities for on-farm collaborative research. Establishing a local bioenergy system can provide a platform for such research and a model for coproduction of biomass feedstocks and environmental benefits. This paper presents historical and policy context and a landscape vision for perennial energy grasses. The Local Bioenergy Initiative being conducted by the Agricultural Watershed Institute (AWI) in the Decatur, Illinois, area is presented as a case study. Insights from the local food movement that can be applied to local bioenergy are discussed. Perennial energy grasses, including prairie polycultures, are discussed as part of a broad effort to promote perennial crops and multifunctional agriculture.

Key Words/Search Terms: bioenergy, local food, prairie polycultures, ecosystem services

HISTORICAL AND POLICY CONTEXT

Between the 1930s and 1960s, a seismic shift took place in agriculture as farms mechanized and industrially produced nitrogen fertilizer largely replaced manure on midwestern farms. In The Omnivore's Dilemma, Michael Pollan (2006) points to the day in 1947 when the Muscle Shoals munitions plant in Alabama switched to making chemical fertilizer as a key turning point in the history of corn and the industrialization of our food. Crop statistics aggregated by Laura Jackson (2002) show that total acreage of row crops and sod crops (hay and small grains) maintained roughly a fifty-fifty balance on Iowa farms from the 1860s to the 1950s, but then soybeans steadily replaced the sod crops as first horses and then cattle disappeared from many farms. By 1970, small grains and hay accounted for only a tiny fraction of total crop acreage in many Corn Belt watersheds.

After the oil crises of the 1970s, agriculture came to be seen as a source of renewable energy as well as food, feed, and fiber. Agricultural bioenergy in the United States has mostly consisted of ethanol made from corn starch. Critics of federal incentives for corn ethanol point to its modest energy return on investment, the adverse impacts of corn production on soil and water resources, and the impact on food supplies of diverting crops or cropland to energy production. Use of cellulosic biomass, including perennial grasses, for bioenergy offers the possibility of improved energy and environmental outcomes. However, this concept also has critics, who doubt the system's economic viability or predict adverse environmental impacts depending on which methods are used for feedstock production and energy conversion.

In recent years, mandates and incentives for next-generation biofuels have been written into federal energy and farm legislation. The Energy Independence and Security Act of 2007 (Public Law 110-140, also called the 2007 Energy Bill) mandates production of 36 billion gallons of renewable transportation fuel per year by 2022, with 15 billion gallons of conventional corn ethanol and 21 billion gallons of advanced biofuels that meet specified targets for greenhouse gas reduction. Of the advanced biofuel goal, 16 billion gallons must come from cellulosic feedstocks. A new Biomass Crop Assistance Program (BCAP) was created by the Food, Conservation, and Energy Act of 2008 (Public Law 110-234, also called the 2008 Farm Bill). Final BCAP rules were published in the Federal Register on October 27, 2010, and codified as 7 CFR Part 1450. BCAP will provide financial assistance to land owners and farm operators in selected project areas for establishment and production of eligible bioenergy crops, including grasses and forbs. To qualify for BCAP payments, biomass must be converted to heat, power, biobased products, or advanced biofuels, in a biomass conversion facility certified by USDA.

With these federal policies in place, it appears that perennial energy grasses are ready to take their place alongside corn and soybeans as a cash crop on midwestern farms, but this may turn out to be a slow process. Accessed on October 31, 2010, the Wikipedia entry on cellulosic ethanol commercialization listed 12 cellulosic ethanol plants in the U.S. as operational or under construction. However, none of the plants on the list showed perennial grasses as the expected feedstock. Most will use either woody biomass or crop residues.

In the near term, grass biomass appears more likely to be used for conversion to heat and/or power, rather than for transportation fuels. This will qualify for BCAP incentives but will not count toward the energy bill's biofuel target. Enzymatic processes to convert cellulose into biofuels are expected to require a uniform feedstock. In contrast, heat and power technologies involving direct combustion or gasification of biomass are generally more tolerant of diverse feedstocks, including prairie polycultures.

There is a window of opportunity to develop relatively small, local markets for perennial grasses before large-scale cellulosic biofuel production ramps up. Grass-farming pioneers, entrepreneurs, and scientists—with the support of energy consumers and funders willing to use their purchasing power to support sustainable biomass energy—can use this time to shape a local bioenergy system that includes native grass/forb polycultures managed for both renewable energy and a variety of environmental benefits. A thriving local bioenergy system would represent an alternative to large-scale commodity-style production of dedicated energy crops.

A LANDSCAPE VISION FOR PERENNIAL ENERGY CROPS

How would a local bioenergy system differ from large-scale production of biomass feedstock for advanced biofuels? There would probably be many differences in the type of feedstock produced, where, how, and by whom. For herbaceous energy crops in particular, there will be—or at least can be—a significant niche for prairie polycultures in the local bioenergy system. By contrast, it appears likely that acreage planted specifically to provide feedstock for production of cellulosic biofuels will consist almost entirely of monocultures of switchgrass (*Panicum virgatum*), miscanthus (*Miscanthus x giganteus*), or a few other highly productive grass and tree species.

To meet the biofuel targets in the 2007 Energy Bill, USDA estimates that 13.4 billion gallons of cellulosic biofuels will be produced annually from dedicated energy crops such as perennial grasses, energy cane, and biomass sorghum by the year 2022 (USDA 2010). The USDA roadmap report indicates that this would require about 27 million acres of cropland, representing about 6.5% of the U.S. total of 406 million acres of cropland. In a 16-state region stretching from the Dakotas across the Midwest to Virginia, USDA's regional roadmap projects that 10.8 million acres, representing 4.5% of total cropland and pasture acreage in those states, will be used for dedicated energy crops. If biofuels development actually happens at that scale and pace, it will represent a dramatic land-use shift in a relatively short timespan.

A positive vision of sustainable biomass production in the Corn Belt is presented in a recent National Wildlife Federation biomass energy primer (Kemp and Sibbing 2010):

> Imagine the diverse and productive midwestern farm country of tomorrow, transformed by the integration of resource-protecting biomass crops grown for energy alongside commodity and food crop production. . . . The highly erodible fields with steep slopes or depleted and windblown soils are now planted to perennial grasses or mixed prairie species to hold the soil. Every stream, lake, drainage ditch and wetland is now buffered by a wide swath of perennial biomass vegetation. Roadsides are seeded with appropriate biomass varieties. Dry fields that used to be irrigated with precious ancient groundwater supplies are now planted to drought tolerant grasses and prairie mixtures. Pastures and grasslands have expanded as farmers learned to manage them for both livestock grazing as well as biomass production, depending on markets and their own chosen options. . . . Low yielding croplands that were designated disaster areas as often as not because of all too predictable floods, late snows, or droughts, are now planted to hardy and tolerant perennial grasses and fast growing trees which yield reliably every year.

From both an economic and environmental perspective, this type of well-designed mosaic of annual and perennial crops, including energy crops grown in both monocultures and polycultures, appears to be a sound approach for sustainable food and energy production. One of the main rationales for cellulosic bioenergy is to substitute renewable biomass for fossil fuels, and thereby reduce greenhouse gas emissions. While energy and carbon analyses are complex and may be controversial, higher yields generally mean that more fossil fuel can be replaced per acre devoted to a biomass crop. Multifunctional agriculture involves tradeoffs between multiple objectives. Beginning now to grow and harvest native perennial energy grasses on a relatively small scale can offer valuable opportunities to demonstrate and refine cropping systems for coproduction of biomass feedstocks and ecosystem services.

LOCAL BIOENERGY IN THE LAKE DECATUR WATERSHED

The Lake Decatur watershed in Central Illinois experienced the decline of pasture and sod crop acreage and a shift to a corn-soybean crop rotation starting in the 1920s, as shown in Figure 1. This transition began early in the Decatur area. In 1922, the Decatur-based A. E. Staley Manufacturing Company opened the first Illinois soybean processing plant, providing a local market for the new crop. The Lake Decatur dam was built in 1921-22 to impound the Sangamon River and provide water for Decatur residents and industry, including the Staley grain mills. Lake Decatur is still the city's water supply and an important recreational amenity with boating and lakeshore parks.

Since about 1970, the nearly level to gently sloping prairie soils that occupy most of the Lake Decatur watershed have been used almost exclusively to grow corn and soybeans. This land cover pattern persists today, as shown in Figure 2. Woodlands are located mainly along the Sangamon River valley and its tributaries. Scattered tracts of native grasslands are found mainly in public nature preserves or private lands managed for wildlife and conservation. Macon County, which has over 300,000 acres of cropland, had only about 3,500 acres enrolled in the Conservation Reserve Program as of 2007.

Erosion and deposition of sediment in Lake Decatur have been problems since the lake was first built. The City is in

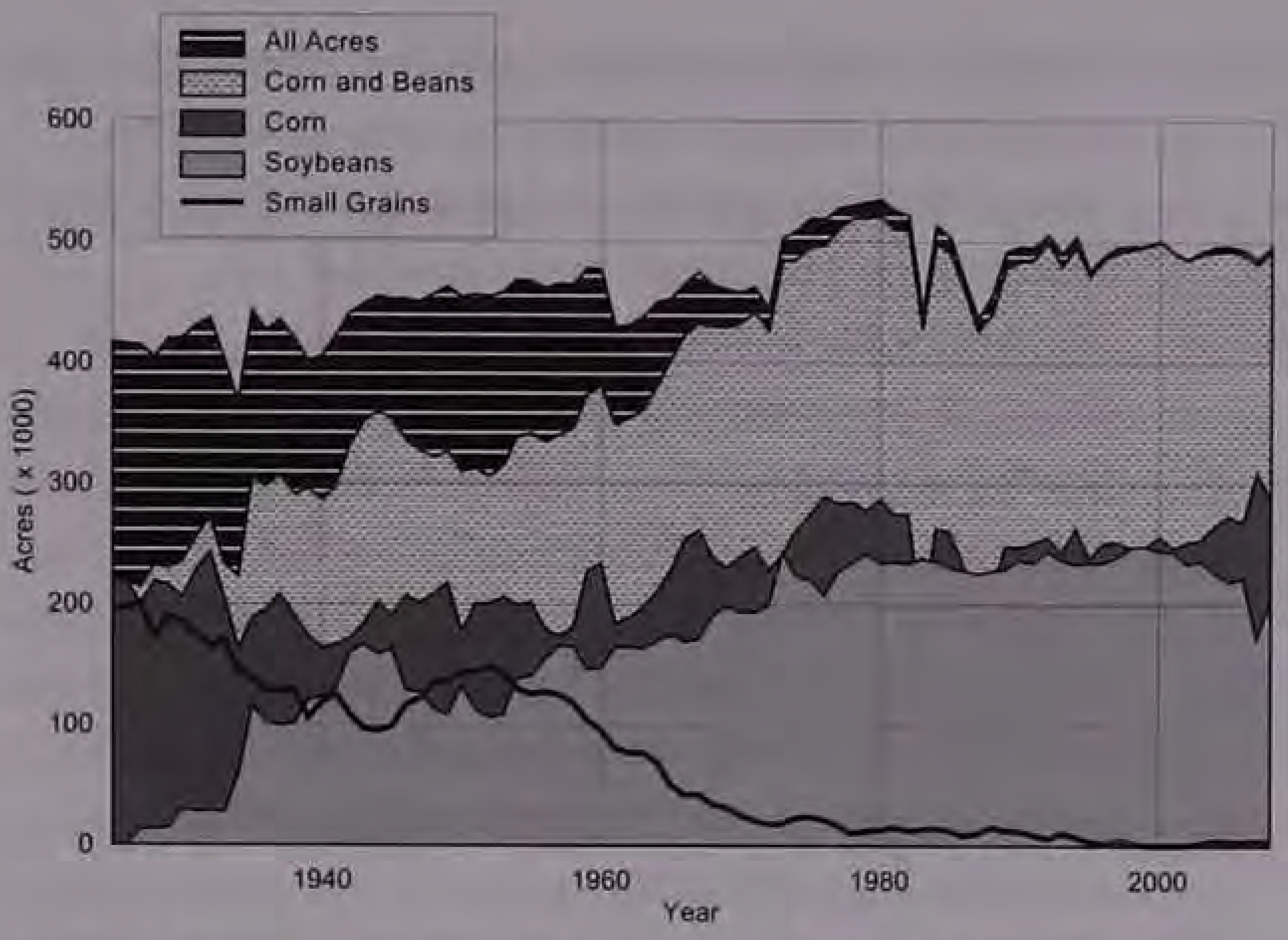

Figure 1. Crop acreage harvested in the Lake Decatur watershed, 1925-2008 (from Keefer et al. 2010).

the middle of a phased dredging program to restore reservoir capacity. Much of the cropland in the watershed is tile drained. Elevated nitrate concentrations in the lake have been a recurring problem since the early 1980s. Decatur constructed an ion exchange treatment facility to ensure compliance with the drinking water standard for nitrates.

Like most of Central Illinois, the Decatur area has high land prices and cash rents for prime cropland. Local farmers and agribusinesses expect corn stover to be the primary feedstock for cellulosic bioenergy, at least in the near term.

Figure 2. Lake Decatur watershed map (from Keefer et al., 2010).

At first glance, this makes the Lake Decatur watershed an unlikely place to attempt to demonstrate production and use of perennial grasses for bioenergy. However, the Agricultural Watershed Institute (AWI) and the City of Decatur recognized the potential for grasses planted on slopes and along streams and ditches to reduce sediment and nutrient loads in the lake. To get more grasses planted, this must make economic sense for landowners and farmers, so AWI began by looking at the economics of growing perennial grasses for bioenergy and conservation purposes.

In 2007, AWI conducted a market development study for energy grasses in the Decatur area (John and Watson 2007). The study looked at co-firing biomass with coal in industrial cogeneration boilers and pelleting biomass for home or commercial heating. The economic analysis showed that current prices of fossil fuels, especially coal, on a million-BTU basis would be less than the price needed for local farmers to profitably grow energy grasses. The research team estimated the price gap between the production cost and the coal-equivalent energy value of biomass under various scenarios. Prospects for carbon credits, renewable energy credits, USDA conservation programs, or other types of ecosystem service payments were addressed. Constructing a 120-ton-per-day grass-pellet manufacturing facility in the Decatur area was found to be not economically feasible in the short term. The report presented a local biomass energy roadmap that included, among other action steps, demonstrating the use of prairie biomass in industrial boilers and home heating furnaces, research on environmental benefits of energy grasses, and further consideration of the feasibility of making pellets on a smaller scale.

The idea for the Local Bioenergy Initiative soon emerged as a way to start creating a local market for grass biomass and to learn about how and where to grow energy grasses to optimize coproduction of harvestable biomass and environmental benefits. AWI and the City of Decatur envision the Lake Decatur watershed as a showcase for producing perennial energy grasses in locations designed to reduce erosion and to protect water quality in a landscape dominated by annual row crops. Components of the initiative include

1. technical and financial assistance to early adopter farmers and landowners,
2. demonstration of landscape design concepts for cropping systems that include perennial energy grasses, and
3. development of local markets for grass biomass and related ecosystem services.

In addition to City of Decatur financial support, grants for the Local Bioenergy Initiative have been received from the Lumpkin Family Foundation and the Walton Family Foundation. Creating a grass-based local bioenergy system involves figuring out how to deal with a chicken-and-egg problem: Who will plant a crop for which there is no existing market? Who will invest in equipment to process or use an energy feedstock that is not yet being produced? Funders are helping to overcome this challenge by enabling AWI to buy equipment and offer incentives to people who want to plant energy grasses or install grass-burning furnaces. AWI bought a mobile, diesel-powered pellet mill. It will be used for demonstration projects and made available to farmers or businesses that want to experiment with making their own pellets. With funds from a State of Illinois biomass-to-energy grant, AWI is buying a trailer-mounted, PTO-driven biomass briquetter that will be used mainly to make grass or stover briquettes for test burns in industrial boilers.

Here are some of the early-adopter farmers currently participating in the Local Bioenergy Initiative:

1. Two organic farmers planted a mixture of warm-season prairie grasses as field borders to separate their organic fields from the adjacent conventionally-farmed fields. With cost-share funding from the initiative, both are planning to install grass-burning furnaces to heat equipment sheds on their farms.
2. A part-time farmer who produces both row crops and hay has installed a pellet boiler to heat his house. (See Figure 3.) He expects to grow prairie grasses in contour buffer strips and make his own grass pellets. AWI is especially interested in the water quality benefits of contour grass strips. Initiative participants who grow, process, and burn grass biomass on a single site will host field days to educate the public about biomass energy.
3. A farmland owner has signed an agreement under which AWI will establish a native grass/forb mix on approximately 13 acres of her property. The field will be managed for both biomass production and game birds. Initiative funds will be used to pay the owner an annual conservation incentive. The land owner will receive additional income by leasing hunting rights to a group of quail hunters.

Figure 3. Installation of a biomass boiler cost-shared by the Local Bioenergy Initiative.

To create a grass-based local bioenergy system will require collaboration with equipment manufacturers, including small businesses looking for a niche in the renewable energy market. Today, wood pellets and furnaces designed to burn wood pellets are widely available. However, grass pellets have different properties, such as a higher ash content, and require furnaces designed to handle this fuel. Equipment to make pellets or briquettes on a large scale is also commercially available, but some modifications may be needed to handle dry grass biomass. Here are some of the entrepreneurs that AWI is working with:

1. A Central Illinois farmer-entrepreneur owns a company that manufactures corn-burning furnaces and boilers. He has successfully modified these heating appliances to burn grass pellets. His business plans include growing energy grasses, making pellets, and manufacturing furnaces to burn the pellets. AWI is arranging to cost-share on installation of several of his grass-burning units.
2. Another Illinois entrepreneur is in the process of commercializing a gasifier furnace system with the flexibility to use various forms of grass, woody, or waste-stream biomass. He is planning to build a prototype biomass conversion facility that can make densified biomass fuel from multiple feedstocks to meet a customer's specifications.
3. A small manufacturing company located in Pennsylvania makes the biomass briquetter that AWI purchased for use in our demonstration projects. This equipment is marketed mainly to farmers or small businesses that want to make briquettes for their own use or for sale.

In addition to these farmers and entrepreneurs, AWI is working with one major corporation to grow energy grasses on company-owned land and with two other companies to conduct test burns of biomass briquettes in industrial boilers. The sustainability plan recently adopted by the Decatur City Council sets a goal of establishing 10,000 acres of perennial energy grasses on marginal land in the Lake Decatur watershed by the year 2020 to provide bioenergy feedstock and also to reduce sediment and nutrient loads in the lake. That would represent about 3% of the crop acreage in the lower portion of the watershed. If the test burns lead to co-firing biomass with coal on an industrial scale, the 10,000-acre goal appears to be realistically achievable.

LOCAL FOOD AND LOCAL BIOENERGY

What are the similarities and differences between local bioenergy and local food? As local bioenergy systems begin to take shape, what useful insights can be drawn from the well-established local food movement? How do these insights apply to the special case of prairie polycultures grown for bioenergy? Here are a few tentative thoughts on these questions:

A PRAIRIE PREMIUM?

Agricultural commodities are, by definition, interchangeable with another unit of the same commodity. At a given time and place, a buyer will pay the same price per bushel for any farmer's no. 2 yellow corn. By contrast, fruits or vegetables at farmers' markets are "differentiated" products that may sell for different prices, based on freshness, visual appeal, and other attributes. Many consumers willingly pay a substantial premium for organic food, presumably because they value its health benefits or other attributes. If it costs more to grow a ton of mixed prairie grasses and forbs than a ton of switchgrass, will homeowners or businesses be willing to pay a premium for the prairie pellets? Perhaps so, if the buyer places a value on supporting the production of diverse biomass. An alternate possibility is that the additional cost of growing a polyculture could be paid by a hunting club (as in the Decatur-area example) or an environmental organization such as the Audubon Society. Third-parties' payments for environmental benefits related to production of perennial crops are sometimes called "stackable" ecosystem service payments (World Resources Institute 2009). This could become an important mechanism for supporting the sustainable production of high-diversity biomass.

ENERGY GRASS FARMING

Farmers with the necessary knowledge, skills, and equipment will be essential to the success of any grass-based bioenergy system. Where will they come from? What will the business of grass farming look like? Expanding markets for local food attract beginning farmers and current farmers, and this is likely to happen for local bioenergy. For beginning farmers, growing fruits or vegetables generally requires less capital investment for equipment than growing corn and soybeans. Likewise, grass-planting and -harvesting equipment is less expensive than the big planters and combines commonly used for row crops. It would be relatively easy for custom hay producers to begin growing energy grasses since they already own most, if not all, of the equipment needed. Land costs are another issue. The cost of buying or renting farmland may be more of a hurdle for grass farmers than for local food producers, since the crop value per acre will presumably be lower for biomass crops than for fruits or vegetables. Conservation incentives can help to overcome this obstacle. AWI is exploring contractual arrangements and cropping systems for one farmer to grow row crops on prime upland soils while another farmer grows energy grasses on sloping, wet, or flood-prone portions of the same field. Local biomass markets coupled with federal BCAP incentives or ecosystem service payments could make this approach economically attractive to land owners, corn growers, and grass farmers. Such arrangements could help to make the National Wildlife Federation's sustainable biomass vision a reality, even in prime row-crop areas of the Corn Belt.

COMMUNITY-SUPPORTED ENERGY

The community-supported agriculture (CSA) business model is commonly used by local food producers. In a typical CSA, members pay a fixed price to a farmer who provides weekly deliveries of locally grown vegetables. The parallel concept of community-supported energy (CSE) has been suggested as a model for bioenergy production on a small scale. A grass-based CSE farmer would grow perennial grasses, make pellets or briquettes, deliver the fuel in bag or bulk form to CSE members, and collect ash to be returned to the soil. A potential food-energy synergy would be for CSA farmers to form CSEs. An organic vegetable grower near Springfield, Illinois, is considering that possibility. With cost-share assistance from AWI, he plans to install a grass-pellet furnace to heat his home with prairie biomass grown on his farm. As the local bioenergy concept catches on, pellet mills and grass-burning furnaces are likely to become more available, dependable, and affordable, which would help to make CSEs a viable business model.

CONNECTEDNESS

For midwesterners, the modifier "local" for food or energy highlights the contrast with vegetables from California, coal from Wyoming, or oil from the Middle East. The low bulk density of chopped or baled grass means that transportation costs would get very high unless it is densified or converted into energy near the fields where it is grown. One could say that grass biomass is inherently local, since long-distance transport is not economical. As applied to food and energy systems, therefore the concept of "local" is about more than just physical distance: It suggests closer connections between producers and consumers and between consumers and the land. Part of the appeal of local (and "slow") food seems to come from knowing who produced it and savoring the experience of food preparation and enjoyment. Pick-your-own farms or orchards and CSAs that host occasional meals on the farm encourage urban residents to reconnect with the source of their food. Local bioenergy also fosters this sense of connectedness. Homeowners who use grass pellets for heat or large companies that co-fire biomass for heat and power can take satisfaction in knowing the farmers who grew their fuel and seeing the fields where their energy crop is reducing soil erosion and providing wildlife habitat.

LOCAL BIOENERGY AND MULTIFUNCTIONAL AGRICULTURE

The Green Lands, Blue Waters Consortium (GLBW) includes land-grant universities and nonprofit organizations located mainly in the Upper Midwest. The mission of GLBW is to support development of, and transition to, a new generation of agricultural systems in the Mississippi River basin that integrate more perennial plants and cover crops into the agricultural landscape.

AWI's Local Bioenergy Initiative uses the theory of change developed by GLBW. It emphasizes use of stakeholder learning groups to exchange information and develop markets for the products of multifunctional agro-ecosystems. A broad goal of the Local Bioenergy Initiative is to help ensure that the transition to cellulosic bioenergy envisioned in federal energy policy is done in a way that enhances water quality, ecosystem health, and local economies. Other GLBW members are involved in projects with the same goal. For example, Rural Advantage is working to implement a local bioenergy system in the area around Madelia, Minnesota. GLBW members are also helping to guide the Koda Energy Fuelshed Project in Minnesota. An aim of that project is to use perennial energy crops grown on 10,000 acres as feedstock for a biomass-based cogeneration system built by Koda Energy LLC (Jordan and Warner 2010).

Learning groups are a good way to synthesize input from stakeholders from different backgrounds and generate innovative ideas for cropping systems and enterprises that are profitable for the farmers and also meet environmental objectives. The GLBW approach can be applied on a watershed scale to develop bioenergy systems well suited to local conditions. As the United States implements policies to increase production of renewable energy from agriculture, the GLBW model could be broadly applied to move toward multifunctional agriculture at the national level (Jordan and Warner 2010).

Bioenergy from perennial grasses is still in its infancy in the United Stated and Canada, but many projects tailored to local conditions and markets are in various stages of implementation. For example, the Show Me Energy Cooperative in western Missouri owns and operates a large facility to make biomass pellets from multiple feedstocks, including native warm-season grasses. The New Jersey Audubon Society installed furnaces to use grass pellets to heat two nature centers, and a New Jersey company, Plainview Growers, is implementing a plan to contract with local farmers for thousands of tons of grass and to make pellets to heat Plainview's greenhouses (Parke 2010). Woody biomass is already used extensively to heat homes or schools and generate power.

The cast of characters needed to grow and use perennial grasses for renewable energy is large: farmers, land owners, businesses in the field-to-flame supply chain, energy end users, seed companies, equipment manufacturers, and many more. A grass-based bioenergy system specifically intended to maximize environmental benefits will also include scientists from many disciplines, conservation professionals and educators, agricultural and environmental program managers, and private or philanthropic purchasers of ecosystem services. For prairie polycultures in particular, many important topics, such as compatibility of harvesting practices with prairie management, need further study. Active engagement of ecologists and biodiversity advocates in stakeholder learning groups and landscape-scale research can help to ensure a significant and positive role for prairie polycultures in local bioenergy systems.

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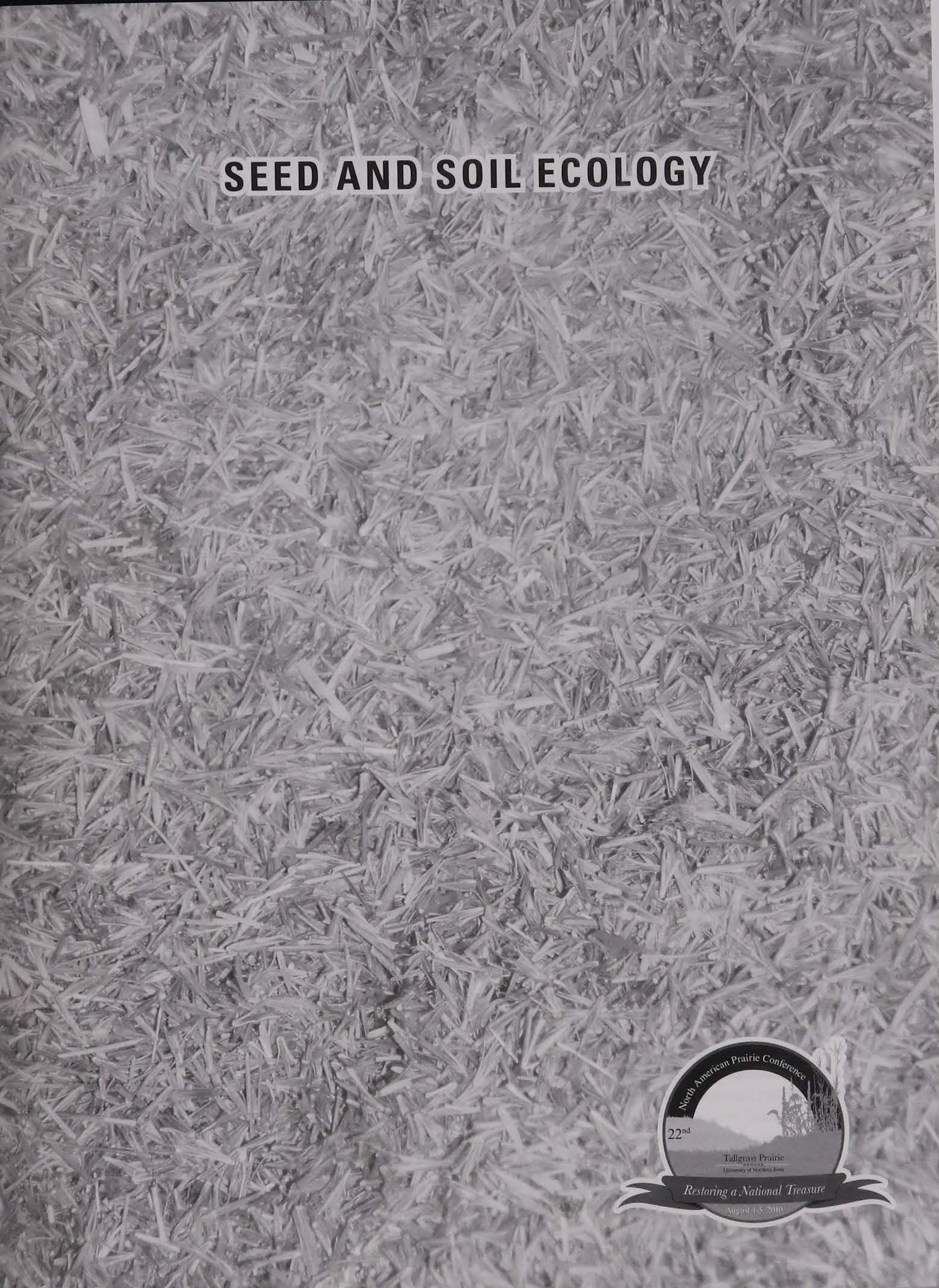
SEED AND SOIL ECOLOGY



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Abstract: The disappearing Cajun Prairies of Louisiana are potential repositories of pollutants that may release lead (Pb) and arsenic (As) into the soil. Two restored prairies—Louisiana State University at Eunice (LSUE) and the Cajun Prairie Restoration Project (CPRP) in Eunice—together with two remnant prairies (Estherwood and Frey) were investigated for their capacities to retain water, lead, and arsenic. Soil samples from three different depths (0-10, 10-20, and 20-30 cm) were taken from each prairie. Eight-gram soil samples were placed in the leaching funnels and 15 ml of distilled water were added to percolate through the soils for 22 hours. In separate experiments, 8 g of soil samples were placed in the leaching funnels, and then 15 ml of Pb treatment solutions or 15 ml of As treatment solutions were added to leach through the soils for 1320 minutes (22 hours). The CPRP showed equal water percolation rates at all soil depths, whereas the other three prairie soils showed faster percolation rate at the 0-10 cm depth than at the lower depths. The data suggest that the slower the percolation rate, the greater the retention of water, lead, and arsenic in the soils. Group comparisons of the prairie soils showed that the volume of percolate water, and the concentrations of lead and arsenic in the leachates, were significantly greater at restored than at remnant prairies. The four prairie soils are better reservoirs of lead than of arsenic. Although the percent soil porosity and percent clay were positively correlated with water-holding capacity of prairie soils, it was only the percent clay that positively influenced the retention of lead and arsenic in these soils.

Key Words / Search Terms: leaching, heavy metals, retention, percolation, soil particles, soil porosity

INTRODUCTION

Lead (Pb) and arsenic (As) are naturally occurring elements found in rocks, soil, air, and water. Natural levels of lead in surface soils are usually below 50 mg/kg (Chaney et al. 1984), and the concentrations of naturally occurring arsenic in virgin soils in the United States range from 1 to 5 mg/kg (Chen et al. 1999). Soils in some rice fields had low levels of lead, ranging from 8 to 13 mg/kg, despite the fact that these soils were affected by mine tailings (Samonte et al. 1992). However, higher concentrations of lead and arsenic may contaminate the soil, surface water, and aquifer. Based on 0.015 mg/L Pb concentrations and 0.010 mg/L As concentrations as the maximum contaminant levels (MCL) in drinking water established by U.S. Environmental Protection Agency (2010a), the drinking water in some cities in Louisiana suffered arsenic contamination more than lead contamination. For example, the arsenic levels in drinking water reached 0.061 mg/L in 2008 in New Iberia and 0.012 mg/L in 2006 in Loreauville, while maintaining the lead below the MCL (Louisiana Water Company 2009). Within the last decade, the data of groundwater in Michigan, Minnesota, New England, Oklahoma, South Dakota, and Wisconsin indicated that arsenic concentrations surpassing the MCL of 0.010 mg/L were more commonly widespread than previously realized (Welch et al. 2000). Naturally occurring arsenic in the aquifer in some regions of Bangladesh and India has increased the levels of arsenic in groundwater (Anawar et al. 2002, Chowdhury et al. 2000). As a result, the use of groundwater for irrigating agricultural plants in Bangladesh and India increased the soil arsenic concentrations in the surface soils (Meharg and Rahman 2003). Concerns about lead and arsenic contaminations are not just local issues but international as well.

Soils have been artificially enriched with lead and arsenic through the applications of arsenic-based herbicides and insecticides in agricultural systems (Chirenje et al. 2003, Peryea 1998). Lead-arsenate pesticides were extensively used to eradicate mosquitoes and the insect pests of fruit orchards, garden plants, and turfgrasses (Shepard 1951). Despite the termination of lead-arsenate use, the residues of these pesticides can bind tightly to the surface soil layer and stay there for decades.

Contaminated surface soil layers can be transported into other farms, grasslands, and residential locations by wind or water erosion. Months after Hurricane Katrina, there were adversely high levels of heavy metals and arsenic in the sediments covering much of New Orleans, with the highest concentrations in the inner city of more than 1500 mg/kg of Pb (Shogren 2006). The U.S. Environmental Protection Agency (2010b) ascertained that lead concentration in soils of 400 mg/kg is the dangerous level in playground areas, whereas 1200 mg/kg is the hazardous level for bare soil in residential areas. The Washington State Department of Health approximated that the safe level of arsenic concentration in soils is less than 37 mg/kg for children, and less than 175 mg/kg for adults (Peryea 1998, 2001).

Southwestern Louisiana once encompassed 1.0 million hectares (2.5 million acres) of Cajun Prairies in 1870 (Lockett 1970). However, this important ecosystem has mostly disappeared and been replaced by rice and soybean fields, pastures for grazing, oil fields, urban/suburban development, and their required facilities. Today, less than 40.5 ha (100 ac) of remnant prairies exist (Allen and Thames 2004, Vidrine 2010, Vidrine et al. 1995) in the form of small, narrow strips located near agricultural farms, on the rights-of-way of railroad tracks, and on abandoned private properties. Chemical analysis of several types of fertilizers (Franklin et al. 2005, McBride and Spiers 2001, Molina et al. 2009) and pesticides (Campos 2003) showed trace amounts of lead and arsenic. In agricultural systems, it is highly possible that long-term use of fertilizers and pesticides may cause lead and arsenic concentrations to increase in agricultural soils. Flood commonly occurs in southwestern Louisiana, and topsoil sediments carried by flood may transport lead and arsenic into nearby prairies.

Due to agricultural and infrastructural development, both restored Cajun Prairies and the disappearing remnant Cajun Prairies of Louisiana are potential repositories of fertilizer and pesticide residues, waste materials, and pollutants, which may release lead and arsenic into the soil. We hypothesized that the retention of water, lead, and arsenic would be greater in remnant prairie soils than in restored prairie soils, probably because the greater accumulation of organic matter would occur in remnant than in restored prairies. Organic matter would provide additional pore spaces for water storage and more surface areas for adsorption of lead and arsenic. The objectives of this study were two:

1. To determine the capacity of restored and remnant prairie soils (a) to hold water, and (b) to tightly adsorb lead and arsenic ions under laboratory experimental conditions.
2. To determine which soil properties (organic matter, bulk density, porosity, and the proportions of sand, silt, and clay) correlate with the retention of water, lead, and arsenic in these soils.

MATERIALS AND METHODS

SITE DESCRIPTION

Two restored and two remnant Cajun Prairies in southwestern Louisiana were chosen for this study. Two restored prairies comprised the Cajun Prairie Restoration Project (CPRP) in the city of Eunice and the Louisiana State University at Eunice (LSUE) prairie on the school campus. Before the restoration, the CPRP was originally an area of grasses and weeds, whereas the LSUE prairie was originally a Saint Augustine grass lawn. The restored CPRP and LSUE prairies were established in 1988 and 1989, respectively. Both prairies have been managed by conducting annual winter burning. The twenty-two-year-old CPRP has been burned in January without mowing, whereas the twenty-one-year-old LSUE prairie has been burned and then mowed every January (Jariel et al. 2010, Vidrine et al. 1995).

The two remnant prairies were located near the Frey and Estherwood communities. Both remnants are at least one hundred years old (Goins and Caldwell 1995) and have been managed by local farmers by prescribed burning at infrequent intervals prior to 1990. Estherwood prairie was last burned in 1996. These remnants have disturbances from agricultural and/or railroad machinery from time to time, and they have been highly susceptible to biocidal drift and fertilizer effluent during the agricultural season. In general, they have not received constant and specific management since 1870 (Lockett 1970) as the plants have been allowed to grow in the wild for at least a century.

The parent material of soil series in southwestern Louisiana is clayey alluvium, which was deposited by the Mississippi and Red Rivers during the Pleistocene epoch (Clark et al. 1959, Murphy et al. 1986, Soil Survey Staff 2008). Soil textures of the four prairies exhibit low water permeability. Soils of all four prairies belong to the soil order Alfisols. The restored CPRP and LSUE prairies reside on a poorly drained Crowley soils series classified as fine, montmorillonitic, thermic Typic Albaqualfs with slopes of less than 1% (Murphy et al. 1986). The soils of the restored prairies have dark grayish, silt loam surface texture (0-50 cm) and light grayish brown, silty clay loam subsoil (50-150 cm).

The remnant Frey prairie lies on an imperfectly drained Acadia soil series classified as fine, montmorillonitic, thermic Aeric Ochraqualfs with 1% to 3% slopes (Clark et al. 1959). The texture of dark yellowish brown surface (0-7 cm) of Frey prairie soils is silt loam, whereas the yellowish brown subsoil (7-16 cm) is silty clay. The remnant Estherwood prairie is situated in deep, poorly drained Midland soil series classified as fine, montmorillonitic, thermic Chromic Vertic Epiaqualfs with 0% to 1% slopes (Clark et al. 1959). The textural class of dark gray surface (0-2.4 cm) soils of Estherwood prairie is silty clay loam, whereas the gray subsoils (2.4-7.0 cm) are silty clay.

The southwestern Louisiana region where the four prairies are located receives an average of 125 cm of precipitation annually (Murphy et al. 1986). None of the prairies have been fertilized. Each prairie was divided into three blocks or replications using a randomized complete block design. Each block measured at least 12 m x 12 m.

SOIL SAMPLING

Soils were sampled from each block of each prairie during the week of June 15, 2008, under the canopy of switchgrass (*Panicum virgatum*). Switchgrass was used because it was commonly found in every prairie and every sample area in this study. Using a sampling tube with 2.25 cm diameter, soil sample cores at different depths (0-10, 10-20 and 20-30 cm) were taken 2.5 cm away from the crown of randomly chosen switchgrass plants. Six soil sample cores around the switchgrass were collected if the crown diameter was one meter or less, and nine soil sample cores if the crown diameter was greater than one meter. The soil sample cores were placed in labeled plastic bags and were prepared for chemical analysis.

SOIL CHEMICAL AND PHYSICAL ANALYSIS

Soil samples were air-dried, pulverized using a soil grinder, and sieved through a 2 mm metal screen (Page 1982). Soil organic matter was determined by loss-of-weight-on-ignition method using a muffle furnace at 360° C for two hours (Schulte 1988, Storer 1984). Soil bulk density, particle density, and % porosity of 2-mm sieved soils were determined by water-displacement method (Palmer and Troeh 1995). The proportions of sand, silt, and clay were determined by the hydrometer method (Bouyoucos 1962).

LABORATORY SETUP

The experimental setup included 58° angle glass funnels with foam disks (Figure 1) for the water percolation experiment, and for the lead and arsenic leaching experiments. The top diameter of the funnel cone was 65 mm, and the length of the funnel stem was 63 mm. The foam disks were cut from a 6-mm-thick foam sheet using a cylindrical metal cork borer with 9 mm diameter. Each foam disk was inserted into the stem, just below the cone of the funnel. The funnels were clamped to the iron stands.

Figure 1. Foam disks cut from foam sheet using a cork borer. Each foam disk is inserted into the stem of glass funnel.

WATER PERCOLATION EXPERIMENT

Eight grams of each air-dried, sieved soil sample were placed into the foam-inserted funnels. Each funnel was lightly tapped to distribute the soil particles evenly inside the funnel. For the water percolation experiment, graduated test tubes were placed under the funnels. When 15 ml of distilled water was poured into the soil, the timer was set to zero minutes, and the water was allowed to percolate for 1320 minutes (22 hr). The volume of percolate water in the test tube was measured every 5 minutes for 300 minutes (5 hr), followed by the last measurement at 1320 minutes. The volume of percolate water collected in 1320 minutes was used to calculate the volume of water held by the soil. Concentrations of lead and arsenic in the percolate water were determined using Perkin Elmer AAnalyst (PEA) 300 Atomic Absorption Spectrophotometer (Perkin Elmer 1996).

LEAD AND ARSENIC LEACHING EXPERIMENT

For the lead leaching experiment, four treatment solutions of lead were prepared (50, 250, 1000, and 2000 mg/L Pb) by dissolving lead nitrate, $Pb(NO_3)_2$, in distilled water. For each treatment solution, 8 g of soils were placed into the foam-inserted funnel and tapped for even distribution in the funnel with the graduated test tube underneath, and 15 ml of treatment solution was poured into the soil and allowed to leach for 1320 minutes. Concentrations of lead in the leachates collected in 1320 minutes were determined using PEA 300 Atomic Absorption Spectrophotometer (Perkin Elmer 1996). The lead concentrations in the leachates were used to calculate the lead concentrations retained in the soils.

In separate arsenic leaching experiments, a procedure similar to the lead leaching experiment was used with some modifications. Two treatment solutions of arsenic were prepared (50 and 250 mg/L As) by dissolving arsenic acid heptahydrate sodium salt, $Na_2HAsO_4 \cdot 7H_2O$, in distilled water. Concentrations of arsenic in the leachates collected in 1320 minutes were determined using PEA 300 Atomic Absorption Spectrophotometer (Perkin Elmer 1996). Based on the arsenic concentrations in the leachates, arsenic concentrations retained in the soils were calculated.

STATISTICAL ANALYSIS OF DATA

Probability (*P*) values of treatment effects (Table 1) on soil variables were analyzed statistically using SAS General Linear Model (GLM) procedure based on a Randomized Complete Block Design (RCBD) with split-plot arrangement (SAS Institute 2003). Each prairie had three blocks (B) or replications. Prairie (P) locations were the main plots, which were split into sampling depths (D) as subplots. The effects of P, D, and P x D interaction on soil variables were determined. CONTRAST statement was used to test the significant difference of variables between groups of prairies.

Significant differences among treatment means were tested using LSMEANS (Least Square Means) statement with STDERR PDIFF option. Significant correlations among variables were tested using Pearson's correlation coefficients (r) (Table 2).

RESULTS AND DISCUSSION

WATER PERCOLATION

Prairies of LSUE, Estherwood, and Frey showed a faster water percolation rate at 0-10 cm depth than at 10-20 and 20-30 cm depths (Figure 2, A, C, D; Figure 3). However, CPRP showed equal percolation rates at all soil depths compared to the remaining three prairies (Figure 2, B). The CPRP soils also showed significantly faster percolation rate at 300 minutes ($P = 0.001^{**}$) when compared to the other three prairies (Table 1, Figure 3). Faster percolation rate through CPRP soils than the other three prairies could be attributed to its significantly lower organic matter content ($P = 0.021^{*}$), higher bulk density ($P = 0.0001^{**}$), lower percent porosity ($P = 0.0001^{**}$) (Figure 4, A, B, C), higher percent sand ($P = 0.048^{*}$), and lower percent clay ($P = 0.029^{*}$) (Figure 5, A and C).

Studies have shown that the movement of water in soil is affected by organic matter (OM), bulk density, porosity, and the proportions of sand, silt, and clay (Brady and Weil 2008, Gardiner and Miller 2008). The amount of OM found at the soil surface can enhance water infiltration, provides more tiny pores than mineral soil particles can, and holds much greater quantities of water (Pidwirny 2010). In our percolation study, soil OM content had no direct or significant influence (r = 0.17 ns) on water percolation, even though that increasing OM diminished the bulk density of prairie soils ($r = -0.41^{*}$) (Table 2). A similar relationship between OM and bulk density was obtained by Tsadilas et al. (2005). They,

Table 1. Probability (*P*)[1] values of percolation of water, lead (Pb), and arsenic (As) in the leachate, and soil properties in response to Prairie (P), Soil Depths (D), and P x D Interactions using SAS General Linear Model Procedure. P includes four Cajun Prairies (LSUE and CPRP are restored, and Estherwood and Frey are remnants prairies). D includes 0-10, 10-20, 20-30 cm soil depths. B includes 3 blocks or replications. The B x P mainplot error term was used to test the significant difference among B or among P. The B x P x D error term was used to test the significant difference among D and the significance of the P x D interactions. CONTRAST statement was used to test the significant difference of variables between groups of prairies.

SOURCE OF VARIAION	WATER THAT PERCOLATED AT 300 MINUTES	Pb IN LEACHATE AT 2000 mg/L Pb TREATMENT	As IN LEACHATE AT 50 mg/L As TREATMENT	As IN LEACHATE AT 250 mg/L As TREATMENT	ORGANIC MATTER	BULK DENSITY	% POROSITY	% SAND	% SILT	% CLAY
Block (B)	0.749 ns	0.451 ns	0.204 ns	0.128 ns	0.693 ns	0.425 ns	0.922 ns	0.692 ns	0.916 ns	0.614 ns
Prairie (P)	0.030 *	0.450 ns	0.156 ns	0.005 **	0.135 ns	0.0001**	0.0001**	0.0001**	0.0001**	0.0001**
Depth (D)	0.0001**	0.0001**	0.068 ns	0.0001**	0.001 **	0.015 *	0.004 **	0.0001**	0.0001**	0.0001**
P x D Interaction	0.003 *	0.0001**	0.045 *	0.280 ns	0.006 **	0.208 ns	0.196 ns	0.0001**	0.032 *	0.0001**
CONTRAST										
Restored vs. Remnant	0.007 **	0.009 **	0.019 *	0.0001**	0.021 *	0.0001**	0.0001**	0.087 ns	0.0001**	0.003 **
LSUE vs. 3 Prairies	0.818 ns	0.053 ns	0.017 *	0.0001**	0.711 ns	0.132 ns	0.0004**	0.984 ns	0.0002**	0.146 ns
CPRP vs. 3 Prairies	0.001 **	0.231 ns	0.742 ns	0.087 ns	0.021 *	0.0001**	0.0001**	0.048 *	0.120 ns	0.029 *
Estherwood vs. 3 Prairies	0.013 *	0.013 *	0.150 ns	0.386 ns	0.981 ns	0.0001**	0.0001**	0.677 ns	0.085 ns	0.767 ns
Frey vs. 3 Prairies	0.480 ns	0.558 ns	0.176 ns	0.0001**	0.009 **	0.695 ns	0.323 ns	0.116 ns	0.0001**	0.0003**

[1]The Probability (*P*) values are not significant (ns) at $P > 0.05$, significant (*) at $P < 0.05$, and highly significant (**) at $P < 0.01$.

found that after three years of application of biosolids into clay loam soil, the cotton yield, OM content, water retention capacity, available water, and infiltration rate increased significantly, whereas bulk density and aggregate instability index decreased. However, Leroy et al. (2008) did not see any improvement in water retention upon the application of organic compost and/or cattle slurry in sandy loam soil over a nine-year period.

Higher bulk density of the soil means lower percent porosity, and the decrease in porosity results in the lower water retention of the soil (Gardner 1979). Our study showed that high bulk density and low porosity in CPRP soils were caused by its high sand, high silt, and low clay contents (Figure 5, A, B, C Group comparisons of prairies (Table 1, Figure 2) also showed that the rate of water percolation was faster in restored than in remnant prairies ($P = 0.007^{**}$) as influenced by the lower soil OM ($P = 0.021^{*}$), higher soil bulk density ($P = 0.0001^{**}$), associated with lower soil porosity ($P = 0.0001^{**}$), higher percent silt ($P = 0.0001^{**}$), and higher percent clay ($P = 0.003^{**}$).

As the soil bulk density increased, percent soil porosity decreased ($r = -0.89^{**}$) (Table 2; Figure 4, B, C). Among the inorganic soil particles, sand has the highest bulk density, whereas clay has the lowest, which explains that the higher percent sand with lower percent clay correlates with the lower water-holding capacity of the soil (Gardner 1979). This relationship was supported by the water retention experiment in situ by Jabro et al. (2009). They found that the field water capacity or the amount of water held in the soil after the excess water was drained away was achieved after 50 hr in sandy loam and 450 hr in clay loam soils. The faster the water was drained by gravity from the soil, the less the water retention in the soil.

The greatest differences in percolation rates among all soil depths occurred at 300 minutes in all prairies (Figures 2 and 3). Volume of percolate water at 300 minutes was positively corre-

Table 2. Pearson's correlation coefficients (r)[1] of properties of the soil against the percolation of water, and the concentrations of lead and arsenic in the leachate at different treatment solutions (2000 mg/L Pb, 50 mg/L As, and 250 mg/L As).

VARIABLES	PERCOLATE WATER IN 300 MINUTES	LEAD IN LEACHATE AT 2000 mg/L Pb TREATMENT	ARSENIC IN LEACHATE AT		BULK DENSITY	% POROSITY
			50 mg/L As TREATMENT	250 mg/L As TEATMENT		
Percolate Water in 300 minutes	1.00 **	0.49 **	0.37*	0.56 **	0.39 *	- 0.41 *
Organic Matter	0.17 ns	0.26 ns	0.35 ns	- 0.01 ns	- 0.41 *	0.44 **
Bulk Density	0.39 *	0.16 ns	0.02 ns	0.03 ns	1.00 **	- 0.89 **
% Porosity	- 0.41 *	- 0.02 ns	0.16 ns	0.01 ns	- 0.89 **	1.00 **
% Sand	0.53 **	0.65 **	0.37 *	0.45 **	0.10 ns	- 0.11 ns
% Silt	0.36 *	0.25 ns	0.42 *	0.78 **	- 0.13 ns	0.12 ns
% Clay	- 0.55 **	- 0.59 **	- 0.47 **	- 0.71**	- 0.01 ns	- 0.02 ns

[1] The correlation coefficient (*r*) between two variables is not significant (ns) at $P > 0.05$, significant (*) at $P < 0.05$, and highly significant (**) at $P < 0.01$.

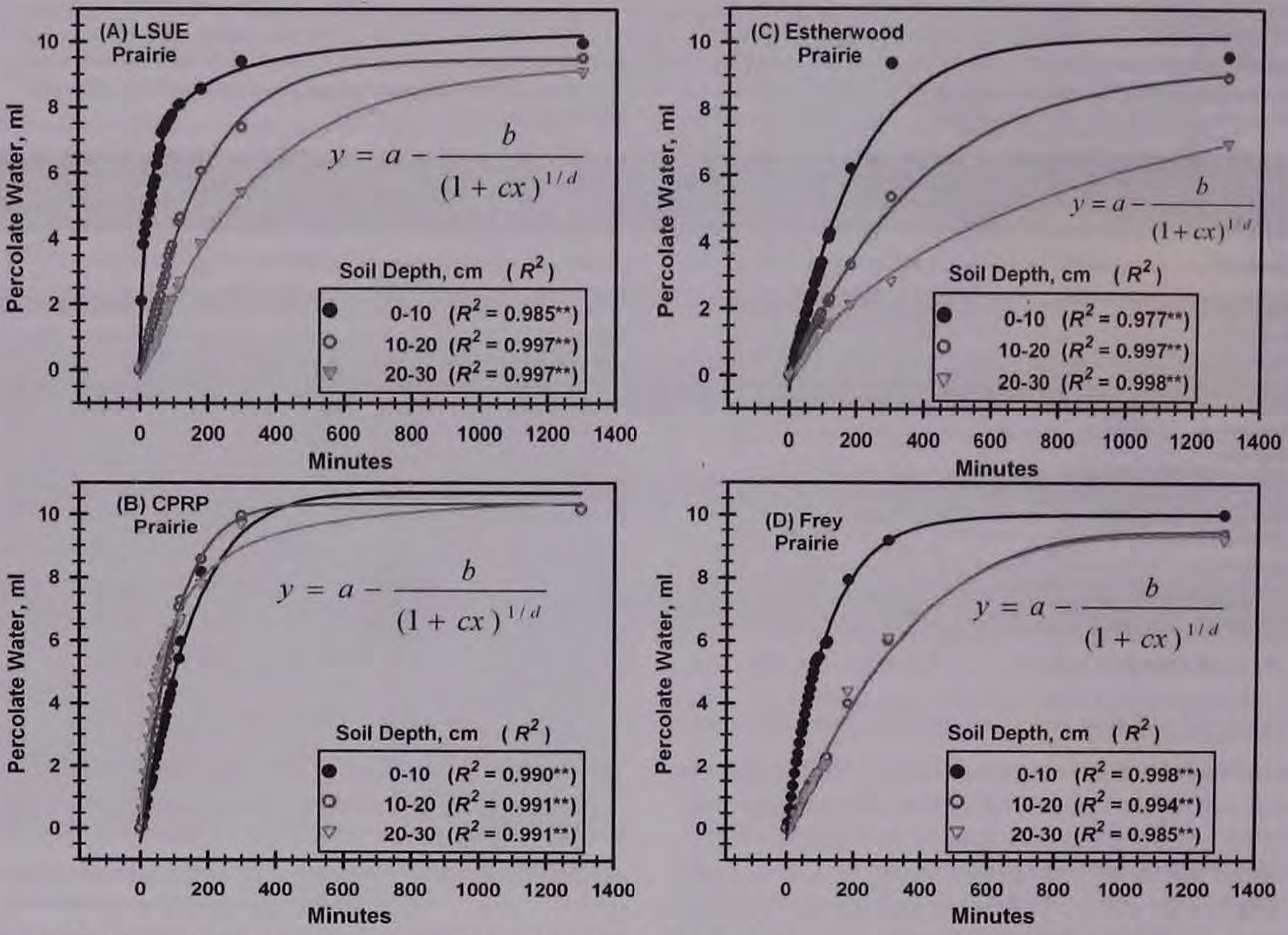

Figure 2. Percolation of water (ml) in 1320 minutes at different soil depths in Cajun Prairies with regression coefficients (R^2). The regression coefficient (R^2) of a nonlinear curve is not significant (ns) at $P > 0.05$, significant (*) at $P < 0.05$, and highly significant (**) at $P < 0.01$.

lated with soil bulk density ($r = 0.39^{*}$), percent sand ($r = 0.53^{**}$), and percent silt ($r = 0.36^{*}$), but negatively correlated with percent soil porosity ($r = -0.41^{*}$) and percent clay ($r = -0.55^{**}$) (Figure 4 and Table 2). Therefore, the capacity of the soils to hold greater amounts of water was dependent on higher percent soil porosity associated with higher percent clay.

Chemical analysis of percolate water showed that there were no detectable levels of lead and arsenic. Pure water alone was not sufficient to leach lead and arsenic ions through the prairie soils. This suggests that these prairie soils have naturally low levels of lead and arsenic, an ideal condition to find out how much more lead and arsenic that the soils can hold by subjecting them to lead and arsenic leaching experiments.

LEAD LEACHING

As the concentrations of lead treatment solutions increased, lead leaching also increased. At 50 mg/L Pb treatment solution, no lead ions leached through the soils of any of the prairies, indicating that all of the lead ions were retained in the soil. At 250 and 1000 mg/L Pb treatment solutions, no lead leached through the remnant prairie soils. However, only 0.02 mg/L and 0.08 mg/L of Pb leached through the restored prairie soils, which was equivalent to only 0.008% Pb of the treatment solutions while 99.992% Pb was retained in the soils. These trace amounts of lead in the leachate showed the immobility of lead in these prairie soils.

The downward movement of lead through the soil was induced by increasing the concentrations of lead treatment solutions from 1000 to 2000 mg/L. At 2000 mg/L Pb treatment solution, the maximum amount of lead that leached through was only 2.5% of the treatment solution and was found at the 0-10 cm soil layer at LSUE prairie (Fig-

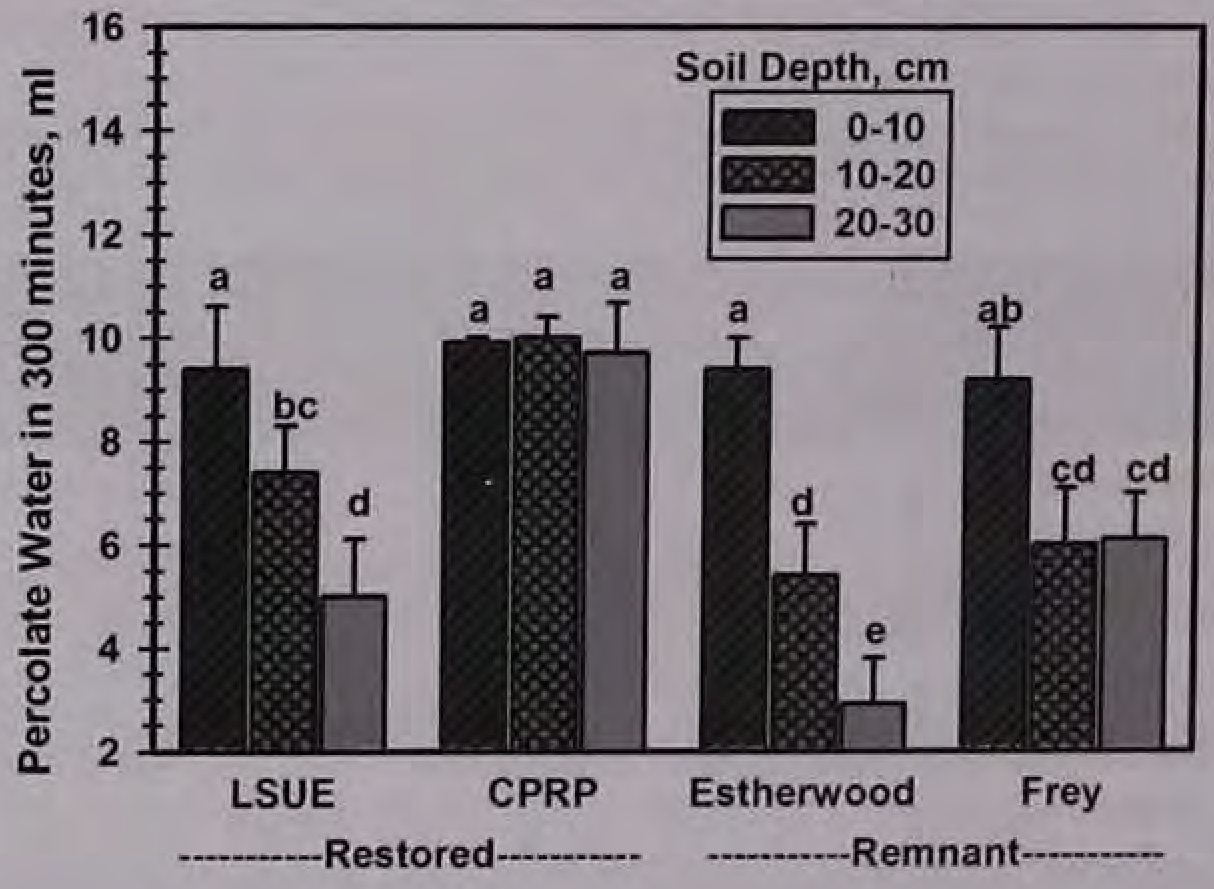

Figure 3. Percolate water in 300 minutes at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different ($P > 0.05$). Vertical lines on the bars are standard errors.

ure 6, A). Group comparisons showed that concentrations of lead in the leachate were higher in restored than in remnant prairies (*P* = 0.009**) (Table 1; Figure 6, A), suggesting that the retention of lead was greater in remnant than in restored prairies.

Concentrations of lead in the leachate were positively correlated with percolate water (r = 0.49**) and percent sand (r = 0.65**), but negatively correlated with percent clay (r = -0.59**) (Table 2). In general, as the amount of lead in the leachate decreased from topsoil to subsoils (Figure 6, A), the percent sand also decreased (Figure 5, A) but percent clay increased (Figure 5, C).

The results of Karathanasis et al. (2005) further showed the direct relationship of percent sand with lead leaching or percent clay with lead retention. They found that the concentrations of lead transported by the colloids of aerobically digested biosolids through undisturbed soil monolith were higher in the leachate from Bruno fine sandy loam than in the leachate from Maury silt loam or Woolper silt loam, which contained lower percent sand but higher percent clay than the Bruno soils. When compared to

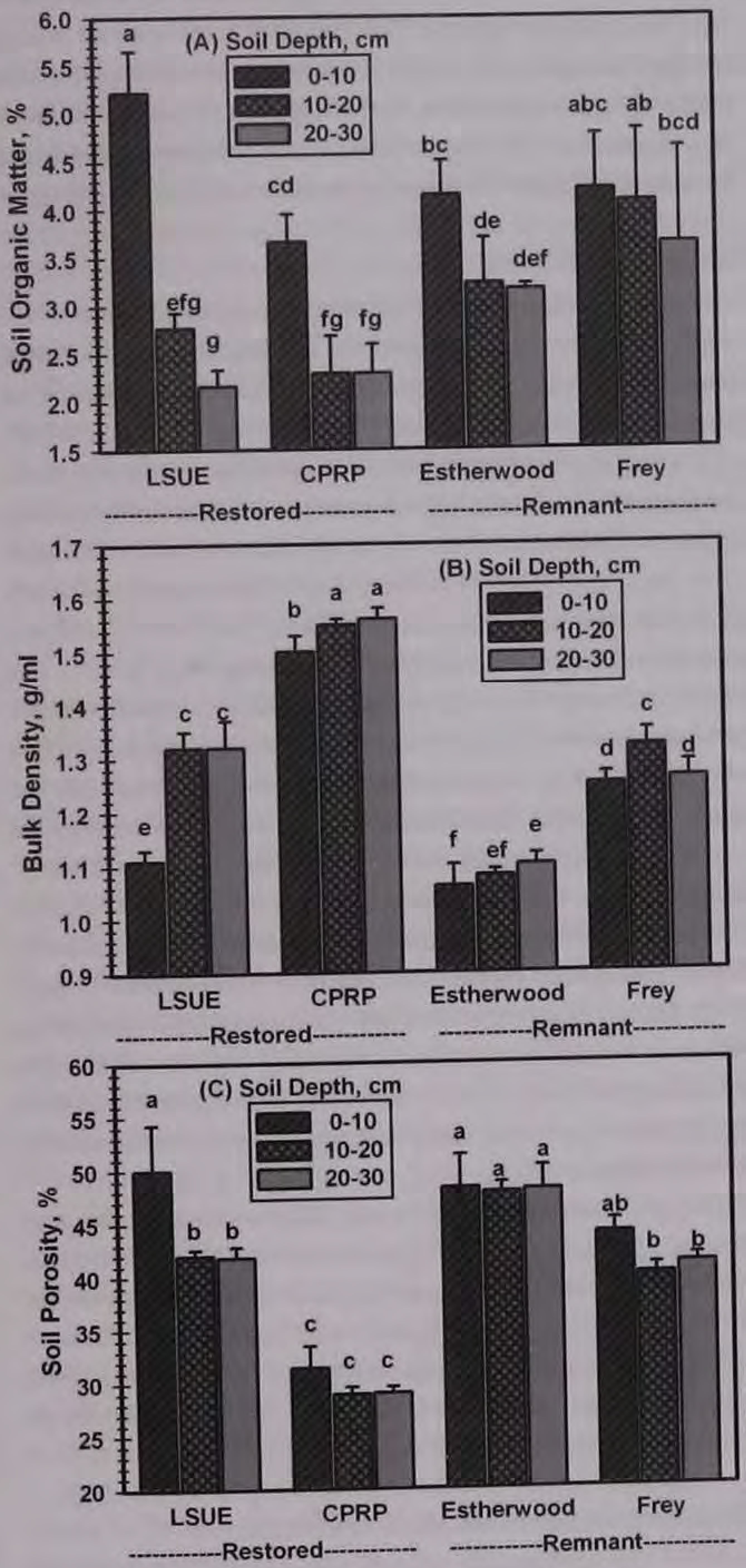

Figure 4. Soil organic matter (A), bulk density (B), and porosity (C) at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different (*P* > 0.05). Vertical lines on the bars are standard errors.

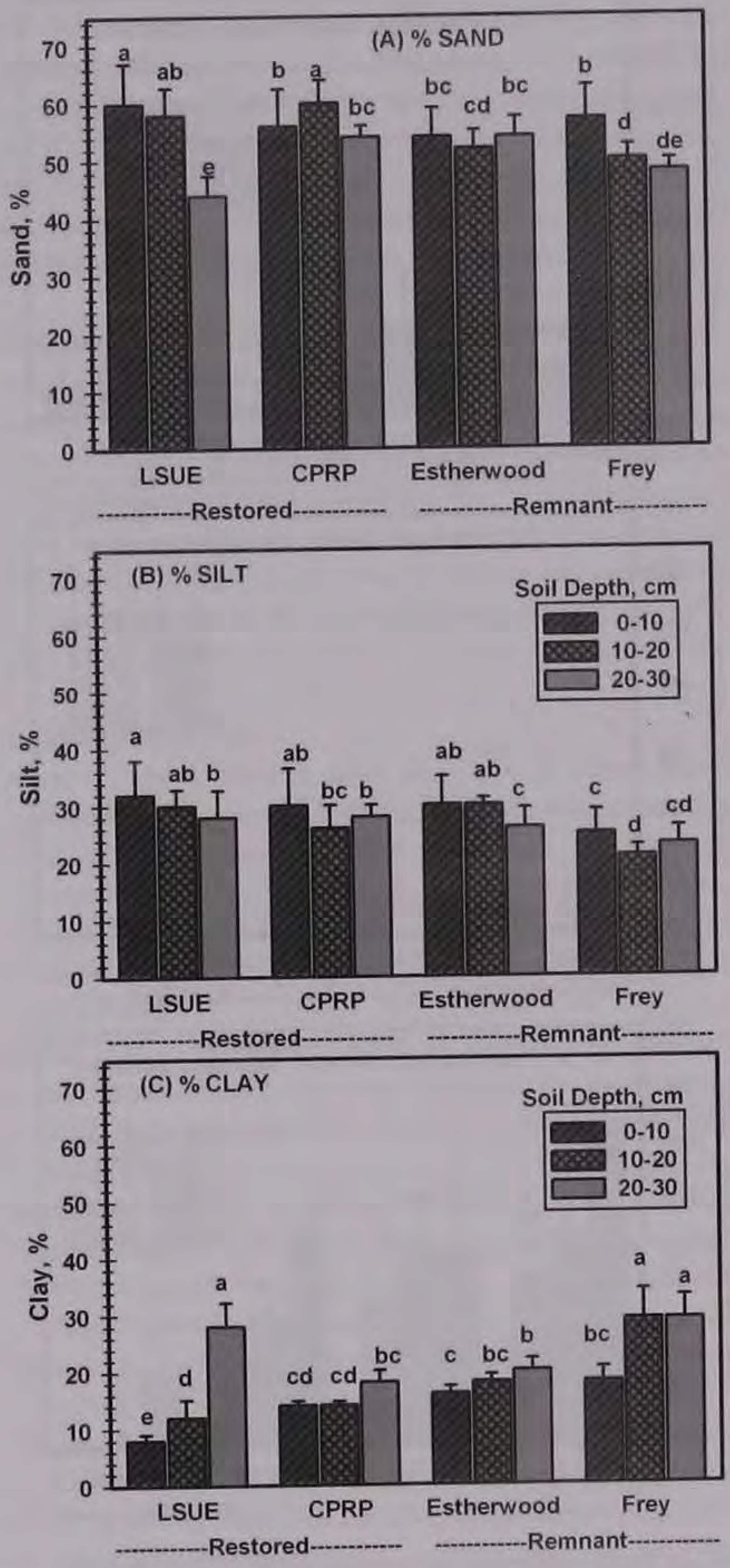

Figure 5. Percent sand (A), silt (B), and clay (C) at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different (*P* > 0.05). Vertical lines on the bars are standard errors.

other metals, retention of lead was greater than copper and zinc in all three biosolid-amended soils. Lead was more strongly held than cadmium and zinc by montmorillonite, kaolinite, and calcite minerals (Wahba and Zaghloul 2007), which are the major components of clay particles not found in sand. In spite of their differential attractions to clay minerals, lead, cadmium, copper, and zinc had a greater tendency to be retained in the 0-15 cm soil layer but moved very slowly into the 15-30 cm depth of reclaimed mine land amended with sewage sludge (Seaker 1991). These results show that lead is not the only immobile elements in soils, but other heavy metals are immobile as well.

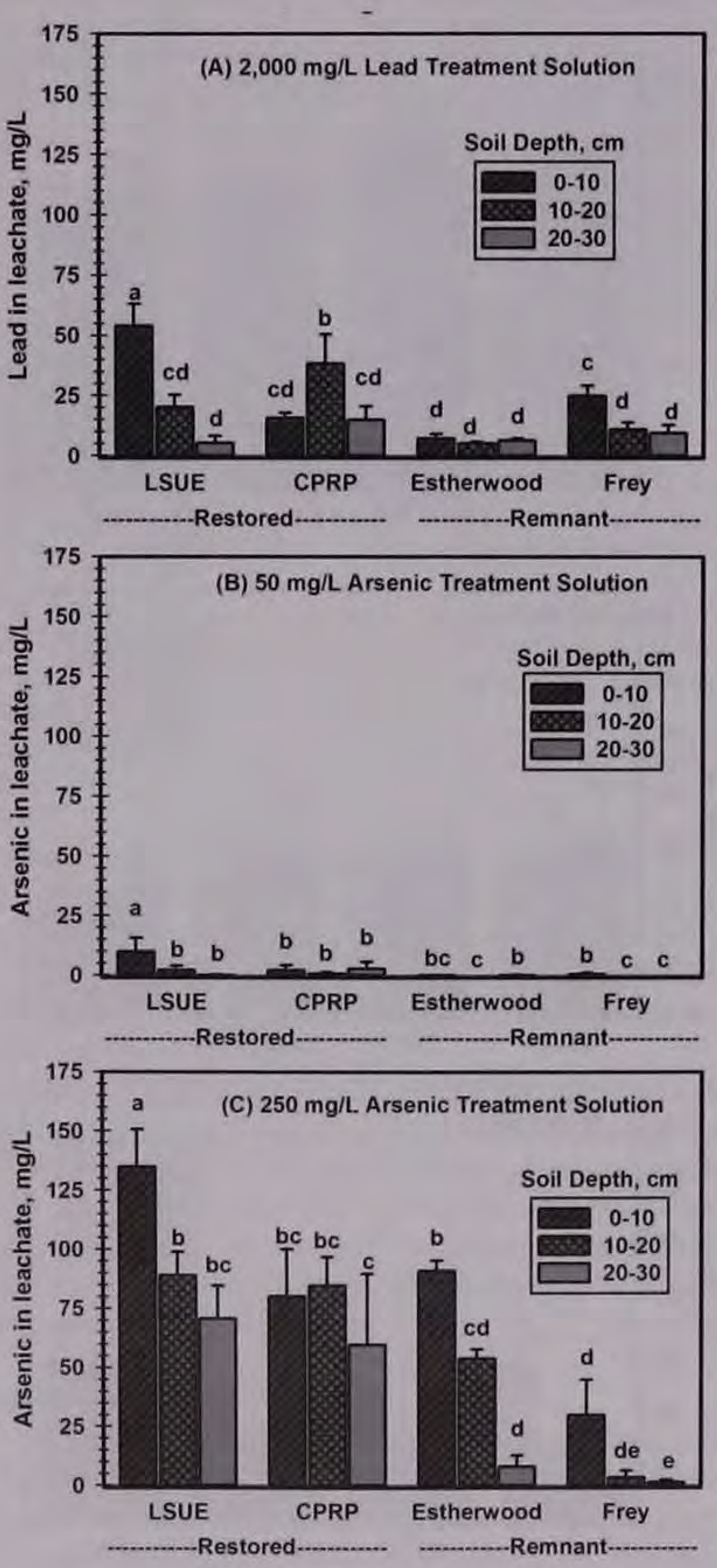

Figure 6. Lead concentrations in the leachate at 2000 mg/L Pb treatment solutions (A), and arsenic concentrations in the leachate at 50 mg/L As (B) and at 250 mg/L As (C) treatment solutions, at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different ($P > 0.05$). Vertical lines on the bars are standard errors.

Lead can be mobile in soils and travel in solutions when lead reacts with organic colloids that form soluble Pb-organic complexes. This explains the positive correlation between lead leaching and organic matter content in biosolid waste-amended soils containing at least 20% OM (Karathanasis et al. 2005). In agricultural soils, Banat et al. (2007) suggested that the negative correlation between OM and lead concentrations in the topsoils was due to the leaching of lead into the subsoils. However, OM content in our prairie soils had no significant correlation ($r = 0.26$ ns) with lead leaching (Table 2; Figure 4, A and 6, A). Perhaps soil OM concentrations were too small (< 5.3% OM) to provide a discernable impact on the formation of soluble Pb-organic complexes in these prairie soils.

ARSENIC LEACHING

As the arsenic treatment solutions increased from 50 to 250 mg/L, concentrations of arsenic in the leachate also increased (Figure 6, B and C). The maximum amounts of arsenic that leached were 20% of the 50 mg/L As treatment solutions and 52% of the 250 mg/L As treatment solutions. Based on group comparisons of the prairies, higher arsenic concentrations were detected in the leachate of the restored than remnant prairie soils ($P = 0.019^{*}$ and 0.0001^{**}) (Table 1). Rapid leaching of arsenic was positively correlated with percolate water, percent sand, and percent silt, but negatively correlated with percent clay (Table 2). The results in these prairie soils were supported by the results of other studies. Mobility of arsenic was greater in coarse sand than in fine clay (Warren et al. 2003), and the higher the clay content in the soil the lower the arsenic concentrations migrated out from the soil (Chen and Ma, 2002). For example, at 250 mg/L As treatment solution, Frey—with the lowest arsenic in the leachate (Figure 6, C)—retained most of the arsenic in the soil due to its relatively higher percent clay content than the other three prairies (Figure 5, C). Arsenic is strongly adsorbed on some minerals present in clay, such as aluminum oxides, iron oxides, and iron sulfides (De Brouwere et al. 2003, Welch et al. 2000). This may explain why a higher percentage of clay indicates greater arsenic retention in the soils of the study prairie.

Organic matter may have a strong affinity to arsenic that may prevent arsenic from leaching, based on the slight positive correlation of soil OM with arsenic (Banat et al. 2007). However, the insignificant correlation of arsenic leaching with OM (Table 2) reflects insignificant attraction between OM and arsenic in these soils. Therefore, OM had no significant role in retaining arsenic in the Cajun Prairie soils that we investigated.

LEAD AND ARSENIC MOBILITY IN SOILS

Faster percolation rates of water correlated with increased concentrations of lead and arsenic in the leachates; however, lead leached more slowly than arsenic (Table 2 and Figure 6). Soils have more negative sites for cation exchange than positive sites for anion exchange. The net negative charge in soils provides their greater capacity to attract or adsorb positive ions than negative ions (Brady and Weil 2008). In aqueous solution, lead exists as a cation, Pb^{2+} whereas arsenic exists as an arsenate anion, $(AsO_4)^{3-}$. The different charges explained the greater affinity of the soils to retain lead than to retain arsenic (Figure 6). Soils contaminated by Pb-As pesticides showed that most of the lead remained in the topsoils, but most of arsenic moved into the subsoils (Merry et al. 1983, Veneman et al. 1983). These findings indicated that the Pb/As concentration ratio decreased with depth, suggesting greater downward movement of arsenic than lead.

The increased lead and arsenic concentrations in the leachates with increasing concentrations in treatment solutions indicate that the adsorption sites in the soils are being saturated by lead and arsenic ions. However, adsorption sites in the soils were responsible for greater retention of lead than arsenic (Banat et al. 2007), which further supports our result that lead was less mobile than arsenic. The immobility of lead also reflects its tendency to form slightly soluble compounds with anions in soil solutions, similar to the formation of lead phosphate of very low solubility (Peryea 2001, U.S. Environmental Protection Agency 2010b). However, phosphate enhances the solubility and mobility of arsenic in soil solutions. Phosphate, which also exists as an anion, has greater attraction toward iron and aluminum oxide than arsenic. Competition between phosphate and arsenic causes the arsenic to leach (Banat et al. 2007, De Brouwere et al. 2003).

Although the leaching of lead and arsenic was diminished by clay content in the prairie soils being studied, other studies also showed that the mobility of lead and arsenic into the subsoils was minimized by plant absorption. When lead- and arsenic-containing fly ash was used as a soil amendment and nutrient source for ornamental plants, lead and arsenic were not detected in the leachate of the container pots after six months of growing *Syngonium podophyllum* (Li et al. 2008). These results indicate that lead and arsenic are either retained in the growth medium or absorbed by the plants. Rice grown in lead-arsenic-contaminated soils had greater concentrations of lead and arsenic in the straw than in the grain under nonflooded conditions. Because of the greater mobility of arsenic in aqueous soil solution, flooding increased the concentrations of arsenic in the grain (Codling 2009). The absorption of lead and arsenic by carrots was greater in Pb-As-contaminated soils than in noncontaminated soils (Zandstra and De Kryger 2007). In Pb-As-contaminated apple orchards, soil lead and arsenic were positively correlated with lead and arsenic concentrations in mushroom fruiting bodies (Shavit and Shavit 2010). Therefore, plants and fungi which have the capacity to accumulate high levels of lead and arsenic in their tissues may prevent or minimize further leaching of lead and arsenic into the groundwater.

Studies regarding the performance of Cajun Prairie plants grown in Pb-As-contaminated soils in southwestern Louisiana are very few. Because Cajun Prairies are susceptible to lead and arsenic accumulations from biocidal drift, fertilizer effluent, and waste deposits, future investigations are necessary to determine the potential of Cajun Prairie plants to extract excess levels of lead and arsenic in the contaminated soils.

CONCLUSION

Group comparisons of the Cajun Prairies showed that the volume of percolate water, and the concentrations of lead and arsenic in the leachates, were greater at restored than at remnant prairies, indicating that the remnant prairie soils can retain more water, lead, and arsenic than the restored prairie soils. Despite the positive correlation of percolation rate of water with concentrations of lead and arsenic in the leachates, lead leached more slowly than arsenic. All four prairie soils retained more lead than arsenic, suggesting that arsenic is more likely to drain faster into groundwater than lead. Although the water-holding capacity of prairie soils was correlated positively with the percent soil porosity and percent clay, it was only the percent clay that had a direct relationship with lead and arsenic retention in these soils. Therefore, clay particles are the dominant factors responsible for the Cajun Prairie soil's ability to serve as reservoirs of water, lead, and arsenic.

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DISTRIBUTION OF SOIL NUTRIENTS AT DIFFERENT DEPTHS IN RESTORED AND REMNANT PRAIRIES

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Abstract: The interaction of biota, soil, water, and management practices contributes to the addition of new surface soil layers in prairie ecosystems. This interaction further modifies the vertical distribution of soil nutrients, as influenced by the physical properties of the soil in a profile. Soil samples from different depths (0-10, 10-20, and 20-30 cm) of four Cajun Prairie sites in southwestern Louisiana, two restored (Louisiana State University at Eunice [LSUE] and the Cajun Prairie Restoration Project [CPRP] in Eunice) and two remnants (Estherwood and Frey), were collected from under a canopy of *Panicum virgatum* and evaluated for their chemical and physical characteristics. Group comparison of prairies showed that soil pH, electrical conductivity, extractable P, organic matter, and exchangeable Ca, Na and K were not significantly different between our two restored and two remnant prairies. Soil available Fe, Cu, and Zn, percent sand, and percent silt were significantly higher, but percent clay was significantly lower in these restored prairies than in our two remnants. Soil pH increased with depth ($P < 0.05$) and was positively correlated with calcium ($r = 0.55^{**}$), magnesium ($r = 0.43^{**}$) and sodium ($r = 0.30^{*}$) concentrations. The LSUE prairie, which received the most-intensive management practices of annual burning and mowing, had the highest pH (6.1). Concentrations of silt, organic matter, exchangeable K, and available Fe, Cu, and Zn were greater ($P < 0.01$) in the topsoil (0-10 cm) than in the subsoils (10-20 and 20-30 cm depths), but percent clay was lower in the topsoil than in the subsoils ($P < 0.01$). Topsoil accumulation of K, Fe, Cu, and Zn suggests that replenishment of these elements by nutrient cycling is faster than leaching; these elements are relatively immobile and may have greater chemical attraction to silt and organic matter in the topsoil. The data refute our hypothesis that clay illuviation would transport these elements from topsoil into subsoils.

Key Words / Search Terms: Alfisols, anion and cation exchange, leaching, macroelements, microelements, mobile elements, soil particles

INTRODUCTION

Soil-nutrient stratification occurs due to the interaction of soil-forming factors, such as parent material, biota, climate, topography, and time (Jenny 1941). Among these factors, biota, such as plants and soil micro-invertebrates, play a major role in providing new surface-soil layers by nutrient cycling, and further influence the distribution of nutrients to deeper soil layers. In addition to biological cycling, the weathering of parent materials, atmospheric deposition, and leaching are some of the major processes that regulate the vertical distribution of soil inorganic and organic particles (Trudgill 1988). Such vertical exchange of materials may result in chemical and physical gradients from topsoil to subsoils (Jobbagy and Jackson 2001). Most studies about nutrient concentrations at different soil depths involve fertilized forest soils, rangelands, pastures, orchards, or cultivated agricultural farms, where plant components are consumed or harvested (Braekke 1999, Crozier et al. 1999, Follett and Peterson 1988, Geiger et al. 1992, He 2009, Yeganeh et al. 2010), but very few studies have focused on ungrazed, unfertilized prairies (Jariel et al. 2002). This current study attempted to investigate soil-nutrient distribution with depth in unfertilized Cajun Prairies whose plants rely on nutrient recycling.

Plant cycling and leaching significantly influence the vertical distribution of soil nutrients (Jobbagy and Jackson 2001) in prairie ecosystems. Nutrient absorption by plant roots during active growth and plant litter deposition contribute to the accumulation of organic matter and elemental recycling in the soil (Jariel Jr. et al. 2010a, 2010b). Under this natural and uninterrupted process of plant cycling, accompanied by various management practices employed to maintain prairie ecosystems, prairies build new surface soil layers on the existing topsoil. How nutrients from the soil surface are distributed to deeper soil layers by nutrient cycling and leaching under a switchgrass (*Panicum virgatum*) canopy is the specific interest of our study. Switchgrass was chosen in this study because it was the predominant grass species commonly found in Cajun Prairies being studied.

Southwestern Louisiana contained 1.0 million hectares (2.5 million acres) of Cajun Prairies in 1870 (Lockett 1970). However, this important ecosystem has mostly disappeared, replaced by rice and soybean fields, pastures, oil fields, and urban/suburban development. Today, less than 40.5 ha (100 ac) of remnant prairies exist (Allen and Thames 2004, Vidrine 2010, Vidrine et al. 1995) in the form of small, narrow strips located near farms, along railroad rights-of-way, and on abandoned private properties.

In southwestern Louisiana, little data exist on the historical aspects of soils as well as the soil physics and soil chemistry of the Cajun Prairie. It is important to gather the basic baseline data on these sites as they may be available for decades for continuous monitoring in the future. Soils of Cajun Prairies were never studied in detail; our research project is now investigating the important products of plant decomposition on topsoil formation, as well as soil-nutrient distribution in remnant and restored prairies, before natural prairies completely disappear in southwestern Louisiana. We hypothesized that the accumulation of soil nutrients [phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sodium (Na), zinc (Zn), copper (Cu), iron (Fe), and manganese (Mn)] would be greater in remnant prairies than in restored prairies at different soil depths, because the old remnant prairies would have higher organic matter accumulation that would provide greater adsorption sites for soil nutrients than the young restored prairies. The objectives of this study were two: 1. To determine the distributions of soil nutrients (P, K, Ca, Mg, Na, Fe, Cu, Mn, and Zn) with soil depth (0-10, 10–20, 20-30 cm) under a canopy of switchgrass. 2. To determine which soil properties (pH, electrical conductivity [EC], organic matter [OM], and the proportions of sand, silt, and clay) correlate with soil-nutrient distribution.

MATERIALS AND METHODS

SITE DESCRIPTION

Two restored and two remnant Cajun Prairies in southwestern Louisiana were chosen for this study. The two restored prairies were the Cajun Prairie Restoration Project (CPRP) in the city of Eunice and the Louisiana State University at Eunice (LSUE) prairie on the school campus. Before the restoration, the CPRP was originally an area of grasses and weeds, whereas the LSUE prairie was originally a Saint Augustine grass (*Stenotaphrum secundatum*) lawn.

The restored prairies were established by LSUE faculty members Charles Allen, Malcolm Vidrine, and Bruno Borsari in 1988 for CPRP and 1989 for LSUE. During the first year of establishment, each prairie restoration site was mowed and herbicided with Roundup in January, burned, and plowed in February. In March, clumps of prairie sod from various remnant prairies were transplanted a meter apart into the prepared Cajun Prairie restoration plots. The winter sod was wet or damp during transplant. Since then, the growth of transplants has been dependent on rain. In the winter of the following year, seeds of different Cajun Prairie plants were broadcast by hand between transplants. Each restoration plot now contains more than 100 species of warm-season perennials (Vidrine et al. 1995). Both restored prairies have been managed by conducting annual winter burns. The 22-year-old CPRP has been burned in January without mowing, whereas the 21-year-old LSUE prairie has been burned and mowed every January (Jariel et al. 2006 and 2008, Jariel Jr. et al. 2010a and 2010b, Vidrine 2010, Vidrine et al. 1995).

The two remnant prairies were located near Frey and Estherwood. Both remnants are at least 100 years old (Goins and Caldwell 1995) and have been managed by local farmers by prescribed burning at infrequent intervals before 1990. Burning of Estherwood prairie has been discontinued since 1997. These remnants have had disturbances from agricultural and/or railroad machinery from time to time, and they have been highly susceptible to biocidal drift and fertilizer effluent during the agricultural season. But, in general, they have not received constant and specific management since 1870 (Lockett 1970).

The parent material of soil series in southwestern Louisiana is clayey alluvium, which was deposited by the Mississippi and Red Rivers during the Pleistocene (Clark et al. 1959, Murphy et al. 1986, NRCS 2008). Soils of all four prairies belong to the soil order Alfisols and exhibit low water permeability. The restored CPRP and LSUE prairies reside on a poorly drained Crowley soil series classified as fine, montmorillonitic, thermic Typic Albaqualfs with slopes of less than 1% (Murphy et al. 1986). The soils of these two restored prairies have dark grayish silt-loam surface (0-50 cm) texture and light grayish brown silty-clay-loam subsoil (50-150 cm).

The remnant Frey prairie lies on an imperfectly drained Acadia soil series classified as fine, montmorillonitic, thermic Aeric Ochraqualfs with 1% to 3% slopes (Clark et al. 1959). The texture of the dark yellowish brown surface (0-7 cm) layer of Frey prairie soils is silt loam, whereas the yellowish brown subsoil (7-16 cm) is silt clay. The remnant Estherwood prairie is situated on a deep, poorly drained Midland soil series classified as fine, montmorillonitic, thermic Chromic Vertic Epiaqualfs with 0% to 1% slopes (Clark et al. 1959). The textural class of the shallow, dark gray surface (0-2.4 cm) layer of Estherwood prairie's soil is silty clay loam, whereas the gray subsoil (2.4-7.0 cm) is silty clay.

The four prairies in southwestern Louisiana region receive an average of 125 cm of precipitation annually (Murphy et al. 1986). All prairies have been unfertilized. For our study, each prairie was the main plot and was divided into four blocks or replications using a randomized complete block design. The area of each block ranged from 12 m x 12 m to 14 m x 14 m.

SOIL SAMPLING

Soils were sampled from each block (replication) of each prairie on the week of June 15, 2008, selecting sites under the canopy of switchgrass. Using a 2.25 cm diameter sampling tube, six to nine soil sample cores at different depths (0-10, 10-20, and 20-30 cm) were taken 2.5 cm away from the crown of randomly chosen switchgrass plants. The soil sample cores in each soil depth were composited for each block (replication), placed in labeled plastic bags, and prepared for chemical analysis. The data presented in the figures are the means of four blocks (replications) in each soil depth for each prairie.

SOIL CHEMICAL AND PHYSICAL ANALYSIS

Soil samples were air-dried, pulverized with a soil grinder, and sieved through a 2 mm metal screen (Page 1982). Soil pH was determined by using a 1:1 (weight/volume) soil-water ratio (Eckert 1988), whereas soil EC was measured by using a 1:2 (weight/volume) soil-water ratio (Dahnke and Whitney 1988). Soil organic matter was determined by the loss-of-weight-on-ignition method using a muffle furnace at 360°C for two hours (Schulte 1988, Storer 1984). Exchangeable Ca, K, Mg, and Na were extracted from the soil with neutral 1 N ammonium acetate ($NH_4CH_3CO_2$) and quantified by atomic absorption spectrometry (Thomas 1982). Extractable phosphorus was obtained by desorption with Bray No. 1 hydrochloric-ammonium fluoride solution and quantified colorimetrically by developing a blue ammonium molybdenum phosphate complex (Watanabe and Olsen 1965). Available Zn, Fe, Mn, and Cu were extracted with diethylene triamine pentaacetic acid (DTPA) and determined by atomic absorption spectrometry (Lindsay and Norvell 1978). The proportions of sand, silt, and clay were determined by the hydrometer method (Bouyoucos 1962).

STATISTICAL ANALYSIS OF DATA

Probability (*P*) values of treatment effects (Table 1) on soil variables were analyzed statistically with SAS (SAS Institute 2003) by following the General Linear Model (GLM) procedure based on a Randomized Complete Block Design (RCBD) with split-plot arrangement. Each prairie had four blocks (B) or replications. Prairie (P) locations were the main plots, which were split into sampling depths (D) as subplots. The effects of P, D, and P x D interaction on soil variables were determined. CONTRAST statement was used to test the significant difference (at 1% or 5% level) of variables between groups of prairies.

Significant differences among treatment means were tested with the LSMEANS (Least Square Means) statement and the STDERR PDIFF option. Significant correlations among soil variables were tested by generating Pearson's correlation coefficients (r) (Table 2).

RESULTS AND DISCUSSION

VERTICAL DISTRIBUTION OF MACRONUTRIENTS

Group comparison of prairies showed that soil pH, electrical conductivity, extractable P, organic matter, and exchangeable Ca, Na, and K were not significantly different between these restored and remnant prairies (Table 1). These results were the opposite of what we expected. Because the population of switchgrass plants in remnant prairies was about a century older (Goins and Caldwell 1995) than those in restored prairies, we suspected that the soils under the canopies of older remnant-prairie plants would have higher expression of these soil properties compared to the soils beneath the canopies of younger restored-prairie plants. In rangeland, this prediction was found to be valid when tested on soils under the canopy of western juniper (*Juniperus occidentalis*), where soil nutrients, pH, and organic matter were greater under mature trees than under juvenile trees within one site (Doescher et al. 1987).

Table 1. Probability (*P*)[1] values of chemical and physical properties of soils in response to Prairie (P), Soil Depths (D), and P × D Interactions. P includes four Cajun Prairies (restored LSUE and CPRP prairies, and remnant Estherwood and Frey prairies). D includes 0-10, 10-20, 20-30 cm soil depths. B includes 4 blocks or replications. The B × P mainplot error term was used to test the significant difference among B or among P. The B × P × D error term was used to test the significant difference among D and the significance of the P × D interactions. CONTRAST statement was used to test the significant difference between groups of prairies.

SOURCE OF VARIATION	CHEMICAL PROPERTIES												PHYSICAL PROPERTIES		
	PH	EC, DS/M	OM, %	P, MG/KG	CA, MG/KG	MG, MG/KG	NA, MG/KG	K, MG/KG	FE, UG/G	CU, UG/G	MN, UG/G	ZN, UG/G	SAND, %	SILT, %	CLAY, %
BLOCK (B)	0.37 ns	0.64 ns	0.89 ns	0.28 ns	0.98 ns	0.90 ns	0.65 ns	0.88 ns	0.16 ns	0.52 ns	0.76 ns	0.20 ns	0.06 ns	0.28 ns	0.32 ns
PRAIRIE (P)	0.001**	0.16 ns	0.40 ns	0.14 ns	0.34 ns	0.37 ns	0.10 ns	0.15 ns	0.002**	0.007**	0.54 ns	0.002**	0.002**	0.001**	0.001**
DEPTH (D)	0.04*	0.33 ns	0.001**	0.28 ns	0.15 ns	0.17 ns	0.001**	0.001**	0.003**	0.001**	0.10 ns	0.001**	0.001**	0.001**	0.001**
P X D INTERACTION	0.04*	0.32 ns	0.002**	0.15 ns	0.10 ns	0.26 ns	0.001**	0.003**	0.006**	0.03*	0.02*	0.001**	0.001**	0.001**	0.001**
CONTRAST															
RESTORED VS. REMNANT	0.52 ns	0.34 ns	0.13 ns	0.06 ns	0.44 ns	0.005**	0.07 ns	0.48 ns	<.001**	0.006**	0.03*	0.03*	0.04*	<.001**	<.001**
LSUE VS. 3 PRAIRIES	<.001**	<.001**	0.79 ns	0.04*	0.04*	0.78 ns	0.59 ns	0.46 ns	0.24 ns	0.48 ns	0.38 ns	0.15 ns	0.95 ns	<.001**	0.10 ns
CPRP VS. 3 PRAIRIES	<.001**	0.005**	0.04*	0.82 ns	0.005**	0.003**	0.01*	0.12 ns	<.001**	<.001**	0.10 ns	0.28 ns	0.02*	0.04*	0.007**
ESTHERWOOD VS. 3 PRAIRIES	0.13 ns	0.90 ns	0.89 ns	0.65 ns	0.57 ns	0.29 ns	<.001**	0.02*	0.003**	<.001**	0.13 ns	0.38 ns	0.51 ns	0.03*	0.81 ns
FREY VS. 3 PRAIRIES	0.03*	0.22 ns	0.11 ns	0.08 ns	0.15 ns	0.02*	0.07 ns	0.001**	<.001**	<.001**	0.32 ns	0.001**	0.07 ns	<.001**	<.001**

[1] The Probability (*P*) values are not significant (ns) at *P* > 0.05, significant (*) at P < 0.05, and highly significant (**) at P < 0.01.

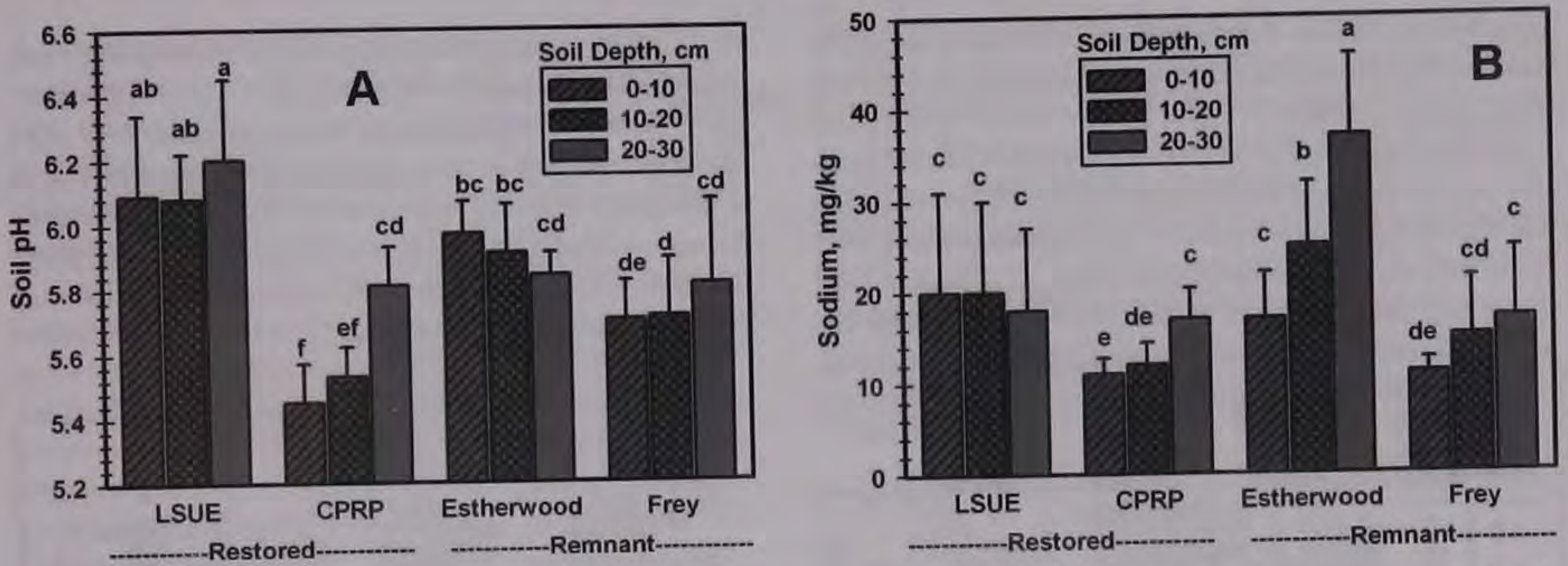

Figure 1. Soil pH (A), and exchangeable sodium (B) at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different ($P > 0.05$). Vertical lines on the bars are standard errors nutrients and micronutrients.

Soil pH levels among the individual prairies were significantly different (Table 1). The restored LSUE prairie, which received the most-intensive management practices of annual burning and mowing, had the highest pH value of 6.1. Pooled over prairies, soil pH increased with depth ($P < 0.05$) from topsoil to subsoils (Table 1, Figure 1a) and was positively correlated with exchangeable calcium ($r = 0.55^{**}$), magnesium ($r = 0.43^{**}$), and sodium ($r = 0.30^{*}$) (Table 2). This explains the direct dependency of soil pH on the amounts of exchangeable basic cations (Brady and Weil 2008, Gardiner and Miller 2008). Among these cations, it was only the sodium ($P < 0.01$) together with soil pH that generally increased significantly with soil depth (Table 1, Figures 1a and 1b). Greater distribution of Na at the lower depths suggests that its net downward mobility is greater than that of both Ca and Mg. Despite the differential influences of basic cations (Ca, Mg, Na, and K) on soil pH, their ability to generate electric current in soil solutions provided a positive correlation with soil electrical conductivity (Table 2).

Organic matter significantly decreased with depth ($P < 0.01$) (Table 1, Figure 2a) and was positively correlated with exchangeable calcium ($r = 0.73^{**}$), magnesium ($r = 0.59^{**}$), and potassium ($r = 0.85^{**}$) (Table 2). However, it was only potassium ($P < 0.01$) that coincided with organic matter by decreasing significantly with depth (Table 1, Figures 2a and 2b), indicating that organic matter in the topsoil (0-10 cm) held K more tightly than it did Ca and Mg in these unfertilized prairies. Similar results were obtained in both unfertilized and fertilized apple orchards, where concentrations of K, Ca, and Mg were greater in the topsoil than in the subsoils, but Ca and Mg leached more rapidly than K (Nachtigall et al. 2007, Neilsen and Stevenson 1982). In contrast, application of lime and potassium fertilizers in drained, oligotrophic or-

Table 2. Pearson's correlation coefficients (r)[1] of soil chemical and physical properties against the soil macronutrients and micronutrients.

CHEMICAL PROPERTIES	MACRONUTRIENTS, MG/KG					MICRONUTRIENTS, UG/G			
	P	Ca	Mg	Na	K	Fe	Cu	Mn	Zn
PH	ns	0.55^{**}	0.43^{**}	0.30^{*}	ns	ns	ns	ns	ns
EC, DS/M	ns	0.77^{**}	0.65^{**}	0.47^{**}	0.49^{**}	ns	ns	ns	0.31^{**}
OM, %	ns	0.73^{**}	0.59^{**}	ns	0.85^{**}	ns	ns	ns	0.45^{**}
P, MG/KG	1.00^{**}	ns	ns	ns	ns	0.35^{*}	ns	ns	0.39^{**}
CA, MG/KG	ns	1.00^{**}	0.84^{**}	ns	0.75^{**}	-0.30^{*}	-0.31^{*}	-0.29^{*}	ns
MG, MG/KG	ns	0.84^{**}	1.00^{**}	0.44^{**}	0.62^{**}	-0.46^{**}	-0.43^{**}	-0.52^{**}	ns
NA, MG/KG	ns	ns	0.44^{**}	1.00^{**}	ns	ns	ns	-0.50^{**}	ns
K, MG/KG	ns	0.75^{**}	0.62^{**}	ns	1.00^{**}	ns	-0.36^{**}	ns	ns
FE, UG/G	0.35^{*}	-0.30^{*}	-0.46^{**}	ns	ns	1.00^{**}	0.81^{**}	ns	0.51^{**}
CU, UG/G	ns	-0.31^{*}	-0.43^{**}	ns	-0.36^{**}	0.81^{**}	1.00^{**}	ns	0.54^{**}
MN, UG/G	ns	-0.29^{*}	-0.52^{**}	-0.50^{**}	ns	ns	ns	1.00^{**}	ns
ZN, UG/G	0.39^{**}	ns	ns	ns	ns	0.51^{**}	0.54^{**}	ns	1.00^{**}
PHYSICAL PROPERTIES									
SAND, %	0.35^{*}	ns	ns	ns	ns	0.53^{**}	0.39^{**}	ns	0.46^{**}
SILT, %	0.34^{*}	ns	ns	ns	ns	0.69^{**}	0.67^{**}	ns	0.67^{**}
CLAY, %	-0.42^{**}	ns	0.32^{*}	ns	ns	-0.73^{**}	-0.73^{**}	ns	-0.66^{**}

[1] The correlation coefficient (r) between two variables is not significant (ns) at $P > 0.05$, significant (*) at $P < 0.05$, and highly significant (**) at $P < 0.01$

ganic soils mobilized exchangeable K into deeper soil layers with increased concentrations being measured to a 60 cm depth (Braekke 1999).

Group comparison of prairies showed that percent sand and percent silt were significantly higher, but percent clay was significantly lower in these two restored prairies than in the two remnant counterparts (Table 1). Across texture types, each prairie had the highest percentages of sand, intermediate levels of silt, and the lowest percentages of clay (Figures 3a, 3b, and 3c), with a textural class of sandy loam at 0-10 cm depth, sandy loam or sandy clay loam at 10-20 cm depth, and sandy clay loam or clay loam at 20-30 cm depth.

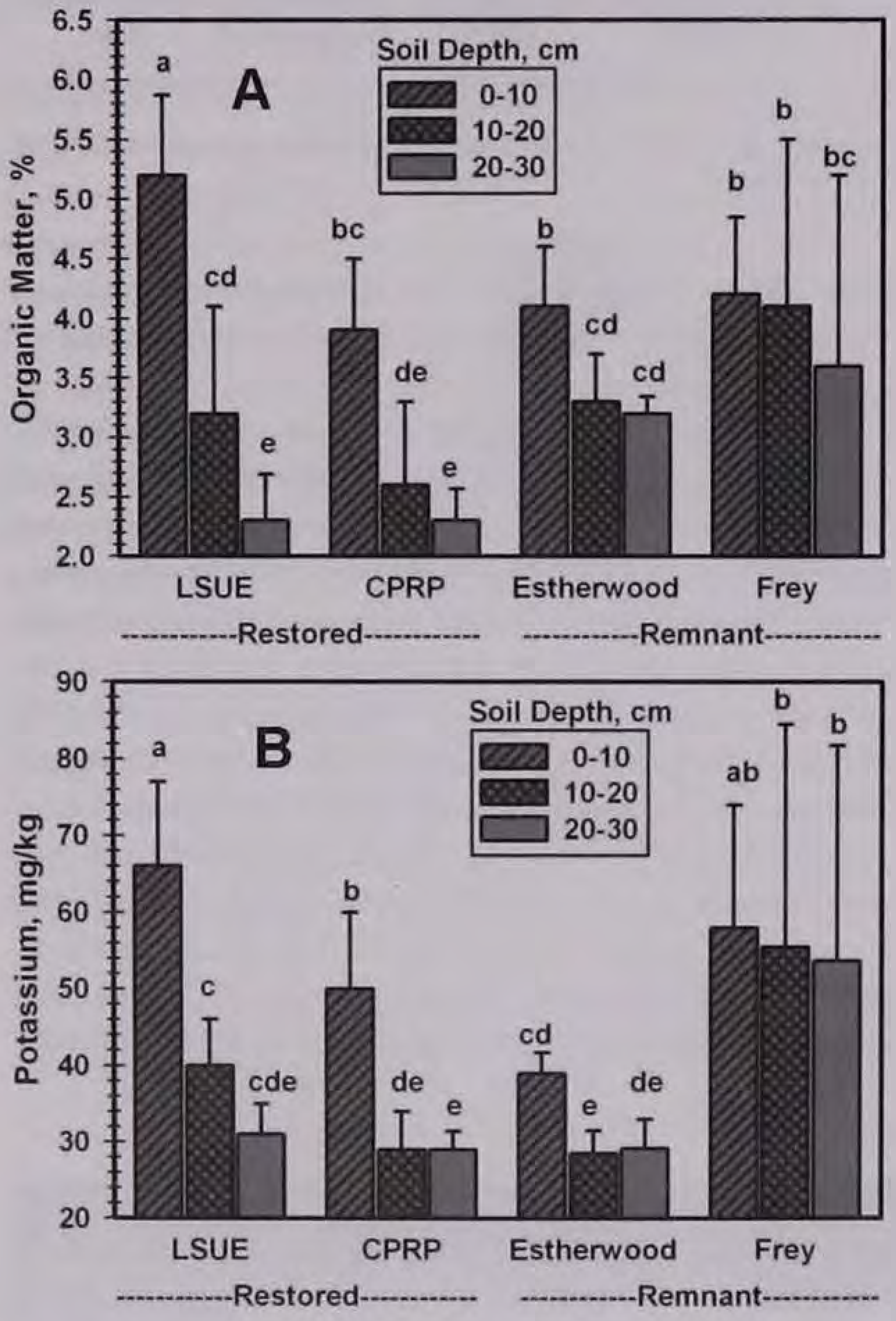

Figure 2. Soil organic matter (A) and exchangeable potassium (B) at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different ($P > 0.05$). Vertical lines on the bars are standard errors.

The distribution of sand and silt with depth varied among prairies. However, clay content significantly increased with depth from topsoil (0-10 cm) to subsoils (10-20 and 20-30 cm) (Table 1, Figure 3c), indicating that clay illuviation occurred in all prairies. Remnant Frey prairie had significantly lower percent silt and higher percent clay ($P < 0.01$) when compared to the other three prairies (Table 1, Figures 3b and 3c). Among the soil particles, clay content was negatively correlated with extractable phosphorus ($r = -0.42^{**}$), but positively correlated with exchangeable magnesium ($r = 0.32^{*}$) (Table 2). The data suggest that soil phosphorus, measured as phosphate anions, was less mobile than was magnesium, due to complexation and precipitation reactions of phosphates with calcium, aluminum, and iron cations forming insoluble compounds in the topsoil (Braekke 1999, Crozier et

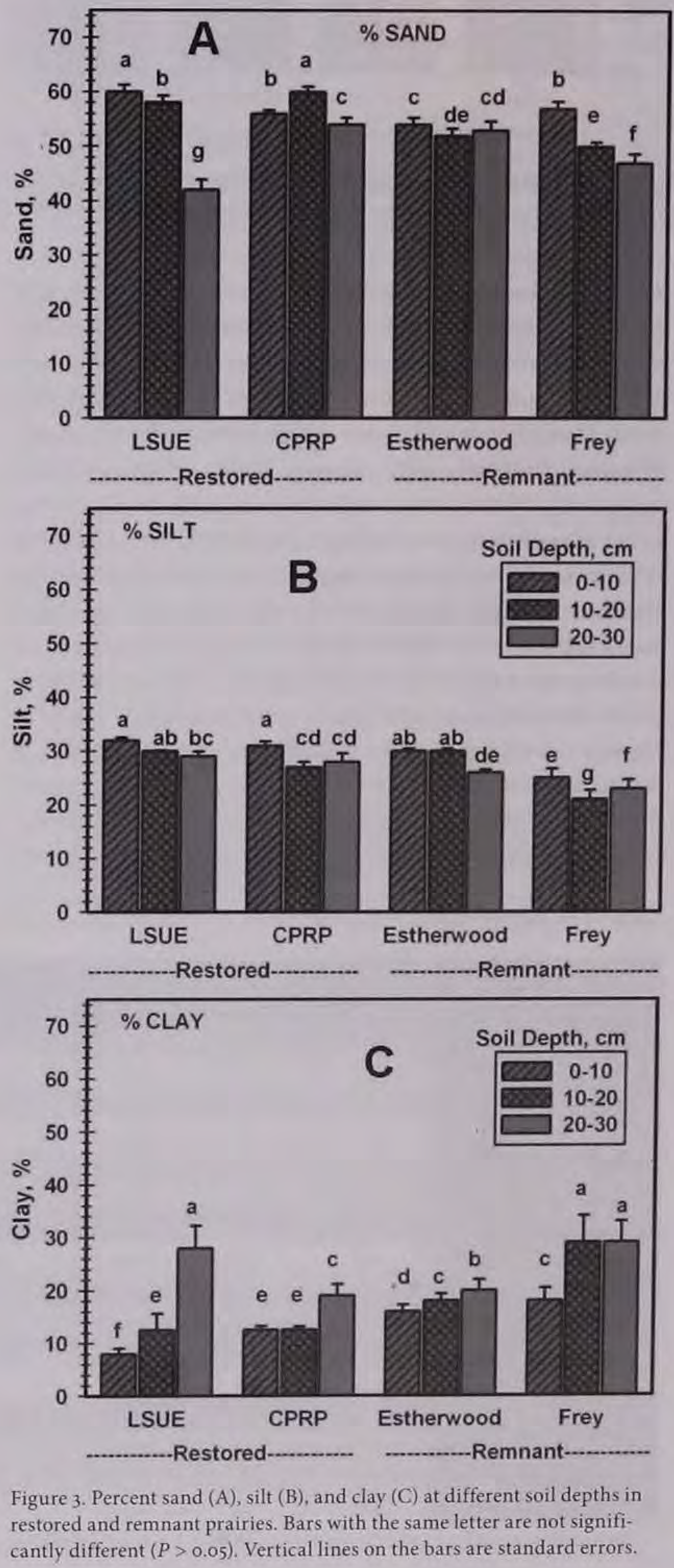

Figure 3. Percent sand (A), silt (B), and clay (C) at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different ($P > 0.05$). Vertical lines on the bars are standard errors.

al. 1999). This explains the immobility of phosphate due to its weaker attraction to the negatively charged sites of clay particles than would be the case for magnesium (Brady and Weil 2008, Gardiner and Miller 2008), and confirms the mobility of magnesium, a cation, which moved together with clay to the deeper soil layers (Jobbagy and Jackson 2001).

The immobility of P and K in Cajun Prairie soils resembled observations from cultivated agricultural fields of other studies. Crozier et al. (1999) and Follett and Peterson (1988) determined that under tillage and no-till management, concentrations of P and K decreased with soil depth whether or not lime and fertilizers were applied. However, the adoption of no-till, compared to tillage, maintained the fertility status of the topsoil closer to that of native prairie soils. Under no-till, the nutrient gradient among soil depths was further enhanced by the surface application of fertilizers. For this reason, besides P and K, the concentrations of Ca, Mg, Mn, Zn, and Cu were also higher in no-till topsoil than in the subsoil (Crozier et al. 1999, Follett and Peterson 1988). Similarly, in our study, the lack of plowing in unfertilized Cajun Prairie soils also favored higher concentrations of micronutrients, such as Fe, Cu, and Zn, in the topsoil (0-10 cm) than in the subsoils (10-20 and 20-30 cm).

VERTICAL DISTRIBUTION OF MICRONUTRIENTS

Group comparison of prairies showed that soil-available Fe, Cu, and Zn, and the percent sand and silt were significantly higher, whereas percent clay was significantly lower in our two restored prairies than in the two remnants (Table 1, Figure 3, Figure 4). Pooled over prairies, Fe, Cu, and Zn concentrations decreased significantly with depth (Table1, Figures 4a, 4b, and 4c). Among the prairies, the Frey remnant had the lowest concentrations of Fe, Cu, and Zn, which were significantly different ($P < 0.01$) from the other three prairies.

Distribution of soil Fe, Cu, and Zn with depth was correlated by soil organic matter or soil particles or both. In our study, soil organic matter correlated significantly with soil-available Zn ($r = 0.45^{**}$) (Table 2, Figure 2a, Figure 4c), and a similar relationship was observed by another study examining a tropical soil after long-term disposal of sewage sludge (Udom et al. 2004). Complexation of available Zn with organic matter probably occurred more readily in the topsoils, where Zn and OM concentrations were higher than those in the subsoils. In other studies, the addition of organic matter to the soil provided greater retention of Cu and Zn in the soil surface. For example, long-term applications of poultry litter or cow manure showed that there was no or little evidence of Cu and Zn translocation past the 0-15 cm plow layer (Brock et al. 2005).

Silt particles decreased and clay particles increased significantly ($P < 0.01$) from topsoil to subsoil (Table 1, Figures 3b and 3c). Based on the significant correlations of percent silt with Fe ($r = 0.69^{**}$), Cu ($r = 0.67^{**}$), and Zn ($r = 0.67^{**}$), silt particles may also have contributed to the retention of these micronutrients more in the topsoil than in the subsoil (Table 2, Figure 3, Figure 4). However, clay particles may provide less contribution to the topsoil accumulation of these micronutrients, as indicated by the negative correlations of clay with Fe ($r = -0.73^{**}$), Cu ($r = -0.73^{**}$) and Zn ($r = -0.66^{**}$) (Table 2). These data contradicted our hypothesis that Fe, Cu, and Zn would leach together with clay to lower depths.

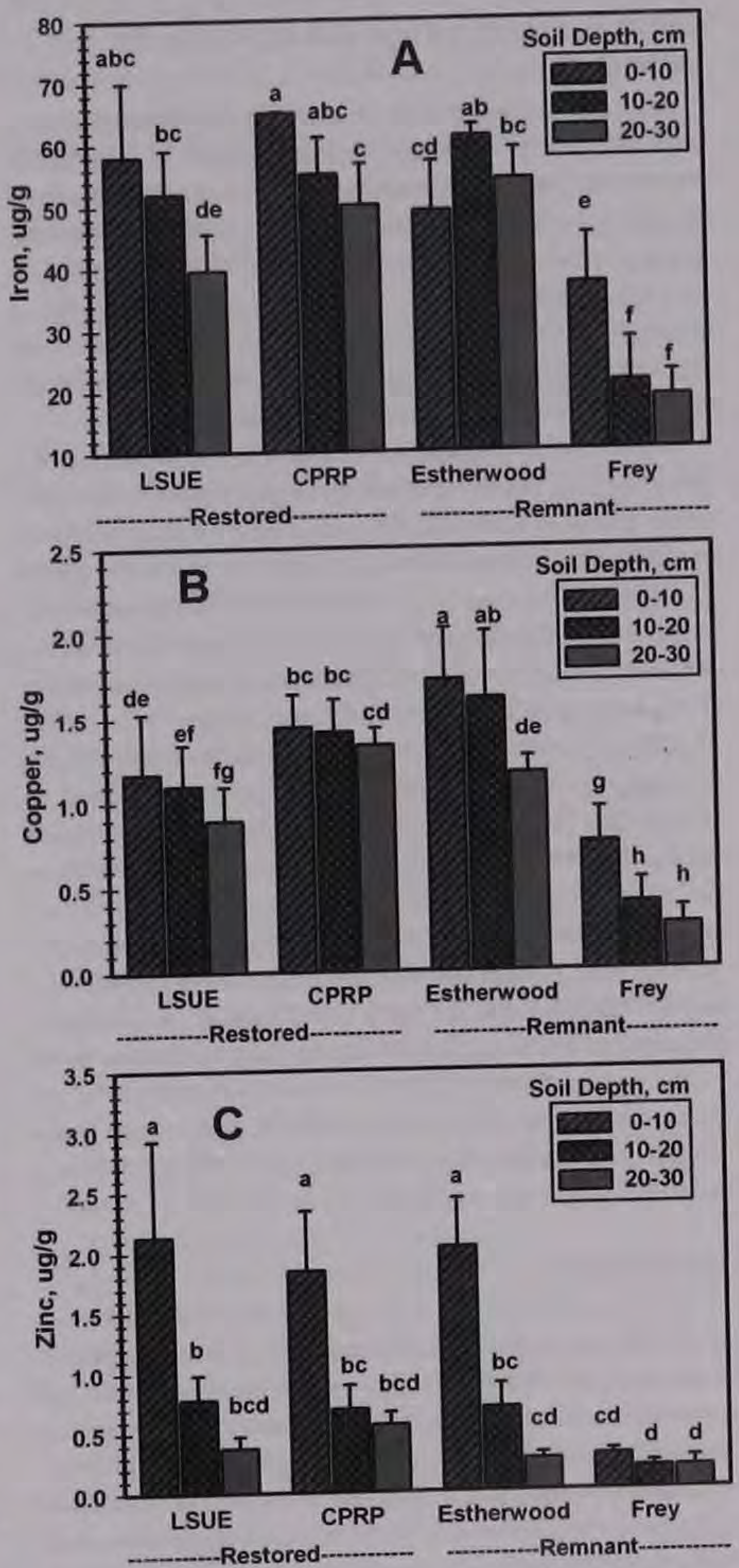

Figure 4. Soil-available (DTPA-extractable) iron (A), copper (B), and zinc (C) at different soil depths in restored and remnant prairies. Bars with the same letter are not significantly different ($P > 0.05$). Vertical lines on the bars are standard errors.

The distribution of Fe with depth was probably also influenced by soil pH, based on our observations of the increasing soil pH (Figure 1a) and decreasing Fe concentrations (Figure 4a) at increasing soil depths (0-10, 10-20, 10-30 cm). However, this relationship was not reflected by a significant negative correlation between soil pH and Fe, although other studies have found this inverse relationship to be significant (Braekke 1999, Jariel et al. 1991 and 2002).

VERTICAL ACCUMULATION AND DEPLETION OF NUTRIENTS

Root distribution in soils influences nutrient stratification with soil depth. Although the root biomass of switchgrass was not measured in this study, several studies (Dahlman and Kucera 1965, Ma et al. 2000, Waldron and Dakessian 1982, Xu et al. 2010) consistently reported that roots of switchgrass and other native prairie grasses were mainly distributed in the topsoil, but to varying depths (0-15, 0-20, 0-30, or 0-43 cm). Root biomass decreased as the roots penetrated the lower depths, which can extend down to 330 cm.

In plant cycling, nutrient extraction by roots (nutrient depletion) in the subsoil and mineralization by litter decomposition (nutrient accumulation) subject the vertical soil layers to differential accumulation and depletion across the profile (Jobbagy and Jackson 2001). In the topsoil, in spite of the predominant root biomass that extracts nutrients, the rate of nutrient accumulation is faster than that of nutrient depletion. Thus, concentrations of OM and K were greater in the topsoil than those in the subsoil in our study (Figure 2), confirming the hypothesis of Jobbagy and Jackson (2001) that topsoil accumulation favored the most limiting nutrients for plants, such as P and K (those required in high amounts in relation to soil supply). However, we found that P concentrations seemed to be higher in the topsoil based only on their significant correlation with percent silt, which significantly decreased with depth (Table 1, Table 2, Figure 3b). Therefore, topsoil accumulation of P was not mainly due to plant cycling but could be due to a combination of factors, such as complexation and precipitation reactions forming insoluble compounds in the topsoil (Braekke 1999, Crozier et al. 1999) and inability to leach with clay particles to the deeper soil layers.

CONCLUSION

The lack of significant differences in macronutrient levels (P, Ca, Na, and K) between restored and remnant prairies suggests that the switchgrass plants in both prairies utilized and recycled these nutrients equally, regardless of prairie age and maintenance practices. However, the significantly higher micronutrient levels (Fe, Cu, and Zn) in restored prairies likely reflected greater soil-nutrient accumulation there rather than depletion in remnant prairies. This may be a product of the annual burns employed in restored (but not in remnant) prairies, which facilitate the deposition of these micronutrients into the soil.

Concentrations of soil-exchangeable K in the topsoil were positively correlated with OM. However, there is no single factor that could explain the retention of soil-available Fe, Cu, and Zn in the topsoil. Nevertheless, higher concentrations of K, Fe, Cu, and Zn in the topsoil than in the subsoils suggest that replenishment of these elements by nutrient cycling is faster than leaching; these elements are relatively immobile and may have greater chemical attraction to silt and organic matter in the topsoil.

ACKNOWLEDGMENTS

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PROPAGATION AND AGRONOMIC SEED INCREASE OF NATIVE SEDGES (*CAREX*)

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Abstract: Plant materials of species native to wet prairie and sedge meadows in Iowa, particularly sedge (*Carex*) species, are in demand for restoration. Commercially available seed in quantity would facilitate restoration, potentially increasing the efficiency and scale of restorations. An agronomic technique using plastic film (mulch) and drip irrigation (t-tape), called plasticulture, was used for establishment and increase of several native *Carex* species. Seed collections from remnant populations were propagated as greenhouse plugs and transplanted into plastic film-covered beds and irrigated with drip irrigation. Species in trial production beds include yellow fox sedge (*Carex annectens*), Bebb's sedge (*C. bebbii*), plains oval sedge (*C. brevior*), Buxbaum's C. (*buxbaumii*), crested sedge (*C. cristatella*), troublesome sedge C. (*molesta*), woolly sedge (*C. pellita*), running marsh sedge (*C. sartwellii*), broom sedge (*C. tribuloides*), and fox sedge (*C. vulpinoidea*). Initial results using these techniques demonstrate a small harvestable seed crop is possible the first growing season for some species. Second-year estimated bulk seed yields ranged from a low of 33 kg/ha (29 lbs/ac) for *C. cristatella* to a high of 498 kg/ha (429 lbs/ac) for *C. vulpinoidea*, while remaining species ranged from 194 to 436kg/ha (168 to 376 lbs/ac). All species were cleaned to purities exceeding 90%. Potential market value, in terms of dollar value of pure live seed per unit production area ($PLS/unit area), was greatest for *C. tribuloides*, *C. annectens*, and *C. vulpinoidea*, at $43.27/m2 (3.89/ft2), $20.93/m2 (1.88/ft2), and $11.18/m2 (1.01/ft2), respectively.

Key Words: *Carex* seed production, drip irrigation, perigynium, plasticulture, stratification, transplant

INTRODUCTION

Carex species occupy an array of habitats from deepwater marshes to open woodlands, to dry, gravelly blufftop and ridge prairies. They are cool-season grasslike (graminoid) plants, and may comprise as much as a quarter of the aboveground biomass in tallgrass prairies (Coppedge et al. 1998). *Carex* species are the dominant vegetation in sedge meadow communities, yet are seldom included or are underutilized in seed mixes for prairie and wetland restorations. One reason for this is that commercial sources and quantities of *Carex* seed are limited. Challenges to *Carex* production include 1) accurate species identification, 2) adequate germination of wildland seed collections, and 3) developing efficient methods of commercial production (Houseal and Smith 2010).

Currently, 120 species of *Carex* are known to occur in Iowa (Norris and Zager 2008). Approximately 53 species might be considered wet-mesic to dry-mesic prairie species, with several additional species predominately associated with wetlands. The main objectives of this project were to 1) evaluate plasticulture techniques for establishment of seedling plugs, and 2) develop seed production plots of several native *Carex* species suitable for prairie and wetland restoration.

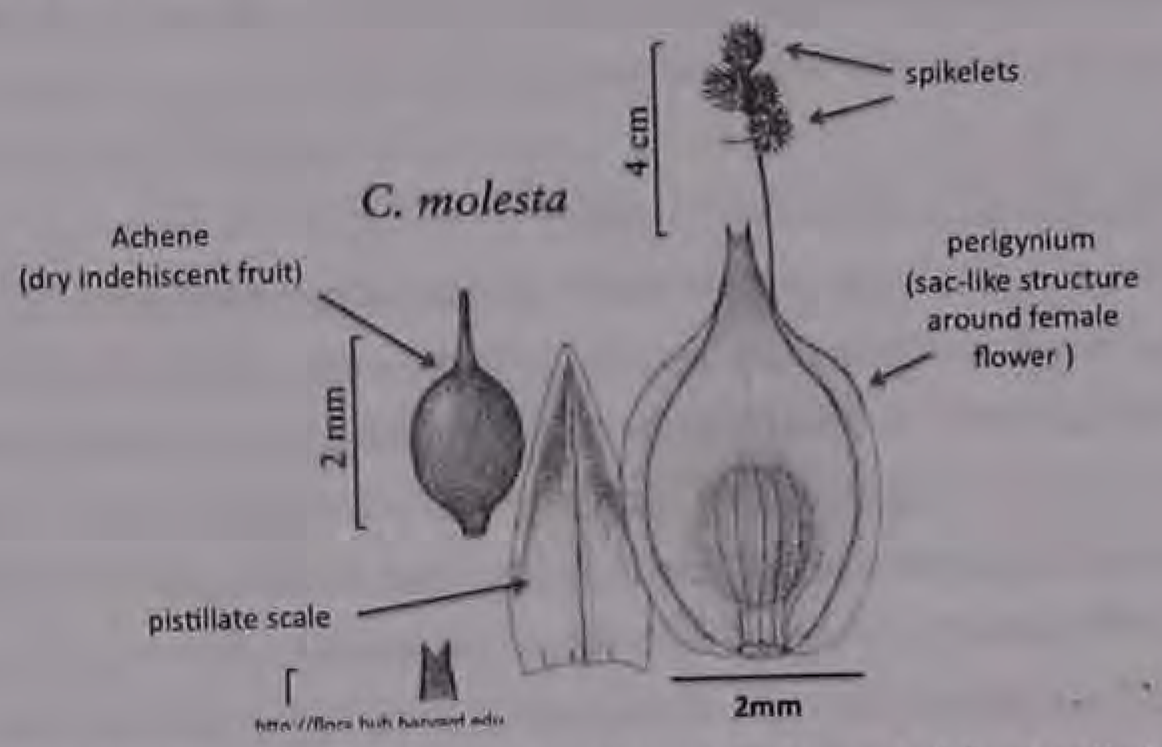

Figure 1. In *Carex*, the seed is enclosed in a single-seeded fruit called an achene, which is enclosed in a sac-like structure called a perigynium.

MATERIALS AND METHODS

COLLECTION OF SEED

In *Carex*, the seed is in a single-seeded fruit called an achene, which is enclosed by a sac-like membranous structure called a perigynium. These perigynia are arranged in spikes of various configurations, depending on species (Figure 1).

Table 1. *Carex* species transplanted into plasticulture beds. For a description of wetland indicator regions and categories see USDA-NRCS PLANTS database http://plants.usda.gov/wetinfo.html#regions. Region 3 is the North Central Region, USA (IA, IL, IN, MI, MN, MO, WI).

SCIENTIFIC NAME	COMMON NAME	WETLAND INDICATOR STATUS REGION 3
Carex annectens (E.P. Bicknell) E.P. Bicknell	Yellow fox sedge	FACW
Carex bebbii Olney ex Fernald	Bebb's sedge	OBL
Carex bicknellii Britton	Bicknell's or Prairie sedge	FAC-
Carex brevior (Dewey) Mackenzie	Plains oval sedge	FAC
Carex buxbaumii Wahlenb.	Buxbaum's sedge	OBL
Carex cristatella Britton	Crested sedge	FACW+
Carex molesta Mackenzie ex Bright	Troublesome sedge	FAC
Carex pellita Muhl. ex Willd	Woolly sedge	OBL
Carex sartwellii Dewey	Sartwell's or Running marsh sedge	OBL
Carex tribuloides Wahlenb.	Blunt broom sedge	FACW+
Carex vulpinoidea Michx.	Common fox sedge	OBL

Seeds (here after referred to as perigynia) were thus collected by stripping perigynia (or entire spikes of perigynia) from individual plants from populations of selected species (Table 1). Collections were made predominately in late June through mid July of 2008 from remnant prairies in east central, north central, and northeastern Iowa. Specimen plants were collected for each species at each site, and perigynia were randomly collected from several individual clumps from throughout the population. Species field identification was checked with a hand lens, and later confirmed in the lab under a dissecting microscope. Because the ultimate goal was to propagate seedlings, all species were collected when perigynia were mature. This facilitated identification since sedges are most easily identified when fruits are ripe (Hipp 2008). Hybrid or questionable forms were avoided. Perigynia were air-dried (ambient, forced-air) for two weeks, and hand-screened to remove sticks, leaves, and larger particles, and then aspirated with a South Dakota seed blower to remove pistillate scales and perigynia with aborted or empty (nonviable) achenes.

GREENHOUSE PROPAGATION

Three factors generally favor germination in *Carex*. These are 1) cold moist stratification, 2) alternating soil temperature (50°F daytime/70°F nighttime) after sowing, and 3) perigynium exposure to light after sowing (Schütz and Rave 1999, Kettenring et al. 2006, Kettenring and Galatowitsch 2007). Achenes were not removed from perigynia, and were cold-stratified intact in moist, sterile sand for 4 weeks at 40°F (4°C). Since light generally enhances sedge germination (Schütz and Rave 1999, Kettenring et al. 2006), perigynia were sown very shallowly at 3mm (1/8 in) depth into potting medium in 70-count trays, 7.62 cm (3 in) deep in February and March 2009. One or two trays (70-140 cells) were seeded for each population and watered with an overhead mist system to avoid seed displacement until germination occurred. Germination began within 2 weeks after sowing, and occurred during a protracted period over the next several weeks. Seedling growth in *Carex* species is generally rapid after germination, and seedlings were grown for approximately two months with natural light in a greenhouse with an ambient temperature of 78°F (25.5°C) (Figure 2).

Figure 2. Crested sedge (*C. cristatella*) seedlings about 3 weeks after germination, growing in Ray Leach fir-cell Cone-tainers (Stuewe and Sons, Tangent, OR; http://www.stuewe.com/).

PLASTICTULTURE PRODUCTION BEDS

Seed production beds were set up using a system known as plasticulture, adapted for *Carex* production (Houseal 2010). Plasticulture has been used in the vegetable industry since the early 1960s (Sweat 2007). Black plastic film (mulch, Pliant USA, 1.0 mil) covers the bed to provide weed control, and drip irrigation tape (t-tape, 10 mil, 30 cm drip spacing) is installed beneath the film to provide supplemental moisture (and fertilizer, if needed). The plastic mulch enhances soil warming and moisture and nutrient retention, and effectively lengthens the growing season, promotes establishment, and increases plant size and potential yields (Lamont 2004). Plasticulture beds were formed at ground level (as opposed to raised beds) to facilitate combine harvesting. Total area for seed production beds of each species is shown in Table 2.

Table 2. Harvest dates, plot size, bulk yield and estimated seed yield/unit area from selected *Carex* species, second full growing season after transplanting.

SPECIES	HARVEST DATE	PLOT SIZE M^2 (FT^2)	BULK YIELD Kg (LBS)	YIELD/UNIT AREA KG/Ha(LBS/AC)
C. annectens	1 July	56 (624)	2.2 (4.7)	383 (330)
C. bebbii	1 July	42 (464)	1.1 (2.5)	267 (230)
C. bicknellii	23 June	9 (104)	0.2 (0.4)	194 (168)
C. brevior	23 June	65 (726)	2.1 (4.5)	315 (272)
C. cristatella	20 July	71 (792)	0.2 (0.5)	33 (29)
C. molesta	29 June	36 (400)	1.1 (2.5)	310 (267)
C. tribuloides	24 June	76 (844)	3.3 (7.3)	436 (376)
C. vulpinoidea	16 July	59 (660)	2.9 (6.5)	498 (429)

Figure 3. Transplanting two-month-old *Carex* seedling plugs (inset) into plasticulture beds in early spring 2009.

Seed production beds were prepared by cultivation and rototillage so that no coarse stubble or stalks were present that could puncture plastic mulch film during installation. Two-month-old seedlings were transplanted at 20-cm (8-in) intervals into plastic film mulch in late April and early May 2009 (Figure 3). Adequate soil moisture and reliable rains at this time of year and cool temperatures favor rapid establishment and growth of sedges. The last frost-free date for the area is approximately May 15. Sedges are generally frost tolerant, but transplanting was timed when the weather forecast predicted mild nighttime temperatures over the next several days. Weeds were controlled by hand pulling from the around base of transplanted seedlings within beds. The area between the beds was cultivated with a small, tractor-mounted 1.52 m (5 ft.) rototiller, so 1.83 m (6 ft.) spacing was left between the plastic mulch-covered beds.

Figure 4. Combining lodged *Carex* seed heads with Hege 125B plot combine and resulting bagged material (inset).

HARVESTING SEED

A Hege 125B 1978 model plot combine was used to harvest selected species on the dates indicated in Table 2 (Figure 4). Harvested material was hand screened through 6.5 mm (½ in) and 13 mm (¼ in) hardware cloth to remove large particles, stems, and leaves and make the material more flowable. Material was then cleaned with a Westrup 3-screen air-screen cleaner (LA-LS Westrup Laboratory air screen cleaner) and submitted for seed test (Figure 5).

Figure 5. Cleaning perigynia (inset) with Westrup LA-LS laboratory airscreen cleaner.

RESULTS AND DISCUSSION

Survival of transplanted *Carex* plugs was essentially 100 percent, and all plots were well established by mid-summer of the first growing season. All species established readily and grew rapidly throughout the spring, mostly remaining vegetative the first growing season with some exceptions. The vigorous rhizomatous spread of some species (e.g., *Carex pellita*, *C. sartwellii*, and *C. buxbaumii*) required opening up plastic mulch around the base of plants to accommodate new tiller growth. Plastic was removed from the *C. pellita* plot later in the season to accommodate vigorous tillering; drip tape was left in place for irrigation. Plastic mulch around clump-forming (caespitose) species was left intact.

Flowering and seed set were delayed the first season, if it occurred at all, relative to mature plants (second-year and beyond). This is normal for first-year transplants of many perennial species. Species that set seed the first growing season in the plasticulture beds included *Carex bebbii*, *C. tribuloides*, *C. brevior*, *C. molesta*, *C. annectens*, and *C. vulpinoidea*. The obligate wetland species *C. bebbii* produced enough seeds the first season to be combine harvested. This species produced an estimated 82.3 kg/ha (73.5 lbs/ac) of seed compared to 267 kg/ha (230 lbs/ac) in year two. All species flowered and set fruit in year two. The highly rhizomatous species *Carex pellita* and *C. sartwellii*, and to some extent *C. buxbaumii*, had very limited flowering and seed production, and were simply hand harvested (data not shown). It was difficult to find quantities of seed of these species in the original native stands, as well, presumably because their primary mode of reproduction is vegetative (clonal spread) and not via seed. However, *C. sartwellii* growing in a pot in the greenhouse flowered prolifically, suggesting that resource limitation (stress) may enhance flowering, at least in this species.

Lodging of seed heads of plants growing along the edge of the beds onto the ground was an issue with several species of *Carex*, notably *C. bebbii*, *C. brevior*, *C. molesta*, and *C. tribuloides*, making combining difficult and resulting in lost seed. An attempt was made to lift lodged plants with a pitchfork, back toward the center of bed onto supporting vegetation, a week or so before seed shatter to facilitate combining, which mitigated seed loss to some extent. Lodging was not an issue with *C. annectens*, *C. bicknellii*, and *C. vulpinoidea* (Figure 6). Our experience has been that over-growing and lodging of shoots is not uncommon in native perennials when they are grown in a production setting, essentially released from competition for nutrients and light compared to the highly competitive environment they are adapted to in complex native plant

Figure 6. *Carex* production beds in late June of second growing season, established from plugs transplanted previous spring. Yellow fox sedge (*C. annectens*, left) and brown fox sedge (*C. vulpinoidea*, right), remain upright, facilitating combine harvesting.

communities. *C. cristatella* grew so aggressively within the bed that a heavy thatch of first-year vegetation formed a mat, suppressing early season growth and possibly seed production in year two. Fall or early spring fire may be beneficial (by removing thatch) for increasing seed production of this species in particular, and of *Carex* species in general. Lodging may lessen in coming years as plants fully occupy available root-space and thus have less-vigorous shoot growth, but seed production will likely decline, as well.

Estimated seed yields for most *Carex* species ranged from 194 kg/ha to 436kg/ha (168 lbs/ac to 376 lbs/ac), with a low of 33 kg/ha (29 lbs/acre) for *C. cristatella* and a high of 498 kg/ha (429 lbs/ac) for *C. vulpinoidea* (Table 2). Seed counts for these species range from 600-4233 seeds/g (17,000-120,000 seeds/oz) (Table 3). Again, issues with lodging of seed heads probably reduced recoverable seed yields of species mentioned previously. Seed yield of *C. cristatella* in production bed was surprisingly low, given its apparent abundant seed production in native stands. Vigorous vegetative growth and complete colonization of the production bed during *C. cristatella* first growing season may have reduced flowering and seed production the second year. Transplanting this species at a lower density (greater spacing between and within rows) may improve seed production.

Table 3. Estimated market value per unit area of seed harvest (2010) based on PLS yield for each *Carex* spp.

SPECIES	MARKET PRICE* ($/PLS#)	SEED COUNT* /G (/OZ)	$PLS/UNIT AREA $/M² ($/FT²)
C. annectens	$300	3,175 (90,000)	$20.93 (1.88)
C. bebbii	$150	1,199 (34,000)	$ 7.56 (0.68)
C. bicknellii	$150	600 (17,000)	$ 5.25 (0.47)
C. brevior	$150	1,023 (29,000)	$ 9.37 (0.84)
C. cristatella	$450	2,046 (58,000)	$ 2.36 (0.21)
C. molesta	Not commercially available		
C. tribuloides	$600	4,233 (120,000)	$43.27 (3.89)
C. vulpinoidea	$120	3,527 (100,000)	$11.18 (1.01)

**Prairie Moon Nursery, Winona, MN catalog price, 2010*

Harvested perigynia were screened and aspirated to high purity and submitted to a certified seed-testing lab for testing for percent purity and viability (tetrazolium or TZ test) (Table 4). High purities are obtainable by keeping production stands free from non-crop species (e.g., weeds as well as other *Carex* species), and by proper and thorough cleaning. High viability is a function of harvesting perigynia at maturity and proper aspiration to remove any light (unfilled) seed.

SUMMARY

The versatility of the plasticulture system provides efficient irrigation minimizes weedy competition during establishment, and shows great potential for seed production of native *Carex* species. Challenges with plasticulture include the cost of specialized equipment and removal and disposal or recycling of plastic mulch at the end of its productive bed life. Equipment costs may be nominal compared to the cost of tractors, combine harvesters, and specialized seed-processing equipment (Houseal 2010). Carrying plastic mulch beds over for two and three growing seasons for the benefits of weed suppression and nutrient and water retention is possible. Using prescribed fire as a management tool would not be advisable unless irrigation t-tape can be placed well beneath soil surface to avoid damage.

Table 4. Seed test results from selected Carex species second full growing season after transplanting.

SPECIES	PURITY (%)	TZ (%)	PLS (%)
C. annectens	98.5	84	82.8
C. bebbii	97.5	88	85.8
C. bicknellii	94.3	87	81.8
C. brevior	99.5	90	90.0
C. cristatella	94.9	76	72.2
C. molesta	98.8	84	83.0
C. tribuloides	96.9	76	75.3
C. vulpinoidea	99.5	86	85.1

Estimated yields for *Carex* species overall were in a range that could be economically viable for commercial seed production if market demand is in place (Table 4). *Carex vulpinoidea* is already available in the native seed trade in the Midwest, presumably because it is in demand and profitable. It is fairly easy to identify and propagate, and seed is easy to harvest and clean. Our work shows that other *Carex* species (*C. annectens, C. bebbii, C. brevior, C. molesta, C. tribuloides*) can be similarly productive in cultivation, and may be equally or more profitable, given their current higher market value (e.g., *C. annectens*, *C. tribuloides*), if markets develop for these species.

Our work also indicates that it is possible to collect and propagate *Carex* as individual species if proper field and lab techniques are used to avoid cross-contamination with similar species. We will continue to assess yields, stand life, and weed pressure in agronomic production of these and additional *Carex* species to determine which are practical for agronomic seed production using this system.

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SEED BANK COMPOSITION AND ECOLOGY IN A DISTURBED FLOODPLAIN GRASSLAND

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Abstract: The seed bank of a reconstructed prairie at Chichaqua Bottoms Greenbelt in Polk County, Iowa, was examined with a seedling assay method from 2009 to 2010. The study site was undergoing secondary succession due to a prolonged flood event in 2008, as well as experimental cattle herbivory to investigate grazing effects on reconstructed prairie. Three environmental factors were incorporated into the design: (1) the location of the seed bank community along an environmental gradient, (2) the impact of cattle herbivory, and (3) the depth of seed in the soil. Soil cores 6.5 cm in diameter and 6 cm deep were collected in September 2009 and stored at 2°C for 12 weeks to provide stratification before the seedling assay was initiated on December 16. Emerging seedlings were identified, counted, and removed over the next five months. A total of 3,223 seedlings were observed, representing at least 43 plant species. Seed bank densities ranged from 9,140 to 12,540 seeds/m^2/6 cm depending on the seed bank community. Statistical analyses (three-way ANOVA) were competed on nine taxa. The seed bank density of certain species varied significantly relative to location, herbivory, and soil depth. Ordination of the seed bank samples demonstrated that location along the environmental gradient and soil depth were the two most important factors affecting species composition of the seed bank. A conspicuous lack of warm-season native perennial grasses in the seed bank was evident despite a decade of dominance and presumed seed production on the site prior to flooding.

INTRODUCTION

Soil seed banks are the collection of viable seeds that have dispersed and are present in or on the surface of the soil or in the litter on the soil surface. Three types of soil seed banks are recognized by the classification scheme of Thompson et al. (1997), which is based on the longevity of seeds. Transient species persist in the soil seed bank for less than one year, short-term persistent species are present for more than one and less than five years, and long-term persistent species are present for five years or longer. Seeds are removed from the seed bank by germination, predation, fungal and bacterial decomposition, fire, flooding (lack of oxygen), or natural senescence.

Soil seed banks serve several important roles in the plant community. First they add to the plant species richness present in the community, as both the extant vegetation and the seed bank comprise the totality of individuals present in an environment. Quite often and depending on successional status, there may be a strong disparity between the species present in the seed bank and those observed in the vegetation (Rabinowitz 1981, Wienhold and Van der Valk 1989, Rosburg et al. 1994). Thus the seed bank increases the true plant species diversity of the habitat to fully represent the entire flora (Major and Pyott 1966). Secondly, the seed bank provides for regeneration of vegetation after disturbance (Harper 1977, Fenner 1985). In this way, seed banks are receiving attention for their potential contribution in the restoration of native ecosystems (Baskin and Baskin 1998), in particular temperate grasslands (Bakker et al. 1991), freshwater wetlands (Van der Valk et al. 1992), and heathlands (Putwain and Gillham 1990). Third, seed banks provide population stability. This is particularly true of persistent seed banks that buffer the variance of population size, reduce the risk of population extinction (Levin 1990), and increase species survival in risky environments (Baskin and Baskin 1998). Fourth, persistent seed banks can affect population gene pools by buffering fluctuation in genetic composition, biasing selection toward traits favored in seasons of high seed production, and providing a source of new genetic variation from mutations that occur during seed dormancy (Levin 1990). The enrichment of genetic diversity and increase in population stability provided by soil seed banks is especially important in endemic species with small population sizes (Baskin and Baskin 1978).

The context of this study is a four-year project designed to investigate the impact of cattle herbivory on the plant species composition and structure of a reconstructed prairie. The hypothesis under examination is that perturbation and disturbance created by cattle grazing will provide a mechanism to increase the plant species richness and structural heterogeneity of the prairie. This in turn is expected to enhance the habitat value for birds, snakes, small mammals, and butterflies. During the first year of the study, in 2008, unprecedented flooding occurred on the study site from late May through early July. Cattle grazing was delayed until early August and ended about a month later. The prolonged flood conditions caused very high mortality to the previously dominate warm-season native grasses, specifically big bluestem (*Andropogon gerardii*) and Indiangrass (*Sorghastrum nutans*). In 2009 cattle grazing occurred as designed, with the cattle moved onto the study site in late May and removed in mid-September. However, the original research goal—to examine the role cattle grazing could have in suppressing and opening a dense sward of warm-season native grass—was seriously compromised by the severe disturbance of the 2008 flood. Because seed banks play a vital role in establishment of early successional communities, it become clear that information about the study site's seed bank would be helpful in understanding the site's vegetation dynamics.

Although numerous studies have examined the seed banks of disturbed habitats like arable fields (Archibold 1981, Roberts and Nielson 1981, Leck and Leck 1998, Shaukat and Siddiqui 2004) and grasslands or prairies (Rabinowitz 1981, Johnson and Anderson 1986, Abrams 1988, Rosburg et al. 1994, Rosburg and Owens 2006), very few if any have attempted to discern the effect of grazing on a grassland seed bank. Herbivores could affect seed banks in several ways. Selective grazing of grassland plants could either prevent or promote seed production and dispersal of plant seeds and subsequently their presence in the seed bank. Species could decrease in the seed bank due to removal of plant biomass and inability to produce seed. Alternatively, species avoided by grazers could increase seed production and dispersal due to reduced competition and improved resource availability. Large ungulate grazers could also create soil surface conditions, either soil exposure or soil compacting, that increase the establishment of species recruited from the seed bank.

The objectives of the seed bank study were (1) to gather descriptive data on the seed bank composition of a disturbed (flooded) reconstructed grassland, (2) to evaluate the effect of cattle herbivory on the seed bank composition of a grassland, and (3) to examine the ecology and influence of seed depth on the seed bank composition of a grassland.

METHODS

STUDY SITE

This research was conducted at Chichaqua Bottoms Greenbelt, a 3,683 ha conservation area centered along the Skunk River in northeast Polk County (2,954 ha) and northwest Jasper County (729 ha). Since property was first acquired in 1960, land acquisition opportunities and a strong focus on restoration by the Polk County Conservation Board have made Chichaqua one of the largest restoration projects in the country. Over 2,020 ha have been reconstructed to a native ecosystem, an unusual accomplishment for a county conservation agency. The area's natural wetland communities include old oxbow river channels and backwaters, marshes, sedge meadows, swamp white oak and bur oak savanna, floodplain tallgrass prairie, and floodplain forest. Chichaqua also contains upland landscapes with reconstructed prairies and native sand prairie remnants.

The study site for the seed bank research is a 188 ha floodplain tract that was formerly an agricultural field (Figure 1). It was seeded to tallgrass prairie in 1998. Over the last ten years, the tract has been burned with prescribed fire four times and hayed twice. The grazing study was initiated in 2008. Approximately 89 cow/calf pairs and 16 heifers grazed the tract from early August to mid-September, after vegetation recovered sufficiently from the flood. In 2009, grazing began in late May when 64 cow/calf pairs were moved onto the site. In late June, 5 bulls were added. The maximum stocking rate for the year was reached when an additional 23 pairs were added in early July. In late August, 39 cow/calf pairs and the 5 bulls were removed. The remaining cattle were moved off in mid-September.

Figure 1. This aerial photo shows the study site at Chichaqua Bottoms Greenbelt in northeast Polk County, in central Iowa. The yellow (solid) line approximates the location of the boundary fence for the grazed prairie reconstruction. The blue (dashed) line identifies the location of the drainage ditch along the east boundary. The boxes identify the approximate locations of cattle exclosures where paired study plots are present. Those labeled west, central, and east are the plots where seed bank samples were collected in this study.

The climate in central Iowa is temperate and continental. The 30-year average monthly minimum and maximum temperatures at Des Moines for January are -11.3° C and -1.6° C, respectively (National Climatic Data Center, NOAA). The 30-year average monthly minimum and maximum temperatures for July are 18.9° C and 30.0° C, respectively. Annual precipitation is 88.2 cm, with May, June, July, and August the wettest months (each with an average of more than 10 cm). The average growing season for Polk County is 171 days, from April 21 to October 9 (McCracken 1960).

FIELD AND GREENHOUSE METHODS

The seed bank of a reconstructed prairie at Chichaqua Bottoms Greenbelt was examined with a seedling assay method from 2009 to 2010. Three environmental factors were incorporated into the design (Figure 2):

1. Environmental variation characterized by potential differences in the local soil environment and the extant vegetation and represented by three distinct sample locations within the grazed reconstructed prairie. These sample locations are incorporated in the design as statistical blocks and were positioned along an east-west transect that represents an environmental gradient associated with flood duration in 2008.
2. The impact of cattle herbivory, attained by sampling paired study plots that were either grazed or excluded from grazing in 2008 and 2009.
3. Depth in the soil, attained by separating soil cores into two equal sections of 0-3 cm depth and 3-6 cm depth.

Essentially, the study consisted of a 2 x 2 factorial (2 levels of grazing and 2 levels of soil depth) with its 4 treatments replicated in three blocks.

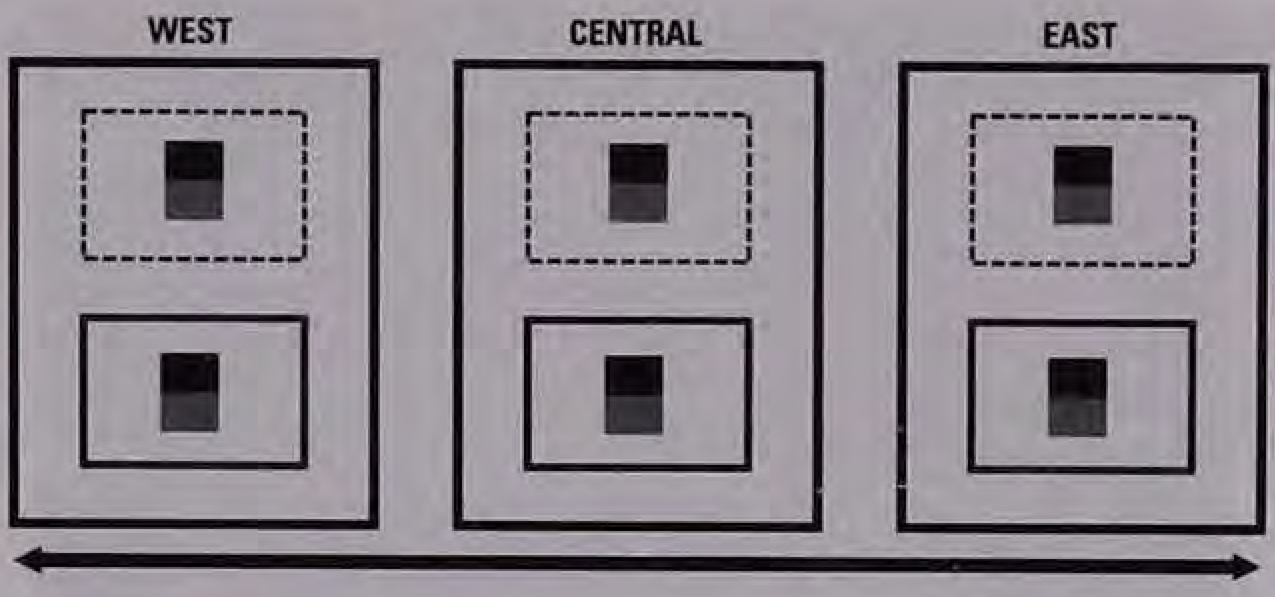

Figure 2. The study design is illustrated diagrammatically, where the largest rectangles represent the three blocks. Two study plots are located within each block: the dashed line represents the grazed plot and the double line represents the exclosure. The small shaded boxes represent soil samples, or soil cores, where the dark shading signifies the 0-3 cm soil depth and the lighter shading corresponds to the 3-6 cm soil depth. Each of these soil samples is the collective soil volume of 16 soil cores extracted along a 30 m transect. The double arrow at the bottom characterizes the location of the blocks along an environmental gradient.

Soil cores were collected in September 2009 from three blocks on a grazed, reconstructed prairie (Figure 1). Each block consisted of a pair of study plots, one inside a 20 x 40 m exclosure and one adjacent and outside the exclosure. The exclosures are constructed with an electrified fence. A 30 m transect was established within each study plot, and 16 soil cores each 6 cm deep and 6.5 cm in diameter were extracted from the soil at 2 m intervals along the transect. The soil cores were divided into two sections, from 0 to 3 cm and from 3 to 6 cm depth, and pooled together (Figure 2). The 16 cores provide a total of 1,592 cm^3 of soil collected per soil sample (a combination of block, grazing/nongrazing, and depth).

Soil samples were stored at 2°C for 12 weeks to provide stratification before the seedling assay was initiated on December 16. At that time that soil was passed through a wire screen (mesh of 0.6 cm) to remove roots and plant debris. The four treatments (resulting from the 2 x 2 factorial) and three blocks resulted in 12 different seed bank samples. Each seed bank sample provided three seedling assay replicates, each consisting of 500 cm^3 of seed bank soil spread in a thin layer over potting soil in 20 x 20 cm trays, and resulting in a total of 36 trays (replicates) utilized in the study. Because seedlings were observed and counted in the 20 x 20 cm trays, and because each tray represents a unique treatment, the trays are the entity providing replication.

Trays were placed on a table in the Pioneer Hybrid Greenhouse at Drake University in Des Moines, Iowa. They were watered every second day to keep the soil moist to encourage germination. Supplemental lighting was provided to maintain a minimum of 12 hours of daylight for the first two months. Seedlings were counted on every watering day for one month to measure germination rates. When seedlings had grown to an adequate size they were identified, counted, and removed from the trays. In some cases, seedlings were transplanted to larger pots to allow them more time to grow and develop sufficiently for identification. The seedling assay was terminated in mid-May 2010, after five months of seedling germination and growth. Plant nomenclature follows Eilers and Roosa (1994).

ANALYTICAL METHODS

Plant species or genera with adequate representation in the seed bank were utilized in a statistical analysis to examine the effect of explanatory variables on seed density. Nine taxa with a minimum of 90 total seeds observed were statistically analyzed. Two of the nine taxa—Carex and Cyperus species (sedge and nutsedge respectively)—were lumped together for this analysis since identification to genera was not possible for most individuals unless they had developed sufficiently to flower. Data were analyzed with a three-way ANOVA using the blocks, grazing or nongrazing treatment, and soil depth as explanatory variables in the model. Replication for statistical analyses was derived from the replicate seedling assay trays, these being the three trays containing seed bank soil from the same combination of block, grazing/nongrazing treatment, and soil depth. An α less than or equal to 0.05 was used for assigning significant results, while an α between 0.05 and 0.10 was assigned a marginal result.

A DECORANA ordination was performed on the seed bank samples to examine the pattern of similarity in species composition among the communities represented by the combination of block, grazing/nongrazing, and soil depth. The seed data from the three replicate seedling assay trays (i.e., from the same seed bank sample) were summed for each of the 12 seed bank samples. The ordination matrix consisted of these 12 community samples and 40 plant species. The unidentified forb and graminoid seedlings were excluded from these data. As in the ANOVA analyses, all Carex and Cyperus species were lumped together and treated as a single taxa for this ordination.

RESULTS

A total of 3,223 seeds were recovered from the soil samples, representing at least 43 plant species (Table 1). It is likely that more species were present since 445 seeds (13.8%) were not identified to genera; rather they were identified only as either a graminod or a forb. Forb species accounted for 30 of the total plant species (69.7%), while graminoids comprised the remaining 13 species (30.3%). Forb species encompassed an even greater proportion of the total seeds in that 2,565 forb seeds (79.6%) were observed, in comparison to 658 graminoid seeds (20.4%). Among the 43 identified taxa, all but two (Lactuca and Amaranthus species) could be classified as native or exotic, and those taxa were represented by 29 native taxa (70.7%) and 12 exotic taxa (29.3%). Among the 2,774 seeds identified sufficiently to classify to origin, native species accounted for an even greater proportion (2,458 native, or 88.6%, vs. 316 exotic, or 11.4%). These data indicate that forb seed was present in the seed bank at a greater rate per species than the seed of graminoids. Similarly, native seed was present in the seed bank at a greater rate per species than the exotic seed. Both of these results can be at least partially explained by the fact that native forbs were the largest group represented with 18 species (44.0%). Exotic forb and native graminoid categories were each represented with 11 species (26.8%), and exotic graminoids were the least rich with one species (2.4%).

Table 1. Total seeds observed for all species by soil depth in all seedling assay trays. The mean seed density estimates an average based on a uniform dispersion of seed over all 12 soil samples. Actual species seed densities are likely either lower or higher in a particular location due to non-uniform dispersions.

SPECIES	SEEDS OBSERVED			MEAN SEED DENSITY
	TOTAL	0-3 CM	3-6 CM	SEEDS/M^2/ 6 CM DEPTH
Veronica peregrina	743	488	255	2,477
Rorippa sessiliflora	636	266	370	2,120
Eupatorium serotinum	361	323	38	1,203
Ammania coccinea	189	175	14	630
Setaria faberii	154	123	31	513
Echinochloa muricata	126	108	18	420
Polygonum persicaria	94	71	23	313
Panicum dichotomiflorum	84	68	16	280
Carex/Cyperus species	59	58	1	197
Oxalis stricta	44	31	13	147
Cyperus esculentus	30	21	9	100
Mollugo verticillata	28	9	19	93.3
Panicum capillare	25	17	8	83.3
Polygonum pensylvanicum	24	16	8	80.0
Portulaca oleraceae	19	12	7	63.3
Lythrum alatum	18	14	4	60.0
Juncus dudleyi	17	17	0	56.7
Ambrosia artemisiifolia	13	11	2	43.3
Capsella bursa-pastoris	13	11	2	43.3
Cyperus squarrosus	13	12	1	43.3
Aster species	12	12	0	40.0
Agrostis hyemalis	11	10	1	36.7
Conyza canadensis	9	6	3	30.0
Lycopus americanus	8	7	1	26.7
Potentilla norvegica/rivalis	8	8	0	26.7
Lindernia dubia	7	7	0	23.3
Chamaecrista fasciculata	5	5	0	16.7
Carex species	5	1	4	16.7
Cyperus strigosus	4	4	0	13.3
Amaranthus species	3	0	3	10.0
Lamium purpurem	2	2	0	6.7
Polygonum lapathifolium	2	1	1	6.7
Bidens species	1	1	0	3.3
Chenopodium album	1	0	1	3.3
Euphorbia maculata	1	1	0	3.3
Lactuca species	1	1	0	3.3
Panicum virgatum	1	1	0	3.3
Silene latifolia	1	1	0	3.3
Solidago canadensis	1	1	0	3.3
Sonchus arvense	1	1	0	3.3
Spartina pectinata	1	1	0	3.3
Stellaria media	1	0	1	3.3
Taraxacum officinale	1	1	0	3.3
Trifolium repens	1	1	0	3.3
Forb Seedling	317	305	12	1,057
Graminoid Seedling	128	102	26	427
TOTAL OF ALL SPECIES	**3,223**	**2,331**	**892**	

Although the study site was dominated by warm-season native grass species (primarily big bluestem and Indiangrass) prior to the flood of 2008, there were only two seeds of warm-season native grass detected in this study. One seed of *Spartina pectinata* (prairie cordgrass) and one seed of *Panicum virgatum* (switchgrass) were observed (Table 1). Therefore, recovery of the tallgrass component on the flooded reconstruction is not likely to occur by recruitment from the seed bank.

Total seed density for all species and for a 6 cm soil depth ranged from 9,140 to 12,540 seeds/m2 (Table 2). In general, the 0-3 cm soil depth contained just over 2½ times (160%) as many seeds as the 3-6 cm soil depth, or, in other words, the 0-3 cm depth comprised 72% of the seeds in the top 6 cm of soil. On average, the grazed samples contained from 10% to 20% more total seeds than the nongrazed samples.

Veronica peregrina (purslane speedwell) was the most abundant species in the seed bank, followed by *Rorippa sessiliflora* (stalkless yellow cress), *Eupatorium serotinum* (late boneset), Ammania coccinea (scarlet toothcup), *Setaria faberi* (giant foxtail), and *Echinochloa muricata* (barnyard grass) (Table 1). These six species each had over 100 seeds observed. Other important species, those with a minimum of 50 seeds observed, include *Polygonum persicaria* (spotted lady's thumb), *Panicum dichotomiflorum* (fall panicum), and the group of Carex and Cyperus species. Among those species with at least 10 seeds observed, only two—stalkless yellow cress and *Mollugo verticillata* (carpetweed)—exhibited a greater number of seeds in the 3-6 cm depth than in the 0-3 cm depth.

Nine taxa were statistically analyzed to more fully investigate patterns of seed density in the seed bank (Figures 3-10, Carex and Cyperus grouped as one taxon). Seven of the nine passed the equal variance test for the three-way ANOVA. The two that failed—purslane speedwell and scarlet toothcup—were analyzed with three-way ANOVA despite the failed assumption for the sake of consistency and uniformity in analysis.

Significant block effects were evident for eight of the nine taxa. Only giant foxtail (Figure 9) demonstrated a uniform density along the east-west transect. Barnyard grass (Figure 5), spotted lady's thumb (Figure 7), and purslane speedwell (Figure 10) were most abundant in the seed bank of the west block. Late boneset (Figure 6) and stalkless yellow cress (Figure 8) were most abundant in the seed bank of the east block. Scarlet toothcup (Figure 3) and sedges/nutsedges (Figure 4) were most abundant in the central block.

Grazing effects were observed for six of the nine taxa. Five taxa—sedges/nutsedge, barnyard grass, spotted lady's thumb, and giant foxtail—exhibited significantly higher seed densities in the grazed samples. Only late boneset displayed higher seed density in the nongrazed plots. Scarlet toothcup, stalkless yellow cress, and purslane speedwell were neutral in their response to grazing.

All nine taxa demonstrated an effect of seed depth on seed density in the seed bank. And all species, except stalkless yellow cress, exhibited significantly higher seed density in the 0-3 cm depth than in the 3-6 cm depth. Stalkless yellow cress presented significantly higher seed density in the 3-6 cm depth.

Table 2. Mean total seed density of all species (seeds/m²/depth) among blocks, herbivory treatments and depth.

	WEST		CENTRAL		EAST		MEAN
	NOT GRAZED	GRAZED	NOT GRAZED	GRAZED	NOT GRAZED	GRAZED	
0 TO 3 CM DEPTH	6,280	9,020	6,540	8,180	8,060	8,360	7,740
3 TO 6 CM DEPTH	2,860	2,060	3,080	2,520	3,140	4,180	2,973
TOTAL CORE	9,140	11,080	9,620	10,700	11,200	12,540	10,713

A significant herbivory and seed depth interaction was observed for six species, while a marginal result was detected for one species. A significant interaction means that the effects of herbivory and seed depth were not additive. In other words, the effect of one factor was influenced by the other factor. The overwhelming nature of the interactions in this study was that the combination of 0-3 cm depth and grazing produced higher seed density than expected if additive. This explains the significant interactions observed for sedges/nutsedge, barnyard grass, spotted lady's thumb, and giant foxtail. In late boneset, the result was opposite; grazing decreased seed density in the 0-3 cm depth. These patterns mimic the pattern described for the main effects of grazing, and should be used in deference to them since the presence of an interaction means that the main effects of factors cannot be simply interpreted. The marginal interaction was observed in stalkless yellow cress, and was manifest as an increase in the seed density of the 3-6 cm depth under the influence of grazing (Figure 8).

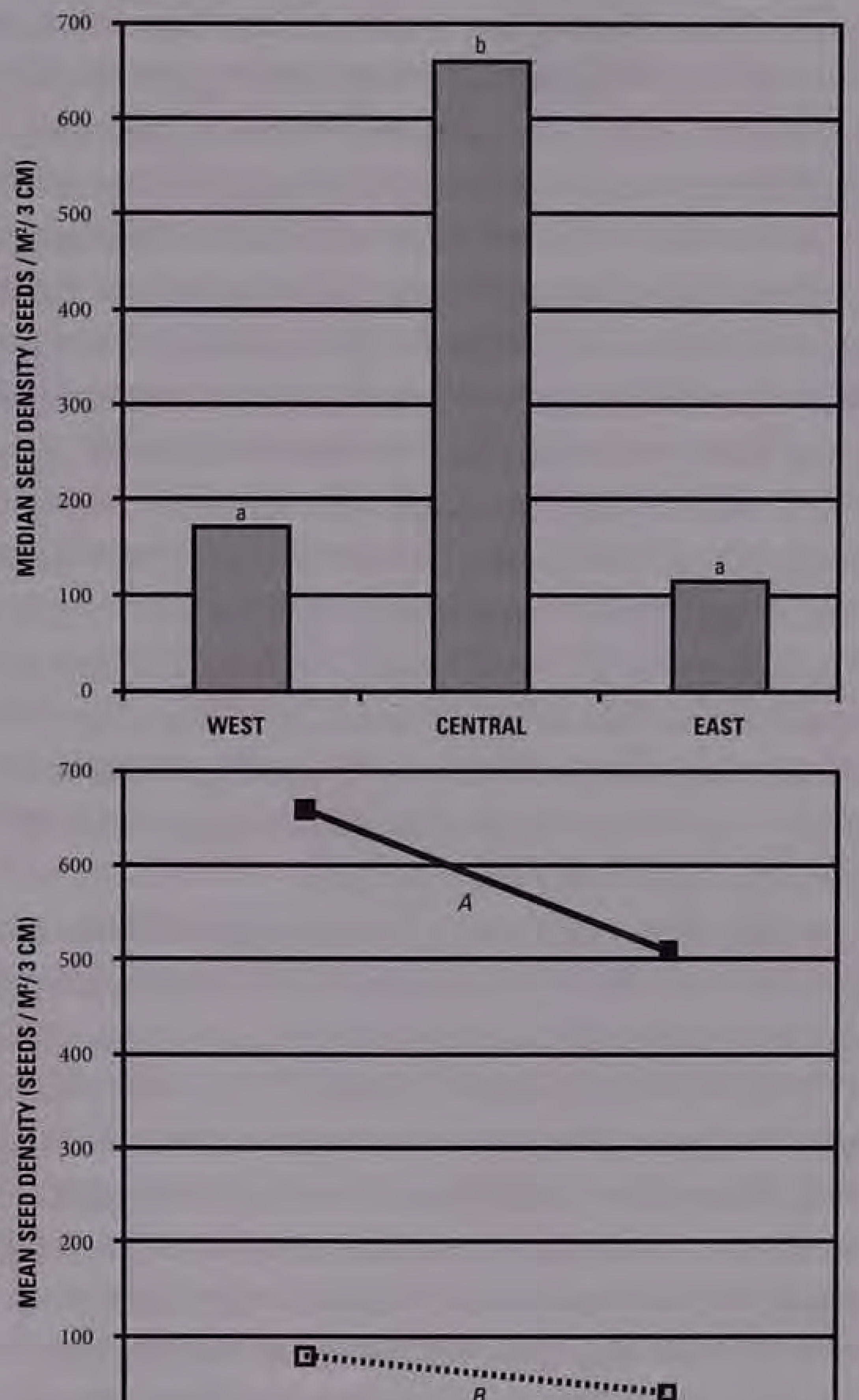

Figure 3. Results for *Ammania coccinea* (scarlet toothcup). The upper graph displays the results for the block effects (p=0.057). Lowercase letters that are dissimilar identify a marginal difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase non-italicized letters, if present, identify a statistical difference in the main effect of herbivory. Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth (p=0.01). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated.

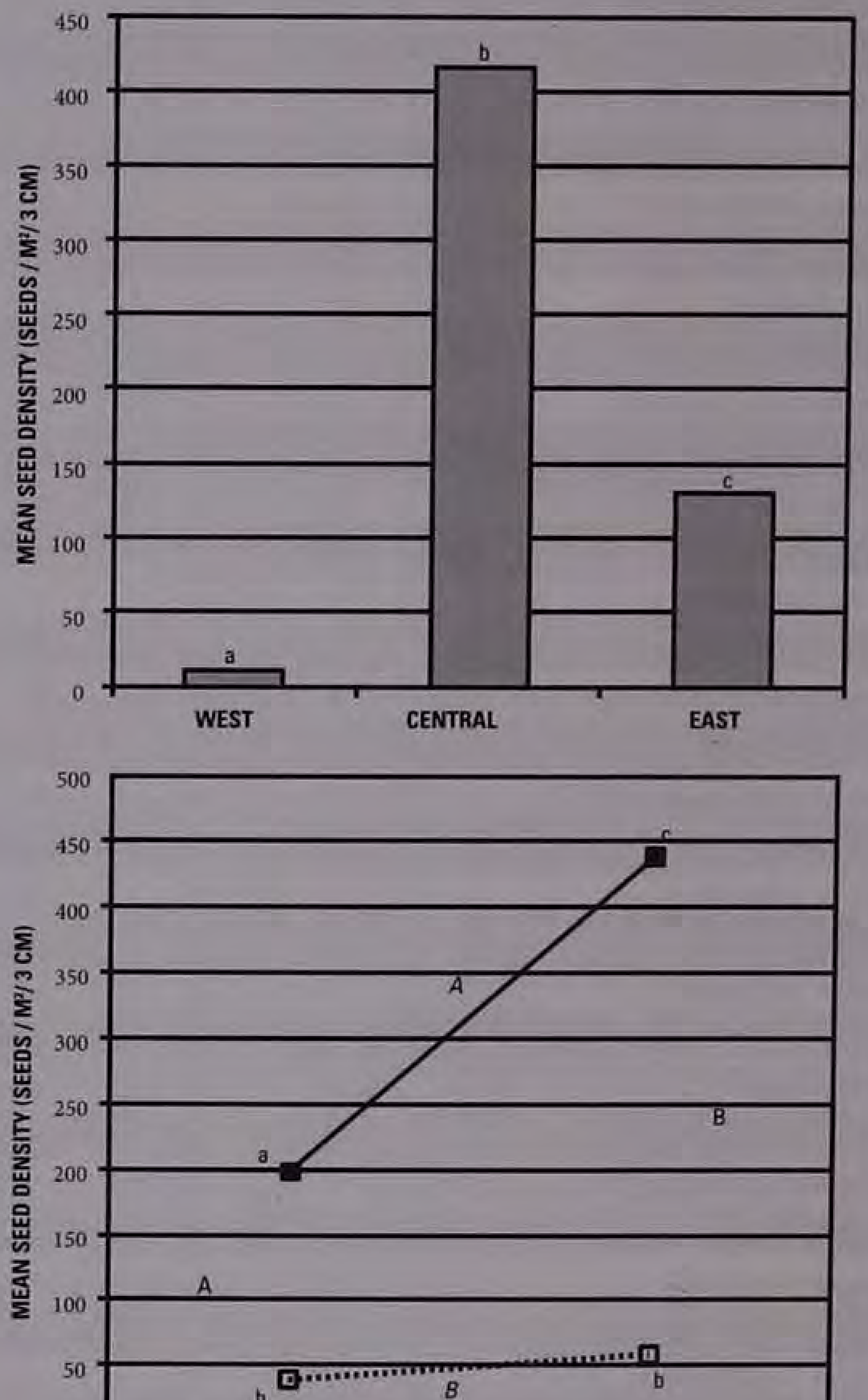

Figure 4. Results for *Carex/Cyperus* (sedge/nutsedge species). The upper graph displays the results for the block effects (p<0.001). Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase non-italicized letters, if present, identify a statistical difference in the main effect of herbivory (p=0.003). Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth (p<0.001). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated (p=0.009).

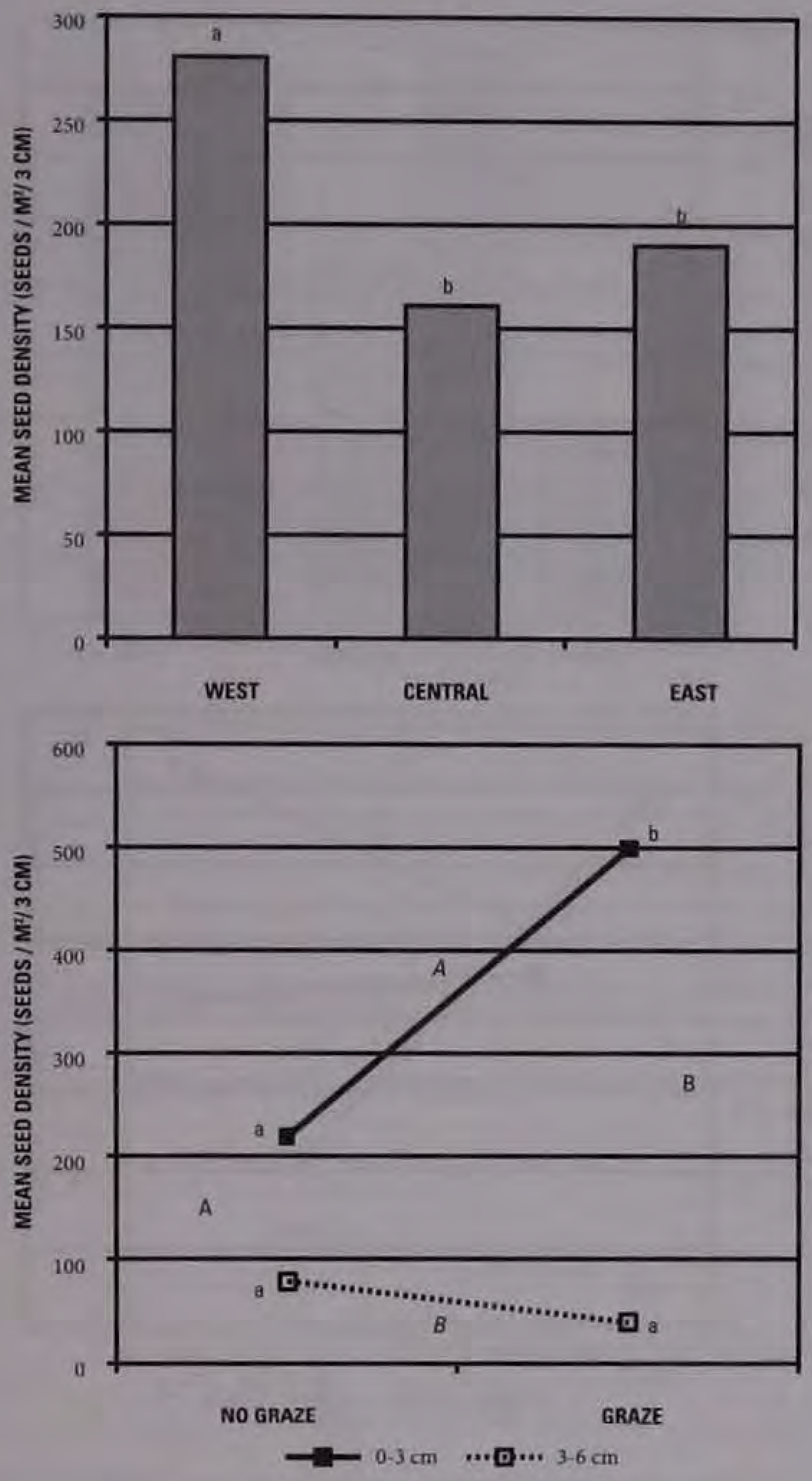

Figure 5. Results for *Echinochloa muricata* (barnyard grass). The upper graph displays the results for the block effects (p=0.03). Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase nonitalicized letters, if present, identify a statistical difference in the main effect of herbivory (p=0.003). Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth (p<0.001). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated (p<0.001).

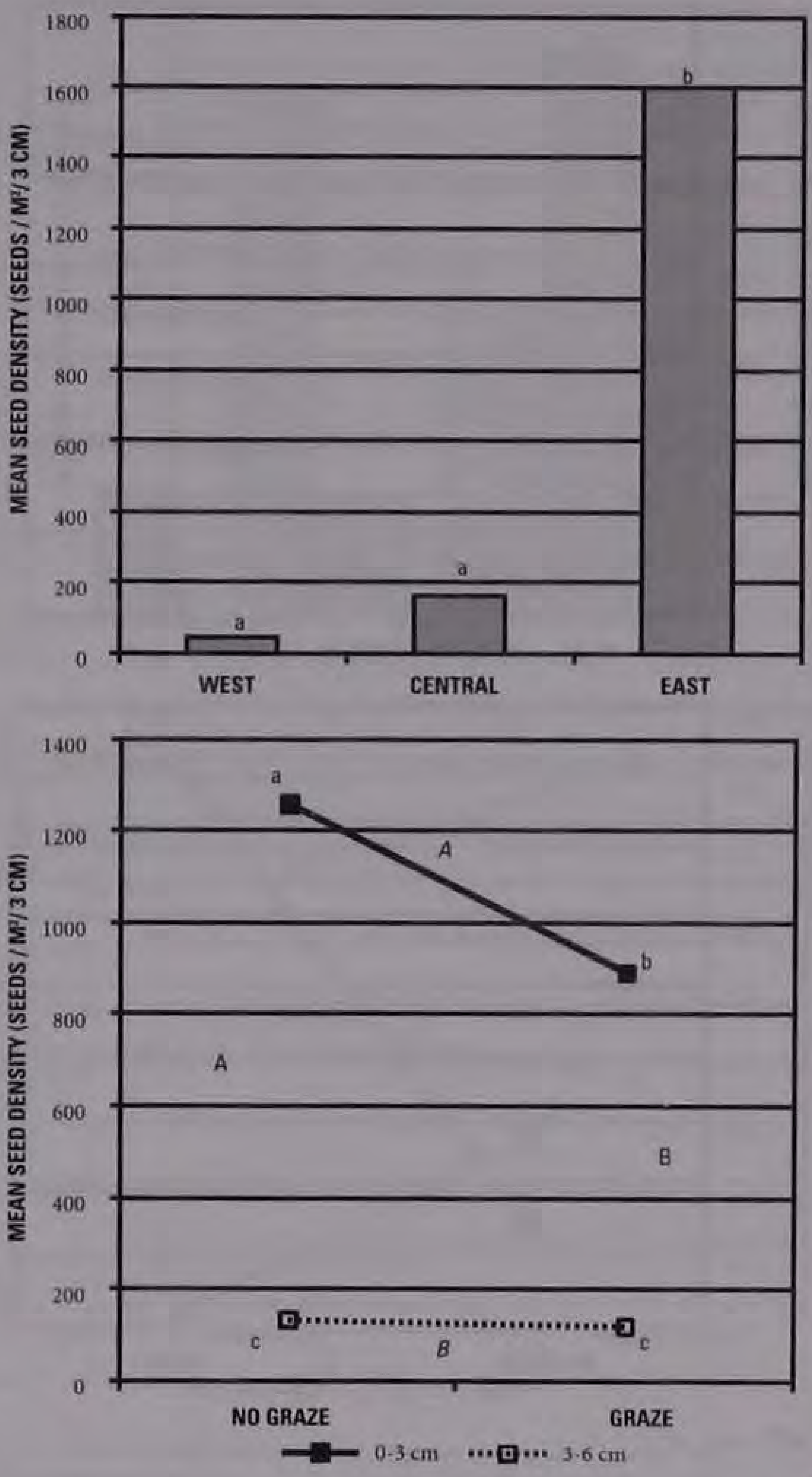

Figure 6. Results for *Eupatorium serotinum* (late boneset). The upper graph displays the results for the block effects (p<0.001). Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase nonitalicized letters, if present, identify a statistical difference in the main effect of herbivory (p=0.001). Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth (p<0.001). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated (p=0.002).

The ordination of the 12 seed bank samples based on their species composition (i.e., the seed density of 40 species) demonstrates that the blocks account for the most variation in seed bank composition (Figure 11). The samples representing three blocks are almost entirely separated along Dca axis 1, the compositional gradient explaining the most variation in the species composition of the seed bank samples. The blocks in the ordination are also correlated to their physical position along the environmental gradient (i.e., the central blocks are positioned between the west and the east blocks). The second most important factor is soil depth. The 0-3 cm depth samples are almost entirely along the lower end of Dca axis 2, while the 3-6 cm depth samples are along the upper end of Dca axis 2. The effect of herbivory is not clearly represented in the first two Dca axes, as grazed and nongrazed samples are intermixed and dispersed throughout the ordination.

DISCUSSION

The flood event in 2008 severely compromised the goals of the cattle-grazing study in its ability to evaluate the hypothesis that cattle grazing could serve to create perturbance and disturbance that leads to greater structural heterogeneity in the vegetation and increased biodiversity. The high mortality of the warm-season native grasses due to the flood confounds the ability to assess grazing effects on the dominant grasses. Because the study site had at least a decade of native grass growth and seed production prior to flooding, there seemed to be potential for recolonization by warm-season native grasses from the seed bank sufficient enough to compensate for the mortality from the flood, and thereby to recuperate some ability to examine the original study's hypothesis. The insight gained from this study into the site's seed bank composition was an important goal, and it strongly indicates

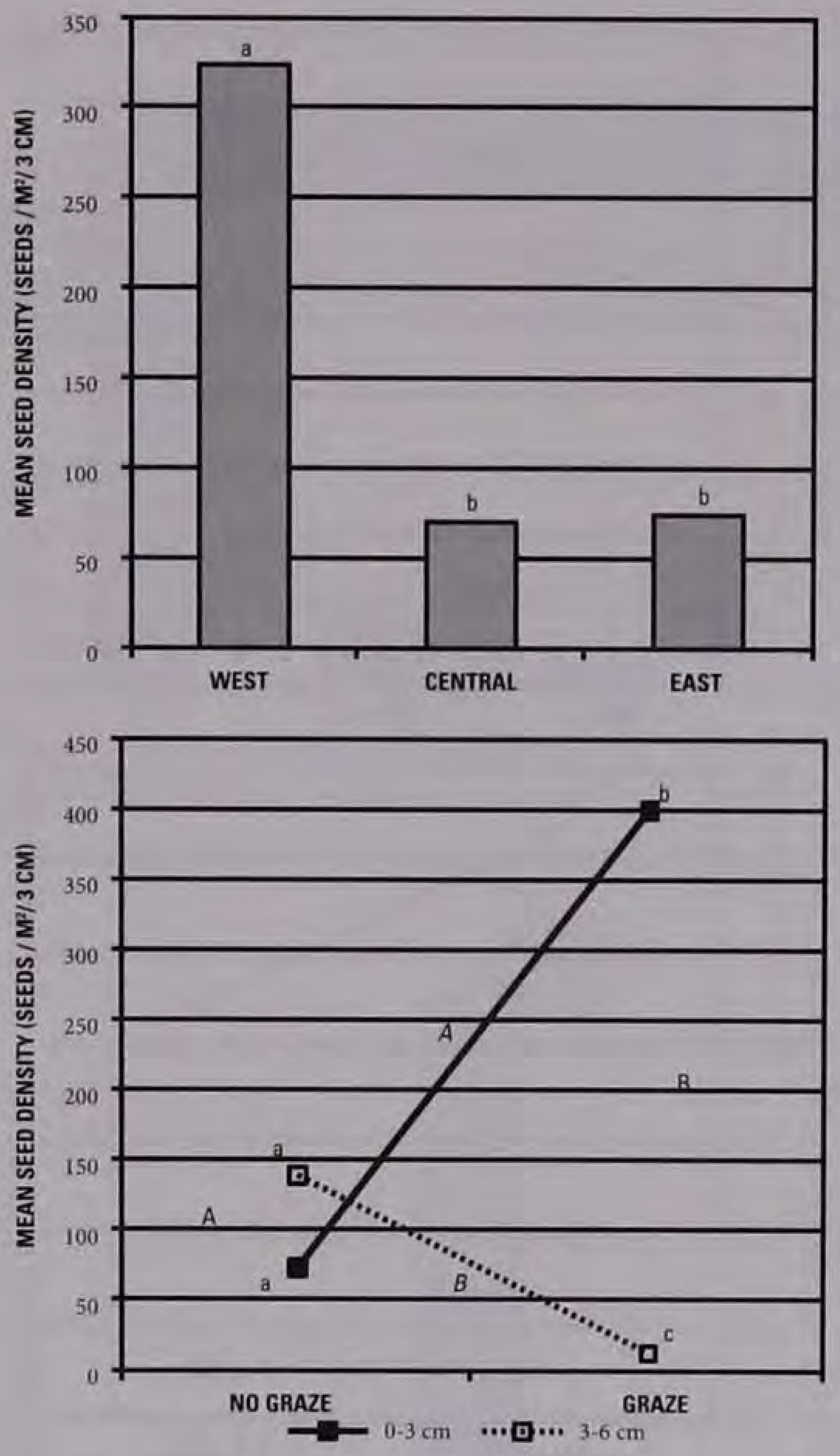

Figure 7. Results for *Polygonum persicaria* (spotted lady's thumb). The upper graph displays the results for the block effects ($p<0.001$). Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase non-italicized letters, if present, identify a statistical difference in the main effect of herbivory ($p=0.002$). Upper case italicized letters, if present, identify a statistical difference in the main effect of soil depth ($p<0.001$). The location of the upper case letters approximate the magnitude of the main effect means. Lower case letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated ($p<0.001$).

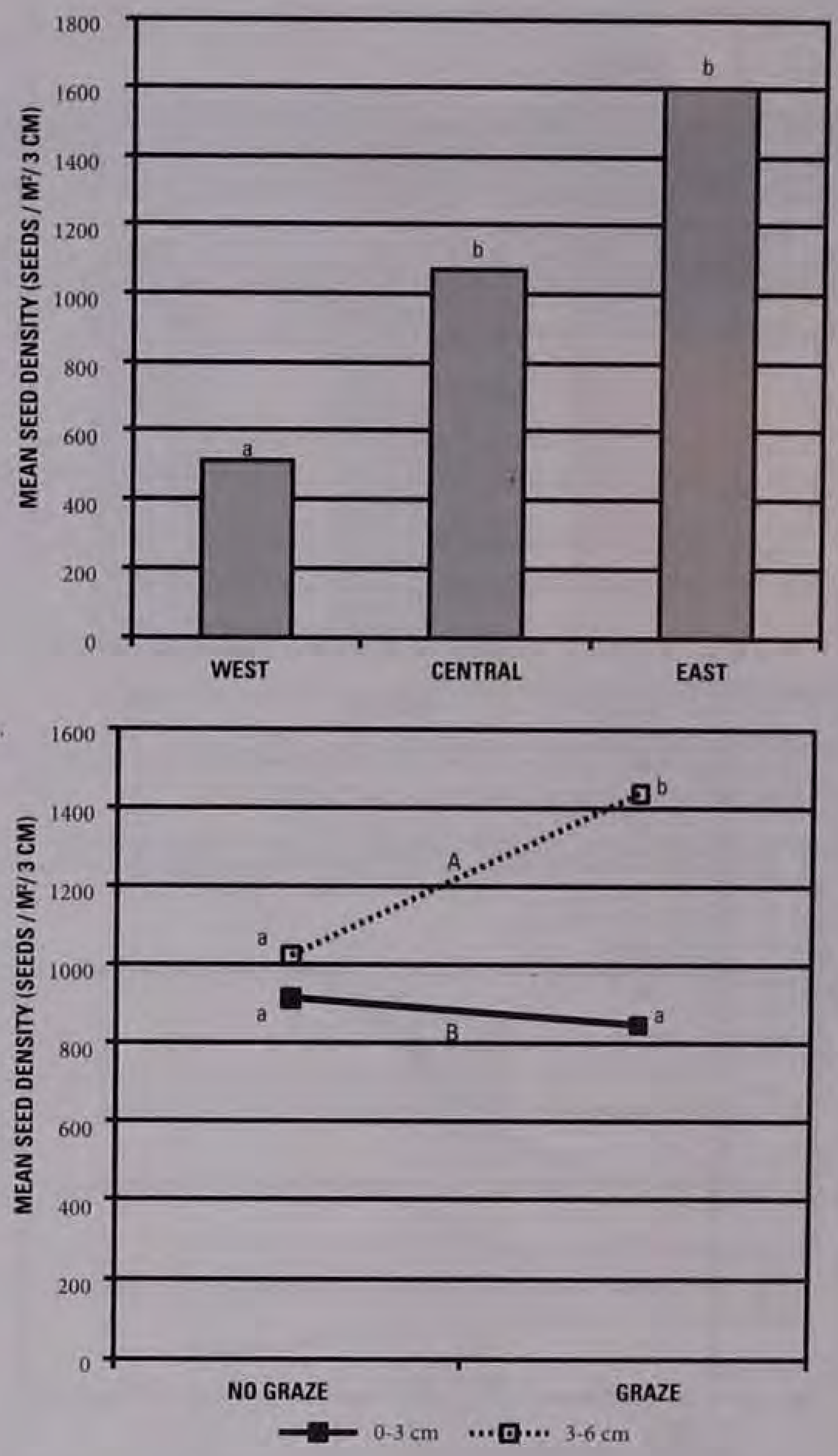

Figure 8. Results for *Rorippa sessiliflora* (stalkless yellow cress). The upper graph displays the results for the block effects ($p<0.001$). Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase non-italicized letters, if present, identify a statistical difference in the main effect of herbivory. Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth ($p=0.015$). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated ($p=0.083$).

that the seed bank should not be expected to provide a source of warm-season grass seed.

The lack of any seed of big bluestem and Indiangrass in the seed bank is somewhat surprising, given their enormous presence in the vegetation and likely copious seed rain over several years prior to the flood. Three explanations are possible—either these species do not form even short-term persistent seed banks (i.e., they have transient seed banks due to low dormancy), or they incur high seed mortality while in the seed bank, or it is possible that their seed was missed by the seedling assay. Row and Wynia (2010) reported seed viability of 60%-65% after ten years for both big bluestem and Indiangrass stored in a controlled environment. After 35 years of storage, Indiangrass maintained 70% viability, while big bluestem had decreased to 30%. They also found a very clear effect of environment on longevity. For all eight of the native C4 grasses examined, longevity decreased dramatically when seed was stored in an uncontrolled environment. Big bluestem and Indiangrass exhibited 35% viability after five years and 0% viability at ten years. Stable temperatures and lower and more stable relative humidity promote greater seed longevity in these grass species (Row and Wynia 2010). The uncontrolled storage experienced temperatures from -8°C to 33°C and relative humidity between 50% and 80%. Temperature in the controlled environment ranged from 18°C (summer) to 13°C (spring and fall) and 3°C (winter) and relative humidity was maintained between 10% and 20%. Two caveats associated with these results are (1) that the eight grasses used were cultivars, and therefore could have somewhat greater seed vigor than native types; and (2) that storage in an artificial environment is not the same as storage in the soil seed bank. Both of these argue for less longevity under natural conditions. Nevertheless, given the general results of Row and Wynia (2010), it seems, at least with respect to the role of seed dormancy and longevity, that some seed of big bluestem and Indiangrass should have been observed. Thus the second factor, high seed mortality, could be an explanation for the absence of big bluestem and Indiangrass seed.

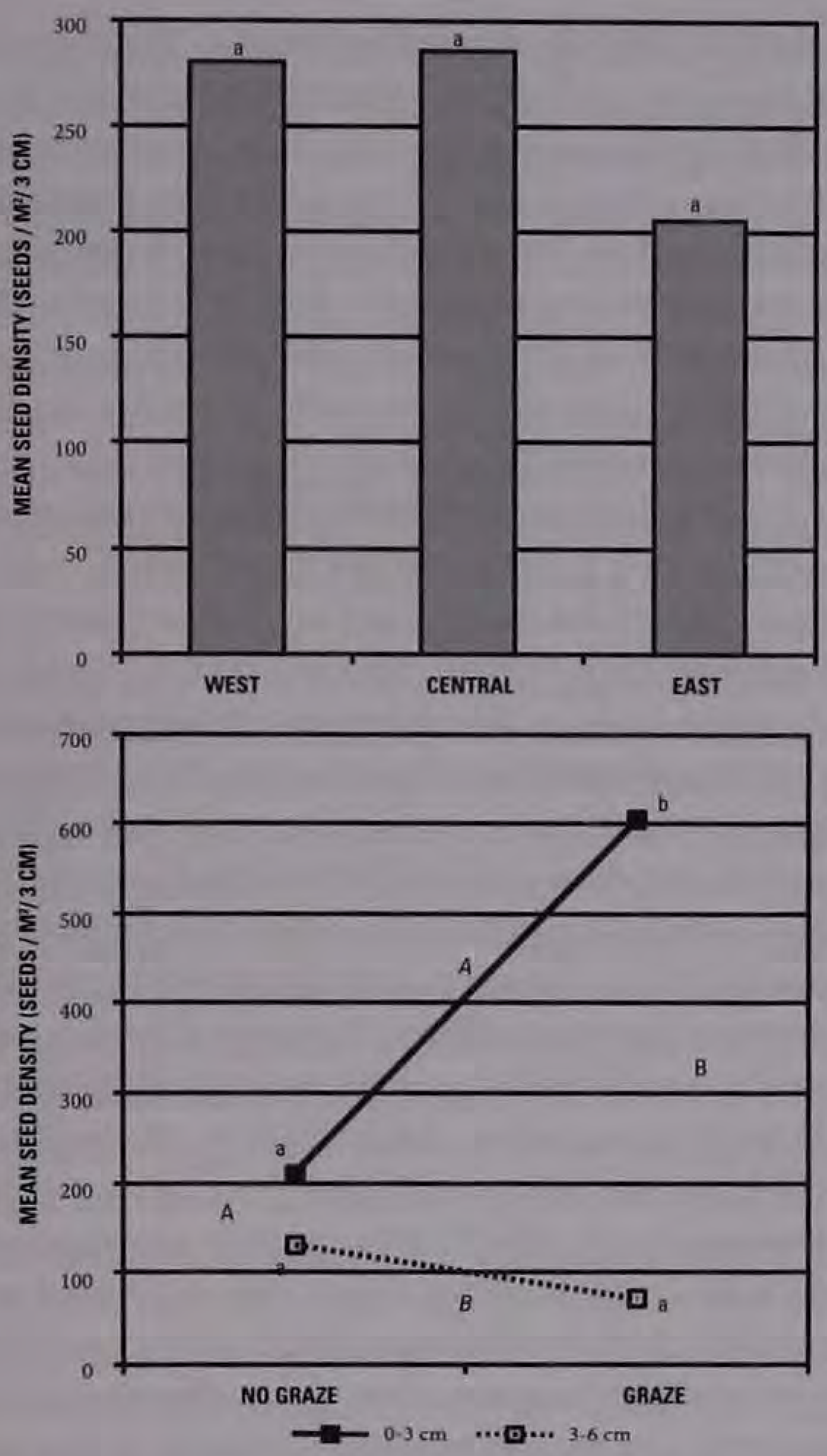

Figure 9. Results for *Setaria faberi* (giant foxtail). The upper graph displays the results for the block effects. Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase non-italicized letters, if present, identify a statistical difference in the main effect of herbivory ($p<0.001$). Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth ($p<0.001$). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated ($p<0.001$).

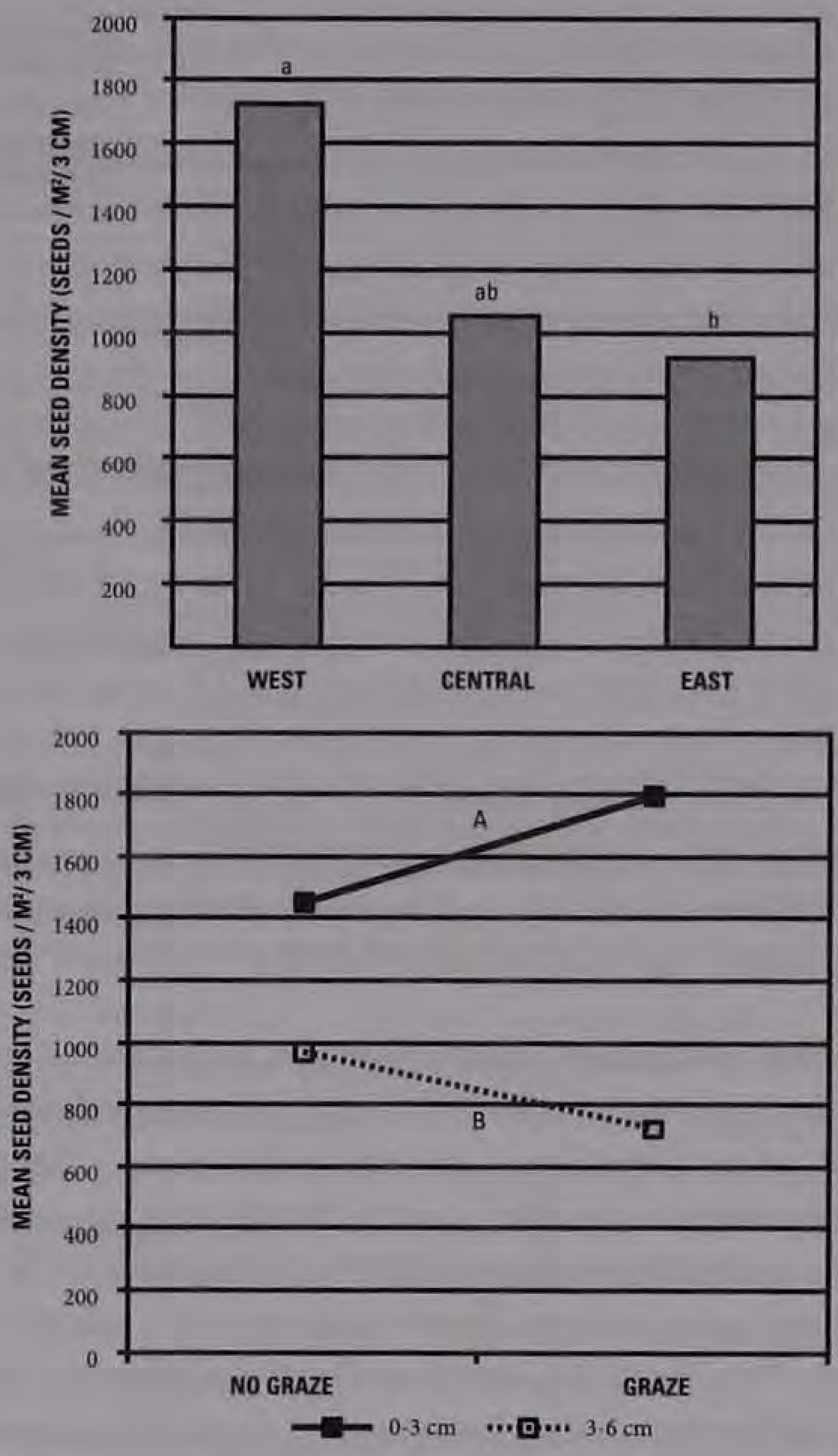

Figure 10. Results for *Veronica peregrina* (purslane speedwell). The upper graph displays the results for the block effects ($p=0.034$). Lowercase letters that are dissimilar identify a significant difference in seed density among blocks. The lower graph presents the results for the factorial effects. Uppercase non-italicized letters, if present, identify a statistical difference in the main effect of herbivory. Uppercase italicized letters, if present, identify a statistical difference in the main effect of soil depth ($p=0.005$). The locations of the uppercase letters approximate the magnitude of the main effect means. Lowercase letters positioned near the four means graphed are used to identify differences in means when a statistically valid interaction is demonstrated.

Seed mortality is very difficult to measure. It includes post-dispersal predation by small mammals, birds, and insects, as well as pathogenic mortality from fungi, protozoans, and bacteria. Clark and Wilson (2003) reported that among three possible fates for seeds—persistence as dormant seeds, germination, and mortality—that seed mortality accounted for the largest proportion among four grassland species in Oregon. A native perennial grass exhibited 80% mortality, of which 21% was caused by vertebrate predators, 8% was due to fungi, and 51% was unmeasured (invertebrate predation, bacterial disease, or senescence). Granivores removed about 90% of test grass seeds dropped into tallgrass prairie plots in Kansas, of which vertebrate predation accounted for nearly all (seed removal by invertebrates was less than 5%) (Reed et al. 2006). Seed predation by vertebrates increased with increasing seed density, from 35% (at 1,000 seeds/m^2) to 75 to 90% (at 50,000 seeds/m^2) for two perennial grass species in a European grassland (Edwards and Crawley 1999). These studies suggest there is a good likelihood that seed predation could significantly contribute to the absence of big bluestem and Indiangrass seed in the seed bank.

The practical absence of warm-season native grasses in the seed bank is not so surprising when the results of other studies are considered. Many prairie seed bank studies have demonstrated that late-successional, K-selected species, such as big bluestem and Indiangrass, are uncommon in the seed bank even though they are common in the vegetation (Rabinowitz 1981, Abrams 1988, Rosburg et al. 1994, Rosburg and Owens 2006). The results of this study are in agreement with those studies. These grasses have evolved good ability for vegetative growth (probably as a response to grazing and drought), thus the importance of seed production is minimized, at least in the short term. K-selected species (or C-selected in the Grime 1977 classification) do not need to find disturbances in time, thus there has not been selective pressure to evolve seeds with high dormancy and longevity.

The third possibility is that their seed was missed by the seedling

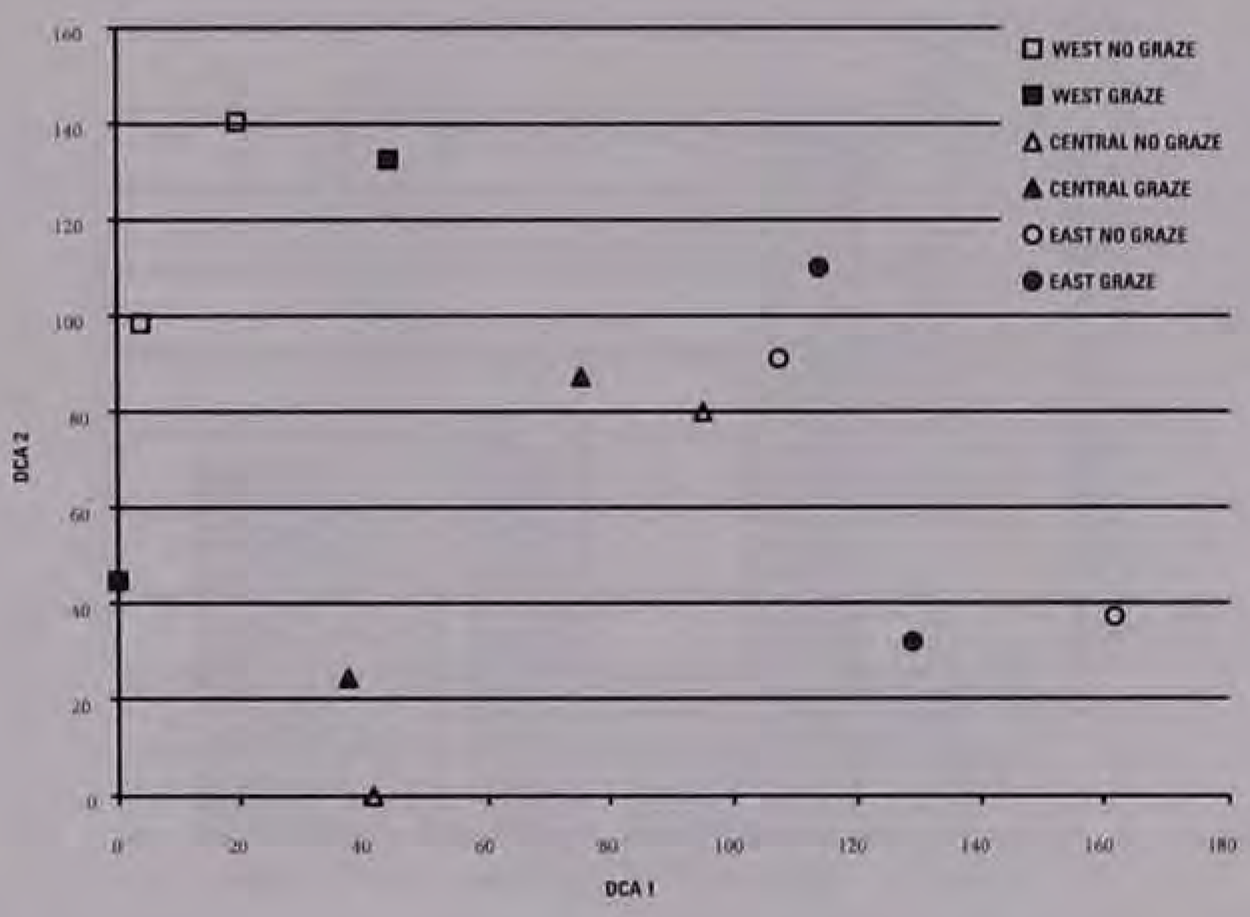

Figure 11. Decorana ordination of the 12 seed bank samples (the combination of block, herbivory, and soil depth) in species space based on the seed density of 40 shared plant species. The legend identifies the sample's block and herbivory state. The uppercase letters identify the sample's soil depth; T represents the 0-3 cm depth and B represents the 3-6 cm depth.

assay. Any seed that was innately dormant or that did not experience the correct germination cues would not have germinated and would not have been observed in the seedling assay. This possibility should have been minimized by the stratification seeds were exposed to and the greenhouse environment provided. Still, there are always going to be seeds that do not germinate in seedling assay studies. Very few studies have examined the same seed bank with both seedling assay and seed assay methods, but in at least one study by Van der Valk and Rosburg (1997), the seedling assay method found only 52% of the species richness that was identified with seed assay methods. For species found by both methods, seed assays also find greater numbers of seeds than do seedling assays (Van der Valk and Rosburg 1997), thus seed densities are more accurately estimated. However, seedling assays can find species that seed assays miss (for example, three species in Van der Valk and Rosburg 1997), and they are much more easily done. In the end, it may be a combination of all three mechanisms that accounts for the very low seed presence of big bluestem and Indiangrass. Their late-successional status, energy allocation to perennial and vegetative growth, strongly competitive strategies, and low fecundity would be associated with low seed longevity and formation of short, persistent seed banks at best, perhaps only two to three years (O'Connor 1991). This life-history strategy, combined with a high rate of seed predation and an assay method that is not well suited to finding species with low seed density, could produce the results observed in this study.

Comparisons among this study and others with regard to total seed densities and species observed are complicated by differences in methods. However, it does appear that the seed bank in this study exhibited total seed densities that were much greater than in seven other grassland studies (Table 3). Since seed density is standardized for area in each study, the chief difference is in the depth of the soil cores and the resulting volume of soil observed. The 6 cm deep cores in this study are midrange among the seven other studies (2.5-12 cm deep), yet the seed density is the highest of all the studies, and with a maximum seed density nearly 2.7 times greater than the next highest maximum (Table 3). The early successional status of the grassland in this study is an important difference from most of the other habitats that could account for the high seed density, since high fecundity and seed longevity are adaptations of ruderal species. Most of the other sites (Table 3) are later successional prairie. Since the samples for the Chichaqua study were collected at the end of the growing season of the year following the flood disturbance, early successional species that colonized the site postflooding in either 2008 or 2009 could have dispersed considerable amounts of seed prior to the collection of seed bank samples. Comparisons of species richness are confounded by variation in the total sample volume. Still, the number of species observed in this study is within the typical range exhibited (Table 3) and is somewhat at the high end of the range, given the amount of soil volume sampled. The percentage of native species is lower than all but one of the other studies. This could also be explained by the relatively early successional status which would promote more ruderal species, which in turn favors exotic species.

Table 3. Comparison of grassland seed bank studies. An entry of "NA" occurs when information was not available.

STUDY	YEAR	LOCATION	HABITAT	SOIL SAMPLE TOTAL SURFACE AREA (CM^2)	DEPTH OF CORE (CM)	TOTAL SEEDLINGS OBSERVED	TOTAL SEED DENSITY SEEDS/M^2/ DEPTH	TOTAL SPECIES	NATIVE SPECIES
Rosburg and Green	current	Iowa	Disturbed recon prairie	3,186	6	3,223	9,140 to 12,540	43	30
Rosburg and Owen	2006	Iowa	Recon prairie	8,482	3	2,696	1,400 to 4,500	62	37
Rosburg et al.	1994	Iowa	Loess Hills dry tallgrass	28,840	6	2,555	720 to 1,160	87	77
Abrams	1988	Kansas	Dry tallgrass prairie	3,534	5	735	1,450 to 2,890	28	≈27
Johnson and Anderson	1986	Illinois	Tallgrass prairie	3,927	10	793	2,019	45	≈38
Rabinowitz	1981	Missouri	Tallgrass prairie	793	12	513	6,470	28	≈26
Archibold	1981	Saskatchewan	Cool-season prairie & grazed pasture	7,238	10	NA	739 to 476	8 to 11	NA
Johnston et al.	1981	Alberta	Grazed cool-season prairie	82,467	2.5	NA	880 to 4,555	33	NA

The preponderance of block effects among the nine taxa analyzed demonstrates that a species' seed bank is typically heterogeneous or patchy over relatively fine scales. The environmental gradient represented by the blocks corresponded to flood duration, as the west end of the gradient was flooded for a shorter period in 2008 than was the east end. The 2008 floodwaters drained off the 170-ha study site by slowly entering a drainage ditch on the east side (Figure 1). This may have caused differential survival of seeds in the seed bank due to low oxygen levels for a longer period in the east block. Thus it is possible that species like spotted lady's thumb, barnyard grass, and purslane speedwell may have lower tolerance to flooding while in the seed bank since their seed density was highest in the west block. Likewise, late boneset and stalkless yellow cress are indicated as species with relatively high tolerance to flooding while in the seed bank.

Differences in seed bank density among the blocks could result from vegetation patterns as well. For example, the high density of sedge/nutsedge species in the central block corresponds to the presence of a large patch of sedge in and near the study plot that presumably provides a source of seed rain. Late boneset was very abundant in the vegetation of the east block and completely absent in the vegetation of the west block in the fall of 2008. The flood created an early secondary successional community on the study site in fall 2008 and for 2009. Seed rain from these plant populations was most likely the source of seed that was observed in the 0-3 cm soil depth. Thus the pattern of the vegetation in 2008 and 2009 would have had a strong influence on the seed bank observed. The nature of that pattern could have been influenced by a suite of several different factors, including the composition of the long-term persistent seed bank present in 2008; differential survival of species during the flood; patchy seed dispersal of species by the floodwaters; differential germination of seed postflood in 2008 due to microenvironment variation in temperature, light, and oxygen; differential deposition of new sediment on the surface and the subsequent variation in seed depth; and the impact of spatial autocorrelation due to population processes such as short seed dispersal, vegetative growth, and local species pools. As a whole, all of these factors could promote considerable heterogeneity in the early successional vegetation, which could lead to heterogeneity in the seed rain and the seed bank patterns observed among the blocks in this study. Late boneset deserves further special mention in that it was virtually unknown at Chichaqua Bottoms, and Polk County in general, prior to the flood. Its distribution in Iowa is confined primarily to the southeastern counties (Eilers and Roosa 1994). It emerged from the flood in late summer of 2008 in high densities all across the eastern third of the study site, in the area that experienced the longest flood duration in 2008. It is not known when or how it established such a huge presence in the seed bank of the study site from an apparent distant source population. One theory is that seed may have been introduced from afar by hay equipment.

Grazing effects were observed for several of the nine species examined. Cattle creating a more opportune environment for seed production and seed rain during 2009 could explain the higher seed bank densities in the grazed plots. The mechanism for this would have to be related to decreased competition due to selective herbivory, which seems plausible for spotted lady's thumb and sedge/nutsedge, species that the cattle probably avoided. However it is not very plausible for barnyard grass and giant foxtail, as these warm-season annuals were providing the principal forage for the cattle in the absence of perennial grasses. For these two species, the higher seed densities on the grazed plots could be the result of higher seed predation in the non-grazed plots due to greater litter depth and habitat cover for small mammals. The decrease in seed bank density for late boneset if due to cattle grazing, could be due to the trampling of cattle reducing the number of plants able to disperse seed. All six of these species demonstrated an interaction between herbivory and soil depth that more fully clarifies the nature of herbivory effects on seed bank density. The herbivory effects either were only observed in the 0-3 cm soil depth or were enhanced in the 0-3 cm depth. This is expected since cattle herbivory can influence only the amount of seed rain or the amount of immediate post-dispersal predation, and either of these effects is manifest only in the 0-3 cm depth where seed rain (or lack thereof) is deposited. Seed is not very likely to go directly from the plant to the 3-6 cm depth; it has to pass through the 0-3 cm depth first.

The three species that did not exhibit grazing effects are all small species that flower and disperse seed early in the growing season. They were likely already in seed-dispersal mode by the time cattle grazing started in 2009, and thus their seed banks would be expected to be unaffected by grazing. The six species that exhibited grazing effects flower and disperse seed at the end of the growing season, which makes them more susceptible to cattle-grazing effects.

Differences in seed bank density due to soil depth were the most prevalent; all nine species exhibited a pattern. Eight of the nine (all except stalkless yellow cress) displayed the "normal" pattern in which seed bank density is higher in the 0-3 cm depth, or, more generally speaking, higher near the surface than deeper in the soil. This pattern is expected for species that are in the vegetation and are producing a fairly consistent annual seed rain. Seed is deposited on the surface and, as it gradually trickles deeper into the soil over time, some of it is lost. Viable seed should always be more numerous near the surface than deeper in the soil. A departure from this pattern suggests a different scenario. For stalkless yellow cress, the higher seed density deeper in the soil suggests a historical occurrence in the vegetation and a pulse of seed rain that has relatively high longevity. The recent lack of seed rain due to absence in the vegetation diminishes the seed density near the surface. Thus stalkless yellow cress would appear to have had an important presence in the vegetation historically, but has been absent or nearly absent from the vegetation in recent years. Vegetation surveys completed at the central and east blocks in the fall of 2008 confirm that the species was not observed. Plant species that exhibit this pattern of seed bank density must have a long-term persistent seed bank. It may be possible to estimate their seed longevity by determining when they last would have flourished in vegetation, this being the last time that seed was produced in large amounts. For stalkless yellow cress, a fairly small annual mustard, that may have been when the study site was last row-cropped in 1997, prior to prairie reconstruction.

Finally, the results of the ordination help to visualize the patterns of variation in the total species composition of the seed bank samples. The samples segregate along DCA axis 1 according to their location along the environmental gradient sampled. This confirms that relatively larger spatial factors, those manifest in the blocks and discussed previously, have the most important effect on the species composition of the seed banks. Keep in mind that all of these samples originate from a floodplain which, compared to the larger landscape, is clearly homogeneous in its physical environment. The composition of all the samples is distinctly floodplain, but within this floodplain seed bank community, intracommunity patterns are apparent due to population-level processes.

The second most important factor is soil depth, which is segregated on DCA axis 2. Movement of seed from the surface into the soil seed bank requires time. Factors that favor seed movement into the soil and deeper burial include small seed size and rounded shape (Fenner and Thompson 2005). These physical traits should enhance movement of seed through soil from mineral expansion and contraction due to temperature change, movement from soil-stirring from animal activity in the soil, and from water percolation. The texture of the soil affects seed movement: fine textures (clay, silt) will trap or hold small seeds more effectively than coarse textures, and large seeds are trapped more successfully by coarse textures than fine textures (Chambers et al. 1991). Animals, both vertebrates and invertebrates, can cause species differences in seed burial—for example, ants (Holldobler and Wilson 1990), beetles (Bernhardt 1994), and earthworms (Hurka and Haase 1982, Grant 1983). Thus the potential for variation in seed burial rate among species is compelling, and, combined with decreasing longevity over time, could exert a strong sorting mechanism to seed presence at depth in the soil seed bank.

Grazing effects on the species composition of the soil seed bank were the least important of the three factors examined. Grazed and nongrazed samples are distributed throughout the ordination, and, more important, their position relative to their paired nongrazed sample is not consistent. Grazing introduces variation in the species composition of soil seed banks, but not in a consistent way (at least with respect to DCA axes 1 and 2). Being a biotic environmental factor, herbivory is more likely to be variable across spatial scales as animals exert a range of plant consumption and utilization due to herd behavior, individual behavior, individual preferences, competition, and animal densities.

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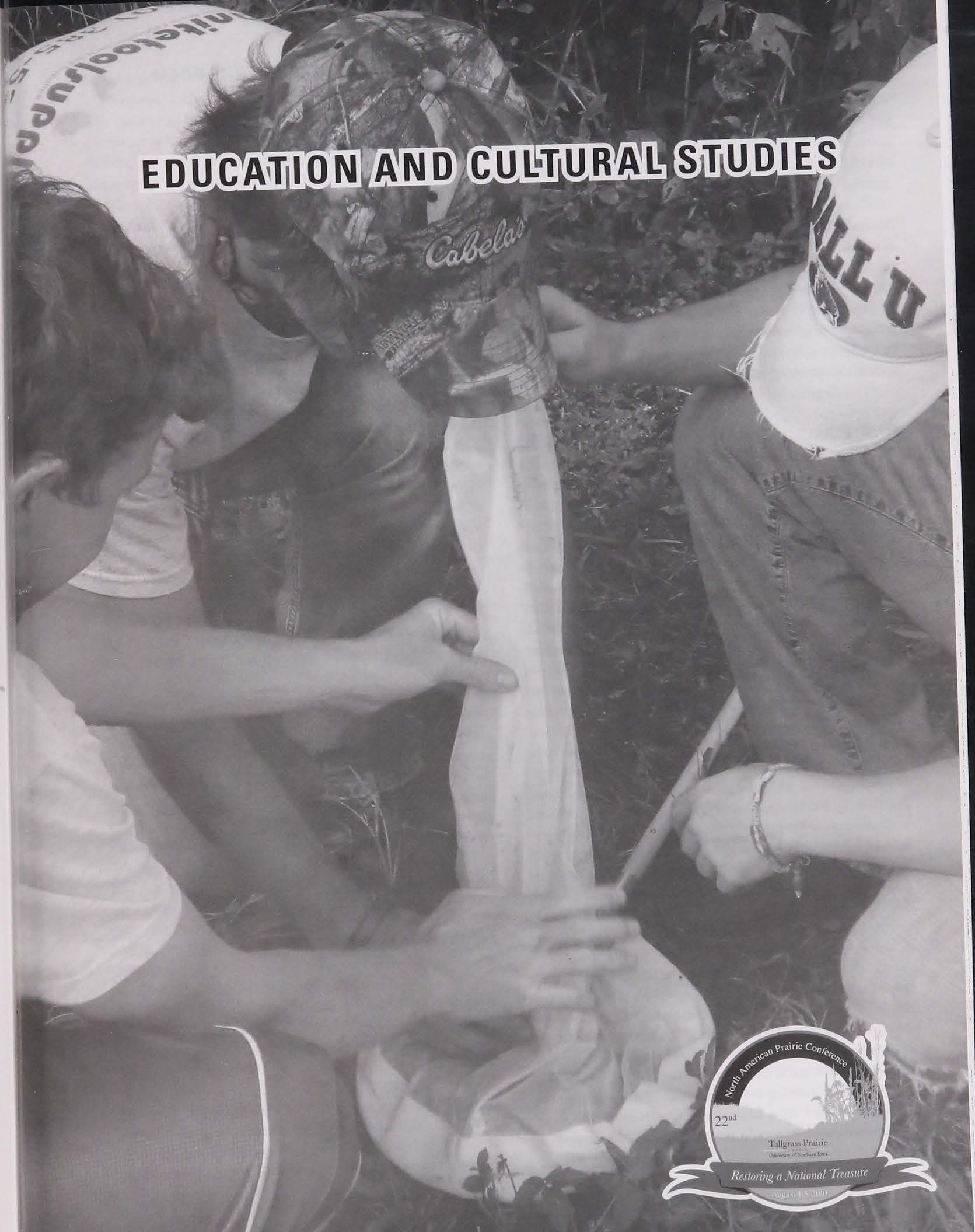
EDUCATION AND CULTURAL STUDIES



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AN INTRODUCTION TO UNDERSTANDING THE FLORA OF IOWA, THROUGH THE POEM "IOWA FLORA (IN MEMORY OF AMY CLAMPITT)"

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Abstract: Edward Hirsch's brief poem "Iowa Flora (In Memory of Amy Clampitt)" was written in 1996 as a part of *Broken Ground*, Jonathan Chenette's symphonic/choral suite commemorating the sesquicentennials of Grinnell College and the State of Iowa. The poem's loose confederation of imagery, mingling references to both native flora and invasive species, has provided us with a platform for teaching ecological literacy, reflecting on the Iowa of presettlement times and exploring issues of human immigration.

IOWA FLORA

During the last ten years, we have been combining our backgrounds in literature and ecology, using art and history as well, to address scientific literacy for non-science students and citizens. As Haufler and Sundberg (2009) have stated, "Society needs science, and scientists need an informed, thoughtful, and open-minded citizenry." Literature can inform through ecocriticism and natural history (ASLE 2010). By interweaving literature and service learning, science lectures, nature walks, photographs, film clips, documentaries, and cartoons, we attempt to enhance a scientifically informed understanding of ecosystems for different audiences, such as nature-writing and environmental literature classes, as well as a forthcoming college liberal-arts seminar, "Exploring Ecotones of Literature, Science, and History, or, What does it mean to be an ecologically literate global citizen in Pella, Iowa?"

For a previous college seminar, "Reading the Landscape: Prairie Education through American Literature," we included a collection of seven poems published in 1996. Jonathan Chenette, a professor of music at Grinnell College, arranged these poems, written by six poets, all with Iowa connections, setting them to music for a piece called *Broken Ground* which commemorated the sesquicentennials of both Grinnell College and the State of Iowa. One poem in the collection particularly caught our attention—"Iowa Flora (In memory of Amy Clampitt)," by Edward Hirsch. The title led us to expect accuracy and literalness in describing the plants of Iowa, but the poem departs from floristics into human-centered issues while equivocating about introduced species (Mack and Lonsdale 2000). An ecocritical reading of this poem seemed challenging. We could instead read "Iowa Flora" with a biographical lens, studying how background on the author and the poetic mentor, Amy Clampitt, might change the way we perceive the poem. We could also address its fleeting mention of mythology, using the poem's reference to "grass of Parnassus" and the allusion to poetic inspiration from the muses. But as a plant ecologist, Stephen was chagrined to think that people unfamiliar with botanical diversity might dismiss the importance of learning the particulars about Iowa flora, or fail to see the poem's symbolic significance as part of the Iowa sesquicentennial. The botanical ambiguity created concern in us, rather than delight, as we pondered it in the context of ecocritism and scientific literacy.

"Iowa Flora" seems to center on the tallgrass prairie, since the poem is immersed in the larger composition with its images of the plow breaking the ground, but it is by no means restricted to prairie. We thus provide a close reading of "Iowa Flora" to try to foster ecological literacy about the ecosystems of Iowa through (1) researching the plants mentioned in the poem and some of their metaphorical ramifications, in order to teach plant identification and ecosystem context; (2) teaching about habitats and migrations by discussing native versus invasive plants; and (3) tracing the words used in the poem that seem to conflate plants and people. Clouding the complexities of ecosystem biodiversity by equating them with human-centered issues may confuse students and distract them from the very ecological intricacies that we want to celebrate.

Our first two goals were to help readers recognize the plants that the author includes, directly or indirectly, in the poem∑∞ and to address the nebulous concepts of what is indigenous and what is introduced. Hirsch begins in the Iowa of a presumably mid-twentieth-century childhood, as the poet harks back to a time spent in idyllic wonder about plants: "*We thought we were having an indigenous childhood / splashed with Indian paintbrush and grassy knolls / thickened by birdfoot violets and ordinary goldenrod.*" He is probably describing one of two species of Indian paintbrush (*Castilleja*) native to Iowa; one (*C. coccinea*) native to rocky and mesic soils in the northeastern and north-central parts of the state, and the other (*C. sessiliflora*) more characteristic of loess and dry bluffs in the northern and western parts of the state. To a casual observer his "birdfoot violets" could be either true birdfoot violet (*Viola pedata*) or prairie violet (*V. pedatifida*). If he saw a landscape "thickened with birdfoot violets" near where he saw either species of Indian paintbrush, he is almost certainly describing prairie violet. Our knowledge of the conditions under which these species grow suggests the better match of Indian paintbrush with prairie violet. As for the "ordinary" goldenrod, Iowa has fourteen species of *Solidago*, all native (Eilers and Roosa 1994), but we interpret the poet to mean a common species such as tall goldenrod (*Solidago* *canadensis*).

As we read through the poem, we show photographs of specific plants and ask students to write and think about the accompanying list that we write on the board. We have no doubt that the poet saw "alien" quackgrass (*Elymus repens*), just one of many species of grasses imported into the Midwest for animal forage or erosion control. (It is a prominent constituent of Stephen's front yard in Pella, Iowa, where it proves difficult to mow.) However, quackgrass is actually native to North America; some populations are native to the New England states (Heiser 2003). Calling thistle "alien" may also be unjustified: the poet may be making an abstraction of the two genera *Cirsium* and *Carduus*. If he means *Carduus*, he saw nodding or musk thistle, *C. nutans*, which by law in drier Kansas is a noxious weed (Barkley 1983). But since the poet's imagery is seemingly of central and eastern Iowa, the likelihood that he would encounter the xerophilous nodding thistle is minimal. Instead he is probably witnessing a species of *Cirsium*. Iowa has eight native species in that genus, and while he probably did see either the non-native bull thistle (*C. vulgare*) or Canada thistle (*C. arvense*), he might also have seen the common and native field thistle (*C. discolor*).

As we continue to tease the identity of plants from the poet's specific names as well as his abstractions, we create longer lists on the board. The poet's "European morning glory that no state legislature could control" struck us as a bit odd in light of more aggressive, and truly difficult to control, introduced species such as leafy spurge (*Euphorbia esula*). Iowa does have two rarely encountered species of non-native morning glory (*Ipomea*); one, *I. coccinea*, is classed as a rare escape from cultivation, and the other, *I. hederacea*, is infrequent in disturbed areas (Eilers and Roosa 1994). Neither is considered weedy in neighboring Kansas (Barkley 1983). It is more likely that the poet saw native false bindweed (*Calystegia*), such as *C. sepium*, whose flowers closely resemble those of the non-native *Ipomea*. So the poem's ambiguity challenges us to investigate the flora while playing with the text.

With the line "We inherited pioneer grasses high as a prairie / schooner," the poem takes us to an early nineteenth-century description of the presettlement landscape. We highlight the historical description of prairie before the plow was invented in 1837; pervasive tallgrass indeed defined the land before the plow tore open the soil, as in the *Broken Ground* imagery of the title of Chenette's work. Hirsch's "fresh settlements / of bog flowers and refugees from the sea / coast marshes" is more problematic: the first botanists in North America encountered several species that we think of as prairie associates in eastern coastal marshes, but they are indigenous to the entire biogeographical region of eastern deciduous forest, including Iowa. Species such as nodding lady's tresses orchid (*Spiranthes cernua*), rattlesnakemaster (*Eryngium yuccifolium*), the moderately tall grasses switchgrass (*Panicum virgatum*) and little bluestem (*Schyzachrium scoparium*), and one that is indeed "as high as a prairie schooner," freshwater cordgrass (*Spartina pectinata*), are all part of the seacoast marsh or marsh-margin flora (Silberhorn 1976, Eleuteris 1990, Stuckey and Gould 2000).

So three of the dominant pioneer grasses are also known as marsh plants. We can't know what the poet meant by "refugees," but when we see an East Coast marsh firsthand, we are confronted by a plethora of graminoid genera such as sedges (*Carex*), flat sedges (*Cyperus*), spike rush (*Eleocharis*), many-spiked sedge (*Fimbristylis*), umbrella sedge (*Fuirena*), rush (*Juncus*), beak rush (*Rhyncospora*), and bulrush (*Scirpus*) (Eleuteris 1990). If these are all "refugees," their appearance in Iowa would be only as aliens. Instead we find that Iowa has 109 native *Carex*, 11 native *Cyperus*, 15 native *Juncus*, and 15 native Scirpus species. Iowa shares graminoid-species pairs with other East Coast marsh flora—*Fuirena simplex* for *F. scirpoidea* and *Rhyncospora capillacea* for *R. fascicularis*—as well as notable forb-species pairs; water hyssop (*Bacopa rotundifolia*), native to Iowa, is very similar to marsh hyssop (*B. monnieri*), and the Iowa native northern frog fruit (*Phyla lanceolata*) is similar to its seacoast-marsh relative common frog fruit (*P. nodiflora*). (Neither species of *Bacopa* is frequent, and *B. monnieri* is rare in East Coast marshes (Johnson 1991) and a species of special concern in Virginia.) Other East Coast marsh forbs are also native to Iowa, including marsh sage (*Teucrium canadense*), Virginia bugleweed (*Lycopus virginicus*), and thoroughwort (*Eupatorium serotinum*), as are still other graminoid species: blunthead spikerush (*Eleocharis obtusa*), and dwarf spikerush (*E. parvula*), Fimbristylis autumnalis, rush (*Juncus acuminatus*), toad rush (*J. bufonis*), soft rush (*J. effusus*), and giant bulrush (*Scirpus validus*) (Silberhorn 1976, Eleuteris 1990, Stuckey and Gould 2000, Radford et al. 1968, Eilers and Roosa 1994).

The poet appears to suggest that "bog flowers" in Iowa are postsettlement or recent additions, and he seems to confuse fen with bog; in the postglacial, mineral-rich soils encountered over much of Iowa, a true acidic bog would be rare. However, there is a small but botanically interesting guild of native bog plants in Iowa. Perhaps the most beautiful is the grass pink orchid, *Calopogon tuberosus*, which is found in places such as Muscatine County's Rhexia Pond, named for a hirsute, acidophilic Virginia plant, Virginia meadow beauty (*Rhexia virginica*); the pond is one of the rare boglike habitats in eastern Iowa (Prior 1991). The poet implies the more common wetland in Iowa, a fen, in his last stanza with his "grass of Parnassus."

The poet's ambiguous terminology provides an opportunity for us to tell audiences who aren't familiar with botanical terminology that some plants belong in Iowa even though their scientific names include, for example, *virginiana* or *virginica*—as in the "silky-leaved Virginia plant." When the poet suggests non-native status for such a plant, he invites clarification. We have already discussed two species of Virginia plants (those ending with *virginiana*, *-us* as well as *virginica*, *-us*). There are 33 such "Virginia plants" in Iowa, all native but five, and none are botanically silky (Harris and Harris 2001). In Iowa there is the silky aster, more commonly referred to as the western silvery or silky aster (*Symphyotrichum sericeum*), but *S. sericeum* is not found in Virginia. Virginia harbors the eastern silvery or silky aster (*S. concolor*) which is not found in Iowa (BONAP 2010, Fernald 1950, USDA PLANTS Database). Futhermore, *S. concolor* is not botanically silky. The difference between *S. sericeus* and *S. concolor* can be determined easily with a good hand lens. The only botanically silky-leaved plant native and common to both Virginia and Iowa is silky willow (*Salix sericea*).

The next two lines also provide a challenge for identification: "*and Texas marigolds, imported seeds and ornamentals, / weeds from the wasted villages of other continents.*" There are no marigolds (*Tagetes* spp.) in Iowa, either native or naturalized, and "Texas marigold" is one of the common names for *Tagetes lucida*—a tropical species from southern Mexico and Guatemala (Bailey, 1976). As such, it would hardly be able to naturalize in Iowa. Furthermore, true marigolds do not closely resemble the burr-marigolds (*Bidens*) native to Iowa forest edges and stream banks. The poet's choice of Texas marigold is odd because *T. lucida* has rather atypical entire leaves while the marigolds familiar to gardeners, such as French marigold (*T. patula*) and common marigold (*T. erecta*), all have pinnately dissected leaves reminiscent of the compound leaves of hickories and walnuts. So the most likely candidate for this Texas marigold is probably an aster or a goldenrod.

Throughout the poem, the poet glosses over the differences between innocuous introduced species such as moneywort (*Lysimachia nummularia*) or unicorn plant (*Proboscidea louisianica*), disturbance-loving species such as the sweet clovers (*Melilotus alba* and *M. officinalis*), and the truly ecosystem-altering aliens such as autumn olive (*Eleagnus umbellata*), Amur honeysuckle (*Lonicera maackii*), leafy spurge (*Euphorbia esula*), and silky bush clover (*Lespedeza cuneata*). One class of weeds from "wasted villages" would be culinary potherbs, or plants grown to flavor food. But it is folly to suggest that there is no difference between the innocuous potherb called common purslane (*Portulaca oleracea*) and highly invasive garlic mustard (*Alliaria petiolata*)! This reference allows us to present information about the invasive species that are threats to ecosystems in Iowa. We ask students to write about the characteristics of invasive species in the context of the plants we've listed on the board.

We then begin tracing the development of the people/plant equation, the exploration of which is our third goal with this poem. From the first line, "indigenous" is associated with people—childhood—as well as with, by implication, plants. The second stanza tightens this merger 11111of plants and people with "alien"; then the poet couples "alien" and "noxious" to describe some threatening plants as having a biological agenda that "no state legislation could control." The word "noxious" might have a hint of indignation if we begin to see invasive plants as second-class human citizens. Further, Hirsch metaphorically links plants and people by personifying certain plants as "refugees" from "wasted villages of other continents." He merges immigration issues with what is ostensibly a botanical discourse in speaking of "fresh settlements of bog flowers."

The fifth stanza, wherein "nature consists of immigrants and mongrels," makes explicit the conflation of people and plants. It ignores, however, some interesting ways in which plants travel with or without human help. The unicorn plant, mentioned above, was first collected in Emmet County, Iowa in 1883 (Eilers and Roosa). It may once have been dispersed by wooly mammoths and American mastodons in the Pleistocene before humans arrived in what are now the contiguous forty-eight states (Barlow 2000), but may have reached its present countrywide distribution (Brandenburg 2010) with the help of domestic livestock and railroads (Barlow 2000). Also, following the relaxation of fire management and conversion of great swaths of land to agriculture, preceding and increasing after statehood, many areas that once were prairie became forest (Scheese 1989). The forest plants followed a natural migration; the oaks and hickories gradually colonized Iowa through animal dispersal, while understory orchids such as oval lady's tresses (*Spiranthes ovalis*) and hooded lady's tresses (*S. ramanzoffiana*), with dustlike seed, relied on wind to establish themselves in Iowa. At approximately the same time, seeds of hooded lady's tresses were carried to western Britain, where they and the seeds of a handful of other eastern North American species established themselves in a guild that became known to British botanists as the "American element" (Sauer 1988, Good 1974). (Another plant migrant of an entirely different geography but the same mode of dispersion is in the African orchid genus *Oeceoclades*. One species, *O. maculata*, apparently through natural wind dispersal of seeds, spread from Africa to South America, where it was collected in the late ninteenth century. From the mid-1970s to mid-1980s this orchid colonized Trinidad, Puerto Rico, and South Florida, stopping only at the northern Florida frost line [Stearn 1988].)

Not at all from "wasted villages of other continents," the large-seeded species of pawpaw (*Asimina triloba*) found its way from the southern United States into the Midwest by prehistoric human agency. Because of its prized sweet fruits, pawpaw was transported by American Indians to the Flint Hills of Kansas and still grows by the remnants of native trails (Brooks and Johannes, 1990). The same agency may have bought pawpaw to Iowa.

The poem's concluding conflation of people and plants as immigrants seems both ambiguous and dangerous for ecosystems if he is suggesting that we accept all plants that arrive in Iowa: "*Nature consists of immigrants and mongrels: / you showed us how to prize coincidence and impurity / in wayward fields, the deserted and marginal . . .* " (the ellipsis points are the poet's). The linkage of "mongrel" with "immigrants" and "to prize" provides us with an opportunity to elaborate upon the poet's apparent championing of acceptance of both immigrant plants and people. Aggressive species such as purple loosestrife (*Lythrum salicaria*) and common reed (*Phalaris* *arundinacea*) outcompete natives and tend to homogenize or otherwise corrupt the ecosystems they infest (Vitousek 1990). Other invasives, like Amur honeysuckle and garlic mustard, further alter belowground ecosystem processes, such as mycorhizzal associations (Burke 2008, Czarapata 2005).

The poem then moves to an aesthetic and romantic tribute: "*I went down to the swamp to mourn for you, Amy, / and it was as if Providence led me to the place / where I stumbled upon yellow swamp betony.*" This verse is one of remembrance little associated with botany. The poet seems to link swamp betony, or lousewort (*Pedicularis lanceolata*), to some aspect of his poetic mentor. The movement to celebration in the last lines eclipses any concern about plant competition or problems caused by people: "*and pink foxglove mingled with something nameless /* (unfathomable the mystery before us, *you said*) */ and the shining, cup-flowered grass of Parnassus.*" The plant known as foxglove (*Digitalis purpurea*) is not found in Iowa. It is naturalized among ancient stone walls in Scotland, and has made its way to North America to naturalize in cold climates such as that of Newfoundland (Fernald 1950). On the other hand, there are two Iowa-native genera commonly called false foxgloves. The pink of the flower obviously eliminates the yellow-flowered hemiparasites of oaks in the genus *Aureolaria*, and instead implies the delicate, pink-flowered hemiparasites of grasses in the genus *Agalinis*. There are six species of *Agalinis* native to Iowa, one of which, *A. purpurea*, it shares with East Coast marshes.

Finally, out of the ambiguity in the stanza comes "the shining, cup-flowered grass of Parnassus." We again show photographs of flora that the poet might be seeing but conclude that he must be observing the only one of the three eastern species of *Parnassia* that is native to Iowa, thick-leaved grass of Parnassus (*Parnassia glauca*), found in fens of the lake region in Dickinson County (Herzberg and Pearson 2001). Like swamp lousewort, grass of Parnassus seems to trigger some association with the poet's muse, one not necessarily botanical. A reader who is a botanist might wonder why the poet would even mention "something nameless"—in botany there are no nameless plants, only those lacking a formal botanical description. Likewise, nothing before us is truly "unfathomable," just something awaiting curious and prepared investigators.

Using this poem, we try to convey to students that knowing names and understanding context can create the imagination and a sense of wonder that infuse Hirsch's poem. Writing about the ramifications of applying metaphors from a poem to a plant system helps students think about the significance of plants. Nature walks, as well as projects like removing aggressive invasives such as garlic mustard, Amur honeysuckle, and autumn olive at nearby parks and prairies, provide specific details and context when the students write in their journals about their experiences.

Ultimately, we are trying to support what John Cozza advocates in "Growing a Green Planet: The Future of Botany Teaching" (2009), as he attempts to help students become aware of and connect to plants, feel included as they explore botany and horticulture, and develop a sense of ownership of the plants in their lives. As they come to love plants, they will be motivated to help conserve them and their habitats by sharing the Earth in a sustainable way.

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PRAIRIE RESTORATION: RECONSTRUCTING PRAIRIE IN THE HEARTS AND MINDS OF THE HUMAN COMMUNITY

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Abstract: For the first time the University of Illinois has begun representing the region's prairie heritage in its campus design and landscaping with the recent establishment of a prairie garden at the College of Veterinary Medicine (CVM) at the University of Illinois at Urbana Champaign (UIUC). The garden marks the successful culmination of sustained efforts by prairie devotees to promote prairie awareness and conservation by growing community interest and promoting ecologically sound management regimens. The project's approval was also aided by escalating costs of fuel and other resource use, which helped motivate financially restricted administrative bodies. In 2008, the CVM submitted a proposal to the Student Sustainability Committee for development of a low-maintenance native planting at the CVM. The proposal was partially funded to promote engagement of volunteers. The plan for the garden was subsequently implemented with site preparation and planting of approximately 10,000 prairie seedlings, completed in the fall of 2009. This project was a result of collaborative efforts by multiple departments at UIUC, local conservation groups, environmentally conscious individuals from the university and surrounding communities, and successful engagement of newly inspired CVM faculty, staff, and student volunteers who will also be supporting the ongoing management of the site. Now almost a year old, the prairie garden features six native prairie grasses and 40 native prairie forbs in a prominent location on the veterinary medicine campus. The project has already made a positive impact by heightening awareness and influencing the perspectives of the CVM community and beyond regarding native plants and the natural history of the region.

IN THE HEARTS AND MINDS

Despite its status as the flagship public university of the "Prairie State," the University of Illinois has historically never represented the region's prairie heritage in its campus design and landscaping. The recent establishment of a prairie garden at the College of Veterinary Medicine (CVM) at the University of Illinois at Urbana Champaign (UIUC) marks the first prairie planting on the university campus. The CVM prairie garden also demonstrates the successful culmination of sustained efforts by prairie devotees to promote prairie awareness and conservation.

The present veterinary complex at UIUC is located southeast of the main campus and was established in 1971 with the construction of the small-animal clinic. Construction of the attached structure to house the large-animal clinic followed in 1976. In spite of encouragement to maintain the adjacent open area to the east, which featured a small waterway, as a natural area (prairie), the creek was filled and the site leveled for construction of the neighboring Veterinary Medical Basic Sciences Building (VMBSB) to complete the veterinary complex in 1982. Although members of the faculty, staff, and student body of the CVM petitioned for conservation measures and ecologically sound practices, their efforts to affect positively the operations at the CVM received little attention. Over the following years, some inroads were made in promoting ecologically sound perspectives, due to escalating costs of fuel and other resource use, in addition to the continuing efforts by dedicated supporters of natural areas for conservation and sustainability.

Ultimately, it was largely the matter of severe budgetary pressures that brought real change to bear. When the CVM, which was one of the two largest energy users on campus but had not previously been individually billed for energy expenditures, received a substantial assessment for energy use, coordinated efforts to reduce energy use began in earnest. The CVM established a Green Committee, charged with examining practices within the college and identifying areas for improvement or implementation of new projects. An early change was to establish a large no-mow zone in the previously lawn like acres around the CVM complex. This change in perspective at the CVM followed a broader initiative across campus which led to formation of the Student Sustainability Committee: this body is charged with allocating funding collected from assessment of student clean energy and sustainability fees to support projects for increasing use of clean energy technologies, reducing energy utilization, or otherwise promoting a sustainable campus environment (waste reduction, recycling, soil and water conservation, engaging the university community).

In 2008, the CVM Green Committee, in conjunction with individuals in Natural Resources and Environmental Sciences and the Illinois Natural History Survey, submitted a proposal to the Student Sustainability Committee for development of a low-maintenance native planting at the CVM. The project was awarded partial funding with a sustainability grant (partial rather than full funding was intended to encourage community outreach and engagement of volunteers). Anticipated savings in energy and labor over traditional landscaping, and the opportunity to establish the CVM as a leader in implementing sustainable practices, were key in attracting college administrators to the project.

The native garden plan was implemented with site preparation commencing in early 2009. Work began with herbicide application prior to the first large-scale volunteer event on May 30, 2009 'Mulch Day' for delivery and spreading of cypress mulch on the previously herbicided grounds. Volunteer recruitment for this event was very successful thanks to Media Communications services of the CVM, other relevant University departments, local conservation groups and gardeners (Natural History Survey, Grand Prairie Friends, Master Gardener, and Master Naturalist groups), and included people with diverse backgrounds and experience, and families as well as individuals. More than one hundred people volunteered. The CVM was well represented by faculty, staff, and students, who were becoming progressively more interested and involved in the project because of the local media exposure, group e-mail updates, and the involvement and obvious enthusiasm for the project demonstrated by some of their fellows. Mulch Day featured food and live music for volunteers to give the day an enjoyable carnival air, to reward their hard work, and to support their continued participation in prairie-garden project activities. Many returned to join new volunteers at the second major volunteer event in June 2009 'Planting Day' for planting native plant seedlings according to the carefully scripted site plan, which was presented and explained on site to volunteers. Teams of volunteers were assigned a specific plant species to work with; this, along with a descriptive introduction and overview of the project, fostered appreciation of the individual plant species and their interrelatedness within the prairie community, as well as the reasons behind the patterns in which they would plant the seedlings. A subsequent volunteer event in June 2010 completed the planting of approximately 10,000 prairie seedlings.

The CVM prairie garden materialized as a result of collaborative efforts by multiple departments at UIUC, local conservation groups, environmentally conscious individuals within the university and surrounding communities, and successful engagement of newly inspired CVM faculty, staff, and student volunteers who will also be supporting the ongoing management of the site (watering as needed, weeding, placing markers to identify the plants to visitors). Now almost a year old, the prairie garden features six native prairie grasses and 40 native prairie forbs in a prominent location on the veterinary medicine campus. The CVM prairie garden has positively influenced the perspectives of the CVM community and beyond. The project is also an excellent demonstration of the eventual success of sustained effort by a few individuals to have a positive impact upon practices and perspectives of the human community.

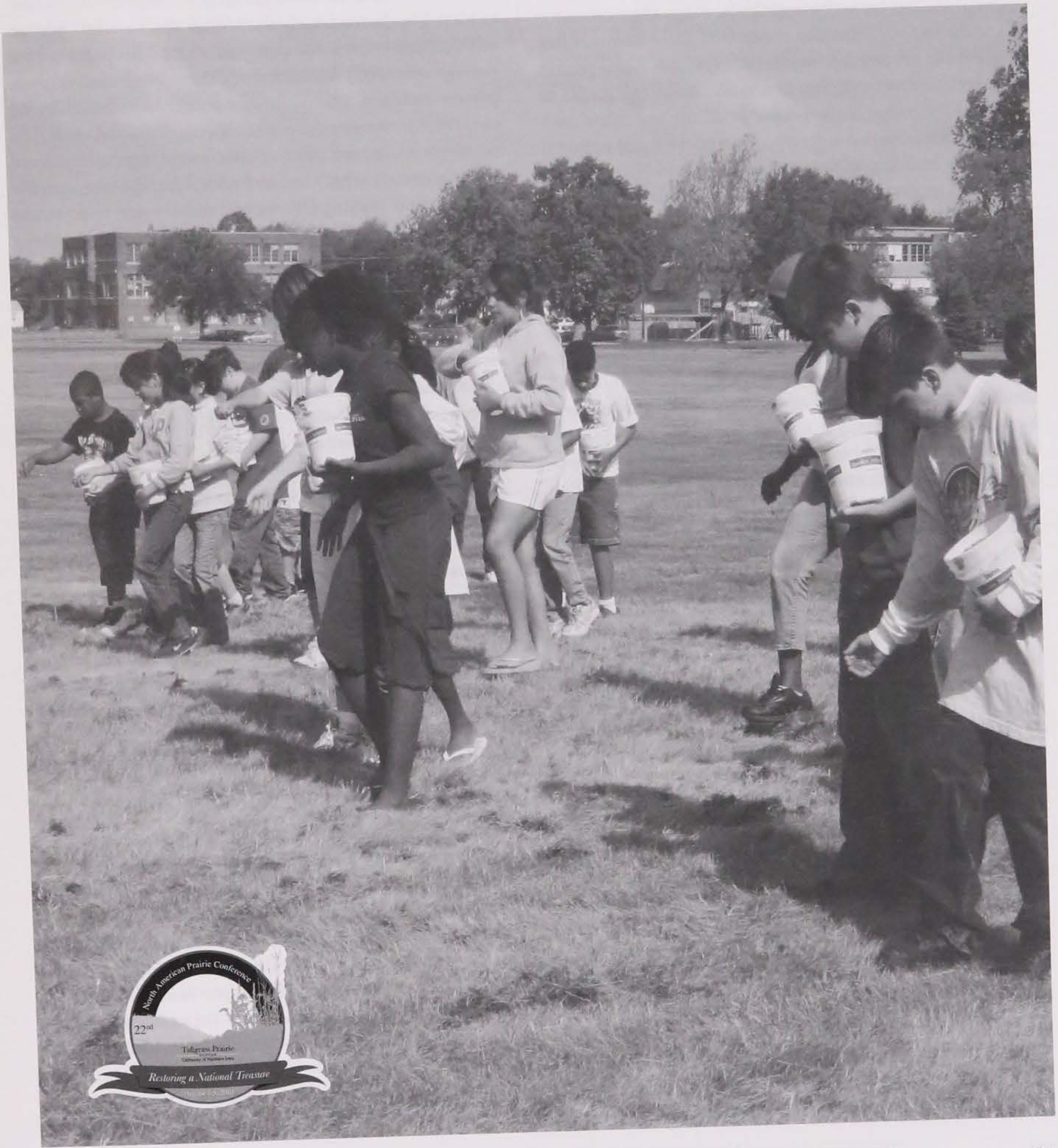
The new prairie garden includes a traffic island at the front entrance of the Veterinary Medicine Basic Sciences Building (VMBSB) and a larger area between the front entrance and the side entrance to the Vet Med Café. This position guarantees that the prairie plantings will be observed by virtually all visitors to the college, including alumni, continuing education attendees, families of students, administrators from other campus units, vendors, maintenance and repair personnel, clients of the veterinary hospital using the café, and those utilizing the Vet Med library, in addition to all in the CVM community. The larger planting surrounds a prominent twenty-three foot-long by twenty foot-high welded-steel sculpture by Illinois artist Richard Hunt titled *Growing in Illinois* (1982). This sculpture was important in initially inspiring the idea of a prairie garden project through its resemblance to prairie-fire flames, noted by a conservation-minded member of the college.

The objectives in establishing and maintaining the prairie garden include beautification of the grounds and education of visitors about the biologically rich tallgrass prairie. Many in the CVM and local community were not aware of the region's prairie heritage, what constitutes a prairie, or what native plants were historically indigenous to the area. The garden provides a prominent reminder of the region's natural history. Signage and plant markers in the garden and an informative mural in the main foyer of the VMBSB support the educational aspects of the project. The garden is intended to symbolize the college's "growing" commitment to environmental sustainability and protection of native species. The college has already realized the benefits of reduced mowing and habitat support for native insect and vertebrate species. With awareness initiated by this project, green initiatives have been gladly expanded at the CVM, including stepping forward to initiate a major project of comprehensive energy-use assessments and implementation of energy-saving strategies in all CVM facilities.

No one who is involved with the prairie garden project, or knows of its planning and installation, can walk by it without noticing that the *site* matters differently, that it has a great deal more significance than the adjacent mown area, even the difference is not fully understood, but the seed of awareness has been planted. Beyond the continuing educational impact of the site, it serves as an ongoing reminder of the experience to those who shared in its planning and birth, and that their efforts made a positive impact which will be amplified with each new generation of students and visitors to the CVM. The prairie garden will remain a part of their training experience at the CVM, one that will help to shape who they are and what they offer to the communities in which they continue their careers.

The benefits realized from this project extend beyond its impact on environmental awareness and appreciation of sustainable practices. In addition to allowing people to reconnect with the land, the prairie garden project fostered establishment of relationships among people.

The volunteers were a diverse group, drawn to the project with differing motivations and interests, resource and energy savings, low-maintenance or nontraditional landscaping, cost savings, interest in the prairie ecosystem, enjoyment of nature or gardening in general and contributed diverse experiences and perspectives so that all developed new views from their collaboration. In this project, it was also evident that, in attracting volunteers and nurturing growth in interest and level of enduring engagement, nothing is more compelling than witnessing the dedication and investment of personal time and physical effort by established supporters-let your enthusiasm and commitment show.



"TO LIVE ONLY IN BOOKS": READING AND WRITING RESTORATION

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Abstract: Restoration ecologists in the Midwest have devoted a great deal of attention to restoring physical prairie landscapes long compromised by the practices of industrial agriculture, the spread of urban and suburban sprawl, the removal of keystone animal species, and the incursion of invasive nonnative plant species. While this work is essential for restoring the prairie's health and prominence as a viable North American ecosystem, little attention has been paid to the equally important work of cultural prairie restoration: namely, the use of art, literature, and media to overturn the image of the prairie as a "lost landscape" and make its preservation an important part of the national environmental debate. Accounting first for those texts that depicted the prairie as a doomed ecosystem early in the nineteenth century, this paper argues that a reprioritization of the types of stories and narratives told about the prairie is in order and long overdue. Working within the parameters of narrative ethics, I use the Anishinaabe writer Gerald Vizenor's concept of "survivance," which emphasizes continuity as a form of cultural resistance, as a model for discussing the prairie not as a "lost landscape," but instead as something both ecologically and culturally viable, alive, and thriving.

Key Words/Search Terms: ecological and cultural restoration, narrative ethics, survivance

"TO LIVE ONLY IN BOOKS"

The quotation in the title of this paper is taken from George Catlin's text *Letters and Notes on the Manners, Customs, and Conditions of the North American Indian*, published in 1841 but based on several excursions Catlin made to the mixed- and shortgrass prairies west of the Missouri River throughout the early and mid 1830s. Ostensibly about the indigenous human communities of this region, Catlin's work is also one of the early touchstones of Euro-American prairie literature. The five words I have chosen here—"to live only in books"—quite adeptly sum up the attitude held by more than a few visitors to the prairie, certainly among them those who stayed and settled but especially those, like Catlin, who returned home to the forested East. To be precise, this line from Catlin's text was written specifically about bison; a fuller segment will contextualize the words and illustrate a phenomenon characteristic of literature from this time and place, namely the trope of the vanishing prairie:

> These noble animals [bison] of the ox species, and which have been so well described in our books on Natural History, are a subject of curious interest and great importance in this vast wilderness; rendered peculiarly so at this time, like the history of the poor savage; and from the same consideration, that they are rapidly wasting away at the approach of civilized man—and like him and his character, in a very few years, to live only in books or on canvass. (247)

First, I've chosen largely to overlook the "canvass" portion of this quote because the argument that I want to make here deals mainly with print culture and the written word; far more people wrote about the prairie than painted it, though in the case of Catlin the two cannot be separated. Furthermore, I do not think it is taking this quotation out of context to expand Catlin's sentiment here to how he felt about the larger prairie ecosystem. If nothing else, the melancholy tone is consistent with comparable reflections throughout his writing, where a sense of transience pervades. Finally, Catlin's comparison of bison to the "poor savage" invokes infinite parallels between the decline of plant and animal communities throughout the prairie and governmentally sanctioned removal policies for the prairie's indigenous human communities. Taken together, what might be most notable about Catlin's words is the confidence and certainty with which he writes them, not to mention the ease with which he distinguishes himself from what he has witnessed on the prairie. Clearly a "civilized man" himself, Catlin keeps a safe distance from his subjects, whom he envisions as being under constant threat of disappearing.

That Catlin and many like-minded writers were, by and large, accurate in their assumptions and predictions about the fate of the prairie does not make this phenomenon of mid and late nineteenth-century American literature any less troublesome. To begin, one might take issue with how quickly the prairie—on all levels, including plants, animals, and people—was literally written off by authors of early texts set within this region, including James Fenimore Cooper, Washington Irving, and Francis Parkman. The consensus view of these writers, each of whom published his definitive work on the prairie before 1850, not only held up this region as a "wasteland" and part of the "Great American Desert," but also boldly envisioned grassland being converted to agriculture, animals being hunted to near extinction, and native populations being overrun by Euro-American settlers—all in less than half a century of meaningful interaction with the prairie. Fortunately, for those who both envisioned the prairie in such dire straits and had aspirations to authorship, the settlement of the prairies coincided with the vast proliferation of book culture in America. Thus it was easy to envision a solution to the problem of the prairie's certain disappearance: what might be lost forever as a result of Euro-American settlement and habitation could nonetheless be perpetuated into eternity thanks to the written (and printed) word. This is not meant to be a cynical overreading of authors actively ensuring a subject and audience by having a hand in the destruction of entire ecosystems and human communities; instead, I mean to suggest a more subtle, perhaps even subliminal, effect of such literature, wherein the idea or image of the vanishing prairie became so commonplace so quickly that texts—not just longer works by the aforementioned authors, but also pamphlets, periodicals, maps, almanacs, paintings, and so on—prophesying its demise were in a certain sense self-fulfilling. Cultural theorist and postcolonial scholar Edward Said, in his seminal work Orientalism (1979), refers to this phenomenon as a "textual attitude," wherein a person "appl[ies] what one learns out of a book literally to reality" (93). In other words, half the argument that I want to make here is that texts—or narratives, more generally—especially those written by and produced for a growing mainstream American readership, might justly be placed alongside the plow and the rifle in looking for the instruments of the prairie's increasing maltreatment throughout the nineteenth century.

The importance of narrative and textual production to the exploration and settlement of the prairies—really the whole West—is well illustrated by three works written around the same time as Catlin's: Washington Irving's *A Tour on the Prairies*, Astoria, and *The Adventures of Captain Bonneville*, published in 1835, 1836, and 1837, respectively. Each work opens with a self-conscious reflection by Irving on how he came to write that text. *A Tour on the Prairies*, a text that Catlin knew and acknowledges in the opening of his work, begins with Irving reluctantly undertaking the task of writing about his recent excursion to the mixed- and shortgrass prairies of Kansas and Oklahoma: "Having, since my return to the United States, made a wide and varied tour, for the gratification of my curiosity, it has been supposed that I did it for the purpose of writing a book; and it has more than once been intimated in the papers, that such a work was actually in the press, containing scenes and sketches of the Far West" (11). The two paragraphs following this comment show Irving feigning surprise at such "expectations" from his readership, as they are something he has commonly met with "repugnance." Yet he then goes on to state how he wishes to satisfy the "desire of the public" and has "plucked a few leaves out of my memorandum book, containing a few months' foray beyond the outposts of human habitation, into the wilderness of the Far West" (11-12). The mere existence of this "memorandum book" should be enough to insist that Irving's travel and tourism is explicitly textual in its origin and purpose as is his awareness of the supposed demands being placed on him as a literary celebrity.

The Adventures of Captain Bonneville opens with an equally straightforward examination of the work of writing that went into the text's production. It also contains one of my favorite images from all of the literature produced during this era, as it so clearly details Irving's fascination with textuality, with narrative, and with the means through which books come into being. The scene in question comes near the end of the "Introductory Notice" and details Irving's encounter with the eponymous captain in Washington, D.C., just after he has returned from a four-year exploration of the Midwest, Rockies, and West Coast:

> I found him quartered with a worthy brother in arms, a major in the army. He was writing at a table, covered with maps and papers, in the centre of a large barrack room.... In a word, the captain was beguiling the tediousness of attendance at court, by an attempt at authorship; and was rewriting and extending his travelling notes, and making maps of the regions he explored. As he sat at the table in this curious apartment, with his high bald head of somewhat foreign cast, he reminded me of some of those antique pictures of authors that I have seen in old Spanish volumes.... The result of his labors was a mass of manuscript, which he subsequently put at my disposal, to fit for publication and bring it before the world. (632)

The brilliance of Irving's portrayal here is his decision to place this image of Bonneville, wrapped up in the act of constructing the text the reader has yet to encounter, *before* any of the actual narrative takes place. In other words, it is the book itself that the reader already has in hand that matters most, more so than the actual events that unfold within it. What's more, Irving informs the reader that he has seen to it himself that what follows is in fact readable: Bonneville's notes only form the "staple" of the text, as Irving has "given it a tone and coloring drawn from my own observation, during an excursion into the Indian country beyond the bounds of civilization"; this is clearly a direct reference to *A Tour on the Prairies*, the publication of which has made Irving an authority on travel writing and thus fit to judge what portions of Bonneville's notes are and are not worthy of mainstream publication. Bonneville might know more about the prairie, mountain, and coastal environments he visited firsthand, but Irving's expertise as writer—and the "textual attitude" he has fostered in his readers—trumps all.

Of course, the actual narratives that unfold within *A Tour on the Prairies* and *The Adventures of Captain Bonneville* are filled with passages that perpetuate the trope of the vanishing prairie. The self-conscious textuality of each work is therefore similar to Catlin's writing, since the job of creating the text must be undertaken—apart from the exasperating expectations heaped upon Irving by his own readership—in order to preserve for the ages what is already on its way to disappearing. Addressing a similar phenomenon in the visual arts of this era, Joni Kinsey has coined the terms "aesthetics of absence" and "aesthetics of plainness" to describe the approach painters and photographers, especially those working in the nineteenth century, took to portraying prairie landscapes (Kinsey et al. 1999). Lacking the trees, mountains, and dominant water features of the East, Kinsey argues, these artists had to adapt to a landscape defined by its absences: "Almost all…artists who encountered the prairies either avoided depicting them or filled their compositions with something, anything, to compensate for the aesthetic void" (19). Some literary critics have argued that a similar sort of aesthetic exists in early writing about the prairie. Such a comparison is certainly apt in some cases—there's no denying the repeated motif of the prairie's emptiness or plainness in many of these texts—but Kinsey's terms for the visual arts don't fully account for the image of the vanishing prairie found in the literature of most interest here. Instead, a more fitting concept is Paul Virilio's term "aesthetics of disappearance," which argues that the art of the modern and industrial West tends to depict the "world only as it is in the process of disappearing" (Snyder 129). In other words, writers like Catlin and Irving, among countless other early prairie writers caught up in the machine of Manifest Destiny, can be seen depicting a landscape endlessly and inevitably slipping away, and so perpetuate for their readership this vanishing prairie motif.

I'd like now to explore the inverse of this argument—namely, that texts and narratives need to play an equally important role in efforts to restore the prairie. There is no denying that the front line of prairie restoration is the physical, hands-on work being done in the field on one-acre remnants, thousand-acre revitalized farmscapes, and everything in between. As someone just beginning to get involved with this kind of work, I know already, firsthand, that it's nearly impossible for any prairie landscape to thrive in an age of invasive species and suburban sprawl without the near-constant effort of prescribed burns, species removal, and seed stratification. The ecological level of prairie restoration is inarguably the most essential for its success. However, I can also attest to the importance and necessity of the cultural level of prairie restoration. For one thing, I have lived my whole life in the American Midwest, descended from German and Danish immigrants who settled in western Iowa, primarily to farm. Yet the word "prairie" was virtually meaningless to me until I was in my mid-twenties and just beginning work on my PhD. That I spent the majority of my life oblivious to the importance of native prairie ecosystems to the health of the places where I was living is, fundamentally, a cultural issue. Prairies were absent from the books I read, the subjects I studied in school, the television I watched, the political debates I followed; in other words, prairies were absent from almost every facet of my day-to-day life. If truly effective prairie restoration is going to take hold in the mainstream, a large part of that work must be done by writers, scholars, teachers, and policy-makers operating outside of purely ecological efforts to revitalize prairie ecosystems.

Though not yet a part of mainstream political or critical environmental discourse, the fate of the prairie does matter beyond the scope of a few preserved remnants or revitalized fields. Ecologically speaking, we are all aware of the "downstream" benefits of a thriving native ecosystem in the American Midwest: diverse plant and animal life, healthier soils, cleaner water, and so on. Similarly, the acts of reading and writing are not ends in themselves; they too have far-reaching consequences. In his work *Narrative Ethics* (1995), Adam Zachary Newton works to show that texts and narratives are implicated in much more than just reading and writing—that they too have important downstream effects. To this end, he offers a different way to think about how/why we read and write, arguing that we ought to approach "narrative as claim, as risk, as responsibility, as gift, as price. Above all, as an ethics, narrative is performance or act—purgative…malignant…historically recuperative…erotic and redemptive...obsessive and coercive" (7). We have already seen those narratives Newton would describe as "purgative" and "malignant," which envisioned the prairie as a thing in the way of Western progress and civilization, and so inevitably on its way out. On the other hand, Newton points out that narratives can also be "recuperative" and "redemptive." These are the narratives upon which prairie studies now need to focus. Texts like James Welch's *Fools Crow* (1986), N. Scott Momaday's The Way to Rainy Mountain (1969), and the poetry of Ted Kooser and Mary Oliver, among many others, offer examples of narratives that have taken on the responsibility of restoring a sense of life, dignity, and potentiality to the prairie. In these works the prairie is not a dying or vanishing ecosystem, but one that thrives upon the possibilities of life. The Anishinaabe writer Gerald Vizenor frequently employs the word "survivance" to describe his efforts to counter the images of oppression, defeat, and "victimry" that dominate mainstream academic and cultural discourses on the settlement of North America. He defines survivance as "more than survival, more than endurance or mere response; the stories of survivance are an active presence" (15). With this in mind, I believe the work of reading and writing the prairie has the chance to replace the "lost landscape" with the idea of "prairie survivance," making continuity and perpetuity the prairie's lasting image.

Turning, then, to a few examples that offer counter readings of the prairie to those posed by nineteenth-century writers like Catlin and Irving, I will conclude on a more hopeful note. An example of such work is Aldo Leopold's *A Sand County Almanac* (1949), perhaps the most recognizable environmental text to emerge from the prairie states. Leopold was fully cognizant of the lack of attention to the problems facing the ecosystems of his native Midwest. One of the most famous passages in Leopold's iconic work comes in the section entitled "Prairie Birthday," in which he simultaneously celebrates the life of the prairie and bemoans its steady decline. Describing an "ordinary graveyard" he routinely passes when driving to his farm, Leopold writes:

> [The graveyard] is extraordinary only in being triangular instead of square, and in harboring, within the sharp angle of its fence, a pin-point remnant of the native prairie on which the graveyard was established in the 1840's. Heretofore unreachable by scythe or mower, this yard-square relic of original Wisconsin gives birth, each July, to a man-high stalk of compass plant or cutleaf Silphium, spangled with saucer-sized yellow blooms resembling sunflowers. It is the sole remnant of this plant along this highway, and perhaps the sole remnant in the western half of our county. What a thousand acres of Silphiums looked like when they tickled the bellies of the buffalo is a question never again to be answered, and perhaps not even asked (44-45).

A Sand County Almanac is itself about a wide range of ecological issues facing a multiplicity of ecosystems, but this passage about the prairie remains one of the text's most moving. Though first and foremost a celebration of the compass plant, and the prairie more generally, Leopold is also well aware of the threat of the vanishing prairie, both in *A Sand County Almanac* and his other writing. For example, in an unpublished manuscript posthumously titled "Prairie: The Forgotten Flora" (1999), Leopold—writing in the context of World War II—ironically observes that

> Half of southern Wisconsin was once prairie. Now that we must fight to maintain our national existence, one might presuppose a universal interest in the raw materials of and on which states were built. Yet I have never encountered, in any school or college textbook, an adequate description of prairie. Prairie, to most Americans, is a flat place once dotted with covered wagons (162).

The further irony of this statement corresponds with Catlin's words that open this paper: namely, by Leopold's time it appears that even the prairie's existence in books was in question. The heart of this problem—namely, the inexcusable underexposure of the prairie in modern environmental discourse—remains true today, though the work of Leopold and some of his contemporaries has given new life and purpose to the prairie's place in books.

Taking inspiration from Leopold, a number of contemporary writers engage the prairie in the spirit—if not directly invoking the name—of survivance. For example, Annie Proulx's novel *That Old Ace in the Hole* (2002) directly addresses the historical degradation of the prairie but also offers glimpses of its potentiality for recovery. Set in the shortgrass prairies of the Texas Panhandle—not too far from the region visited by Irving in *A Tour on the Prairies*—the novel opens with a compelling description of the region as Bob Dollar, the novel's protagonist, first makes his way onto the prairie:

> He knew he was on prairie, what had once been part of the enormous North American grassland extending from Canada to Mexico, showing its thousand faces to successions of travelers who described it in contradictory ways: under gritty spring wind the grass blew sidewise, figured with bluets and anemones, pussytoes and Johnny-jump-ups, alive with birds and antelope; in midsummer, away from the overgrazed trail margins, they traveled through groin-high grass rolling in waves; those on the trail in late summer saw dry, useless desert studded with horse-crippling cactus.... Bob Dollar had no idea he was driving into a region of immeasurable natural complexity that some believe abused beyond saving. (3)

As the novel progresses, Proulx ultimately allows the voice of those who do not view the prairie as "abused beyond saving" to dominate, in particular an old windmill man and a "monk-cowboy" with plans to bring the bison back to the panhandle. Though images of the degraded prairie find their way into the novel—especially through scenes at hog-confinement facilities and large cattle ranches—Proulx's work is ultimately about the possibility that restoration can take place. It is the prairie's "immeasurable natural complexity" that Proulx prioritizes in the end, not the two centuries of abuse chronicled in the works of Irving, Catlin, and others. Thus, from the perspective of narrative ethics, *That Old Ace in the Hole* serves as a clear example of contemporary prairie writing that advocates both ecological and cultural prairie restoration.

Perhaps more familiar are those works that might be called "memoiristic restoration guides," in which the author narrates firsthand experience in restoring prairie ecosystems. A bevy of these texts have emerged in the last decade or so, the majority of them adhering to a Leopoldian "land ethic" and promoting ecological restoration as an important component of environmental advocacy. But because these guides use narrative as a means for documenting their hands-on ecological fieldwork, they also serve as excellent examples of cultural prairie restoration as well. Taking one recent example, Steven I. Apfelbaum's *Nature's Second Chance: Restoring the Ecology of Stone Prairie Farm* (2009), it is possible to illustrate the means through which this sort of dual restoration can take place. Apfelbaum reflects on the challenges of ecological restoration in a manner applicable to the cultural restoration advocated for in this paper, writing: "The restorer empathizes with the land, identifies with its struggle, and heads straight into the breach.... We humans must strive to resuscitate this ecosystem or suffer the same fate as those species and habitats we have already destroyed or imperiled" (108-109). This notion of "empathy with the land" is precisely what is missing from the earliest accounts of Euro-American interaction with the prairie. The development of a sense of empathy—with humans or the environment—also happens to be the strongest indicator of a successful narrative and is thus a key component of narrative ethics.

Finally, empathy is one of the primary components of Leopold's land ethic, therefore as a key link between the text, the reader, and the land. As a reader works through a text like Apfelbaum's, taking in his firsthand accounts of restoring his degraded farmscape in southern Wisconsin, the end result is empathy not just for the human labor exerted by Apfelbaum, but equally for the prairie's recovery as well. Like Proulx's novel, Apfelbaum's memoir—and many others like it—ascribe to what Vizenor calls the "aesthetics of survivance," presenting an image of the prairie as recoverable, vibrant, and alive.

To conclude, it seems fitting to quote the work of the keynote speaker of this conference, since the writing of John Price works in much the same way as the texts I've noted here. The premise of his first book, *Not Just Any Land* (2004), arises from a concern similar to that of this paper: namely, the struggle to overcome the absence of the prairie in both the ecology and the culture of the American Midwest. To confront the former, he visits places like Buffalo Gap National Grassland in South Dakota and Neil Smith Wildlife Refuge in Iowa. To confront the latter, he undertakes personal encounters with living prairie writers like William Least Heat-Moon, Linda Hasselstrom, and Mary Swander. Some of the conclusions he draws from these interactions leave the reader with a sense of hope or optimism that the prairie has not been forsaken by all, including a number of prominent writers. To this end, Price writes that

> in a region where the land is often seen as flat and uninteresting, its future already written in the furrows and pastures it has become, I held before me...the possibility that the pieces might be made whole, a prairie place once again. Also, the possibility that however broken, this land still had the power to inspire, to surprise, to create and demand a rebirth of the self (24).

For some, such inspiration might come from helping with a prescribed burn; that the inspiration for others might come from reading works like *Not Just Any Land* or *A Sand County Almanac* is an indication of the importance of cultural restoration and the need to prioritize the prairie in our national environmental discourse.

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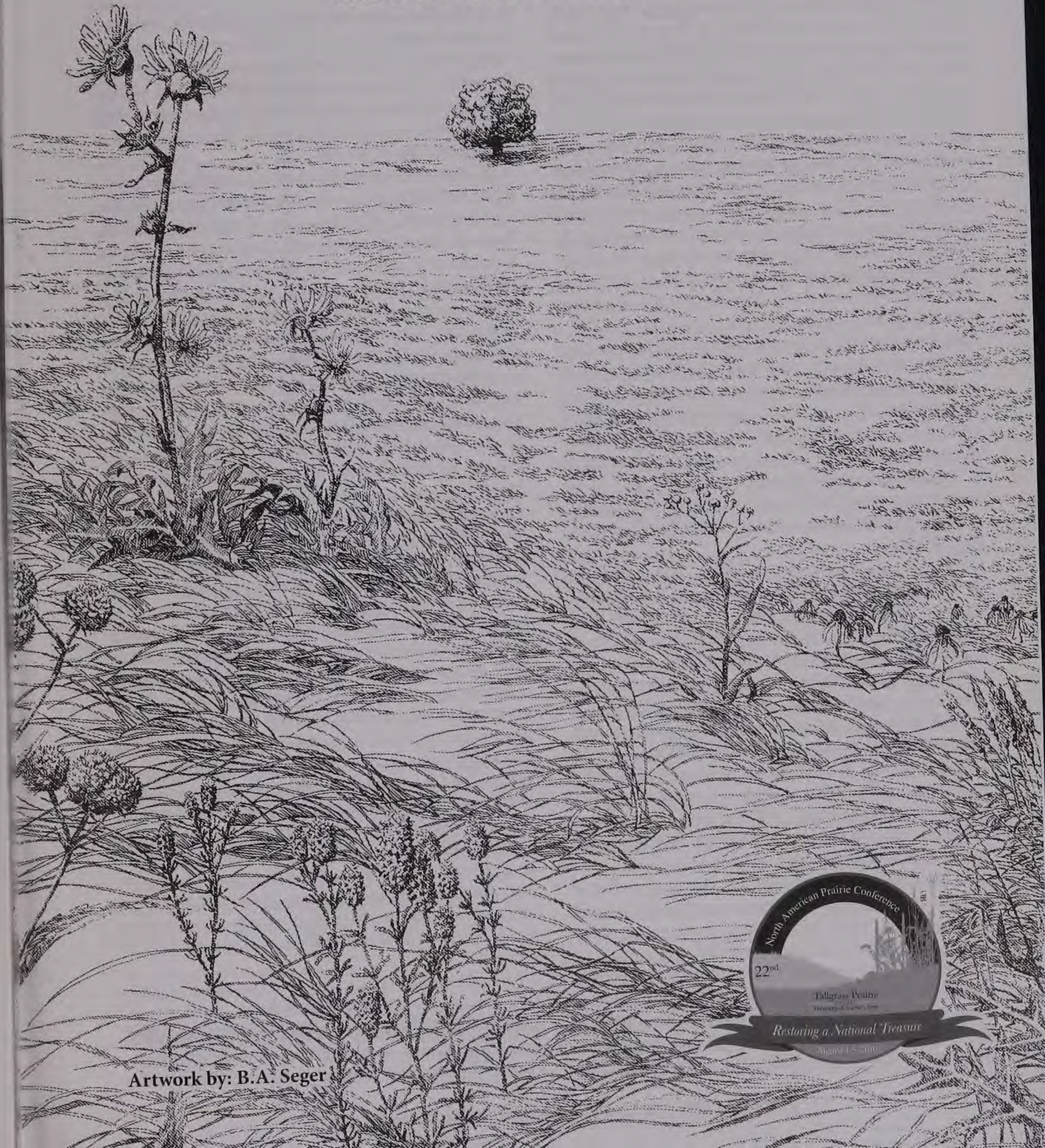

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RESTORING A NATIONAL TREASURE: INVESTMENT FOR THE FUTURE

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Tallgrass prairie is the most decimated ecosystem in continental North America. Less than 2% of the original tallgrass prairie remains, most on untillable land such as the shallow, rocky soils of the Flint Hills in eastern Kansas. Many states with large stretches of tillable soils, like Iowa and Illinois, have lost more than 99.9% of their prairie ecosystem. In these states, prairie exists as isolated remnants, tiny islands awash in an agricultural sea, scattered in railroad rights-of-way, roadside ditches, old settler cemeteries, rocky outcrops, and out-of-the-way places. Occasionally, larger pieces persist because they were retained for prairie hay. Consequently, much of the landscape has lost its capacity for water adsorption and infiltration, soil formation, wildlife habitat development, and ability to harbor a diversity of organisms or support indigenous cultures.

When the settlers first encountered the tallgrass prairie in the 1830s, they thought it was so vast that it could never be subdued (Edwards 1948). However, conversion to cropland was rapid and extensive. The changes in Iowa were a microcosm of the overall tallgrass prairie conversion. The transition of Iowa prairie landscape to agriculture occurred in seventy years, in the span of one lifetime (Smith 1992). When Iowa was considered settled, near the end of the nineteenth century, 97% of its land had been put to the plow (Harter and Stewart 1930).

Two Iowa botanists, Thomas Macbride and Bohumil Shimek, were aware of what was happening and expressed concern. Macbride (1902) observed, "The prairies are plowed almost to the last acre." A few years later, with almost 98% of Iowa prairie gone, Shimek (1911) wrote of how rapidly the tallgrass prairie was being converted to cropland. Their views about the future of prairie differed, however, in the mid-1920s when less than 1% of Iowa prairie remained. Shimek was surprisingly optimistic about the capacity of the tallgrass prairie to recover. He noted that continuous cultivation was required to keep prairie plants out of cropland, and opined that the widely scattered remnants of prairie contained sufficient numbers of prairie plants to reestablish and spread onto suitable sites (Shimek 1925). At the same time, Macbride (1925) was wondering, "How shall one bring back the upland with its velvet grasses in rippling changing shades . . . or picture in the lowlands the wide mantle of the sedges, swarthy green with lustrous sheen?"

Although Iowa was at the forefront of forest preservation near the end of the nineteenth century (Macbride 1895), prairie preservation was not a priority. Hayden (1919) was the first to suggest preserving small prairie plots in each county for educational purposes. The *Iowa Twenty-Five-Year Conservation Plan* (Crane and Olcott 1933) acknowledged a need for preservation of prairie as less than 1% remained in the state. When the first prairie preserve was dedicated fourteen years later (Parker 1947), it had further dwindled to about 0.3%. Today less than 0.1% of Iowa tallgrass persists in small, isolated remnants scattered across an agriculturally dominated landscape (Smith 1998).

When prairie preservation began in Iowa near the middle of the twentieth century, the Clementsian concept of "climax community" (Clements 1928) strongly influenced the thinking regarding community structure and dynamics. Scientists were slow to realize that we had so modified the landscape that tallgrass prairie could not recover without human assistance. Conservationists believed that preservation alone was sufficient to recover and retain tallgrass prairie. Ecologists assumed that a plant community would reestablish through succession and return to its prior climax condition, i.e., tallgrass prairie. Consequently, little thought was given to restoration or management of prairie remnants. Land managers protected prairie preserves from human impact and left them alone so they could return to high-quality prairie.

The continued deterioration of prairie remnants in the latter part of the twentieth century was convincing evidence that ongoing human management was needed. By 1970, it was obvious that most prairie remnants were being degraded by invasion of woody species and nonnative weeds, and had lost conservative prairie species. Contributing to this degradation were increases in invasive species, herbicide drift, sedimentation and nutrient overload from adjoining cropfields, genetic isolation, and nitrogen levels in the atmosphere. Smaller isolated remnants were further diminished as species critical to the prairie community declined and some disappeared (Leach and Givnish 1996). Extirpated species could not be replaced because extensive cultivation of surrounding areas had eliminated proximal seed sources. Chances for survival of these tallgrass prairie remnants were minimal, and recovery unlikely. Unfortunately, the confluence of land, climate, biota, and native American culture that created the tallgrass prairie was gone and would never occur again (Simpson 2008). If we were to retain tallgrass prairie, humans would have to assume the responsibility for doing so.

I consider prairie to be a part of our biological and cultural heritage, a national treasure that we should cherish, nurture and understand. Prairies must be walked upon and experienced for one to really know and understand them. Therefore, examples of tallgrass prairie should be available and accessible to society. However, the number of preserved remnants is not sufficient to accommodate the numbers of people needing to experience prairies. Prairie restoration activities must be increased and pursued more vigorously. Existing prairies remnants need to be upgraded and managed and new prairies reconstructed.

Prairie restoration has been used to describe a variety of practices. While the term itself is not contentious, it needs some clarification and more consistent use. Kline and Howell (1987) distinguished two basic approaches to prairie restoration. One approach involved creating prairie on a site with no existing prairie species, and the other approach consisted of upgrading a degraded remnant containing relict prairie species. They clearly delineated the two processes, but didn't distinguish them with names. For many years, restoration was used to refer to both approaches, with no attempt to differentiate between the two. Many continue to use it in the same general sense.

To distinguish the two approaches, we began to use the term *reconstruction* in the late 1980s when referring to prairie establishment on sites with no prairie species. As no relict prairie plants are present when the project is initiated, the reconstructed prairie is the result of the addition of prairie seeds and/or seedling transplants to the site. Use of the term "reconstruction" had become generally accepted for this process when Carl Kurtz (2001) published, *A Practical Guide to Prairie Reconstruction*.

Improving a degraded remnant involves the use of specific practices designed to reduce the degradation, improve the existing prairie, remove invasive species, and possibly return extirpated species to the site. Selecting an acceptable term for this process of upgrading disturbed or degraded remnants was more difficult. As mentioned previously, some continued to use prairie restoration for both that approach and the overall process. In one section of *The Tallgrass Restoration Handbook* (Packard and Mutel 1997), the term "rehabilitation" is used for this process, while in another section it is called "remnant restoration." However, rehabilitation commonly refers to reparation of ecosystem processes without particular attention to species composition and community structure. Remnant restoration is an appropriate descriptor for upgrading prairie remnants even though it may initially seem a bit awkward. In *The Tallgrass Prairie Center's Guide to Prairie Restoration in the Upper Midwest* (Smith et. al 2010) we used *remnant restoration* to delineate the process of enhancing prairie on a degraded site containing relict prairie plants (Table 1).

Table 1. Types of Prairie Restoration.

PRAIRIE RESTORATION	
RECONSTRUCTION	**REMNANT RESTORATION**
No prairie species present-Cropland	Degraded remnant with prairie species

The successes of early workers in prairie reconstruction set a high bar for a quality restoration. Far-sighted individuals like Norman Fassett, Aldo Leopold, Ted Sperry, and later John Curtis laid the foundation for prairie restoration at the University of Wisconsin Arboretum. From 1936 to 1941, Sperry directed Civilian Conservation Corps crews that planted and transplanted plugs from remnants in what would later be known as the Curtis Prairie. Later, Curtis added monitoring, restoration, and management research studies to the prairie reconstruction project. An important contribution of these studies was the determination that fire was an important tool in restoration and management of prairie communities (Jordan 1982, Anderson 2009). At about the same time period, a prairie reconstruction project at Homestead National Monument was initiated in 1939 to control sheet, rill, and gully erosion. Sod was transplanted from a local prairie and interseeded with a mixture of grasses from a local prairie five miles away. Later the priority of soil stabilization changed to more natural management of vegetation.

Henry Greene designed and almost single handedly completed the second University of Wisconsin Arboretum project from 1945 to 1953. With a thorough knowledge of the soil and moisture requirements for each species and a knack for knowing where to plant them, he established 200 species on a 50-acre (20-hectare) sandy soil site using seeds, greenhouse-grown seedlings, and transplants. Greene Prairie gained recognition as one of the most successful restored prairies anywhere, with diversity comparable to good-quality native remnants (Anderson 2009). When I visited the Greene Prairie in 1972 and later in the 1980s, I felt that I was walking in a native prairie - indeed a treasure.

A prairie reconstruction project initiated in April 1955 at the Knox College Green Oaks Field Study Center was inspired by the two previous projects. After a visit to the UW-Madison arboretum, George Wade and Paul Shepard used seed from railroad rights-of-way, pioneer cemeteries, and the University of Wisconsin Arboretum to plant a prairie (Allison 2002). In the next eight years, Shepard expanded this planting and initiated a second reconstruction project at the center. Shepard's replacement, Peter Schramm, expanded the second planting in 1966 and initiated a third. He used the latter two plantings as laboratories to test reconstruction techniques and various seeding mixtures (Allison 2002).

Ray Schulenberg initiated a project in 1962 that was to become a sterling prairie reconstruction on 55 acres of long-cultivated farmland at the Morton Arboretum, Lisle, Illinois. He and summer-intern high school students collected seed from local remnants and transplanted the resulting seedlings into the site. They also transplanted sod from relict prairie spots and spread seed gathered from the Gensburg-Markham Prairie. During the growing season, they carefully hand-weeded the new plantings. Most consider this project to be an excellent example of a self-sustaining tallgrass prairie reconstruction. However, Schulenberg referred to it as a "planting of prairie plants," as a reminder that although it was a successful reconstruction, it didn't include all the soil microorganisms, insects, mites, fungi, and bacteria of the original prairie (Dredze 1998),

The success of these projects generated interest in prairies and prairie reconstruction, an interest that blossomed in the late 1960s and early 1970s. Prairie reconstructions flourished and prairie plants began to be used in roadside plantings for weed control and to landscape yards and corporate grounds, to renovate pastures, and to develop outdoor laboratories for schools (Smith and Christiansen 1982). Roger Landers and Paul Christiansen led the way in initiating reconstruction of Iowa's prairie treasure. Schramm (1970) convened the first Midwest Prairie Conference in 1968 in response to the interest in and enthusiasm for prairie restoration. Thereafter, biennial conferences were excellent venues in which to exchange prairie restoration information. Schramm, Schulenberg, and Christiansen were major sources of information at the early conferences. Anderson (2009) points out that the methods and procedures outlined by Schramm (1970) at the first conference became the standard guide for most prairie reconstructions of the 1970s and 1980s. The method consisted of intensive tilling of the site, including fall plowing, disking and harrowing, to reduce weed competition and create a smooth, firm surface. He recommended leaving the site fallow for one year and keeping it weed-free. Then cold-moist-stratified native seed was broadcast or drilled in late spring or early summer (Anderson 2009). The only modification of this method was the addition of the use of glyphosate in the late 1980s as an alternative in the last weed-control phase prior to planting (Schramm 1992).

I was infected with the prairie restoration bug early on during this era. Subsequently, in early June 1973, I seeded eight acres on the University of Northern Iowa campus with five species of warm-season grasses. Forbs were added at various later dates. Little did I know what the future held as I drove across the planting site, pulling a Nesbitt drill and hoping I would get it right.

Bob Betz and his associates increased the scale of prairie reconstructions significantly in 1974 when they began a 700-acre prairie planting within the main ring of the Fermi Accelerator Laboratory (Betz 1986). By 1988, prairie plantings totaled 700 acres and the project was billed as the largest prairie reconstruction in the world. They also introduced a new approach to seeding prairie reconstructions. The first stage of their plantings consisted of a prairie matrix mixture of aggressive, tenacious species selected to compete with exotic weedy species. Later, as the matrix species became established, less-aggressive native species were added (Betz 1986). The advisability of their method is now being disputed (Weber 1999), but it worked for the Fermilab. Ultimately, the Fermilab Prairie would consist of 1200 acres of reconstructed prairie.

The Fermilab Prairie Project paved the way for larger prairie reconstruction projects near the turn of the twentieth century—projects such as the 8,600-acre prairie and savanna reconstruction/remnant restoration initiated in 1991 at the Neal Smith National Wildlife Refuge near Des Moines, Iowa, and multicommunity reconstructions/remnant restorations on 19,000 acres at the Midewin National Tallgrass Prairie established in 1996 near Joliet, Illinois.

From the prior historical summary, it is apparent that prairie reconstructionists contributed much to maintaining interest in prairie restoration over the decades. Interest became widespread within the general public, conservation groups, students, public agency personnel, and employees of private companies. The number of businesses marketing prairie-related products grew to include native-seed growers, landscaping companies, nurseries, equipment companies, and chemical companies. In Iowa, we progressed from no native seed growers in the 1960s to a dozen or more currently. Private prairie groups such as the Iowa Prairie Network, Prairie Enthusiasts, Grand Prairie Friends, Wild Ones, and Save the Prairie Society formed to promote prairies, help save prairie remnants, and contribute to prairie restoration. State and national private nonprofit organizations such as The Nature Conservancy, the Audubon Society, Ducks Unlimited, Pheasants Forever, the Iowa Natural Heritage Foundation, and the Missouri Prairie Foundation added prairie restoration and management to their preservation and protection activities. Several state departments of transportation and most Iowa counties began to use prairie plants in rights-of-way as a part of their roadside vegetation management programs. The Federal Highway Administration and the USDA Natural Resources Conservation Service funded programs to support prairie plantings and education. The Conservation Reserve Program alone has funded the planting of hundreds of thousands of acres of prairie species on highly erodible soils of marginal farmland. Homeowners and corporations often use native prairie plants to landscape their property.

Interest in prairie and prairie restoration has become widespread and continues to increase. When I first became involved in prairie restoration in the early 1970s, I was aware of most, if not all, prairie-related projects in Iowa. Today, I know of only a small portion of the prairie activities in the state. Attendance at the North American Prairie Conferences has paralleled the increased interest. One hundred twenty people attended the first conference at Knox College in 1968. There were more than 500 participants at the twelfth conference at UNI in 1990, and attendance at the seventeenth conference in Mason City in 2000 exceeded 650. In a difficult economic period, 560 participated in the 2010 conference.

In the past forty years, there have been some significant advances in procedures and techniques by practitioners of prairie reconstruction. Many of the advances were the result of trial and error or fortuitous experiences. In the past four decades, seeding rates of prairie grasses were reduced by 5 times or more, from 30-40 lbs/acre (33-45 kg/ha) to 6-8 lbs/acre (7-9 kg/ha); seed mixtures are now designed with specific numbers of grass, sedge, and forb seeds per unit area (e.g., 40-50 seeds/square foot; 430-540 seeds/square meter); and a new establishment technique of frequent mowing at 3-6 inches (7-15 cm) the first growing season replaced a one-time 12-14 inch (30-36 cm) mowing at the end of the summer. Extensive tilling for site preparation has been almost entirely replaced with glyphosate application, and there is much more attention to drilling depth. Although some reconstructions are fall seeded, most are still seeded in the spring. High-quality, certified, source-identified seed is more readily available at a reasonable price and is increasingly used in plantings. Unfortunately, prairie seed mixtures containing species not endemic to a particular area may still be planted by the uninformed. However, the use of seed of western cultivars and inappropriate sources has been greatly reduced.

A common restoration goal in the tallgrass prairie region is to recreate a prairie like that which existed prior to Euro-American settlement (Allison 2004). Such a goal faces some stiff challenges. Very little detailed information is available regarding plant and many animal species of the pre-settlement prairie. Iowa has no available plant-species lists of pre-settlement prairies to guide composition determinations for either prairie reconstruction or remnant restoration. In Illinois, Mead (1846), a physician and botanical enthusiast, compiled species lists for different locations with notations on which plants were common or rare. However, his work was limited to one county on the wesern edge of the state, and no information was provided regarding relative proportions of species or species associates (Allison 2004).

In the absence of species lists, existing prairie remnants are often suggested as references or resources for species presence and composition. Even with remnants available, replication of the original prairie is difficult. The remnants may not be representative of pre-settlement prairie. They have been isolated for more than a century and deprived of fire, resulting in changes in species composition and possible loss of original species. In addition, most remnants occur on sandy soils, rocky outcrops, steep hills or other nontillable land areas. These prairie types comprised a very small portion of the pre-settlement prairie and differed considerably from the more abundant blacksoil prairie. Even the blacksoil remnants that survived as hayfields have had their species composition modified if they were hayed annually at about the same date.

Ecological considerations may also limit prairie reconstruction or remnant restoration. As discussed earlier, isolated, unmanaged prairie remnants in a human-modified landscapes lose their historical character and native biodiversity. If these remnants can't recover and undergo succession toward their pre-disturbance form with human assistance, then restoration may be impossible or at least untenable (Simpson 2008). Critics of restoring prairie ecosystems to a historical reference condition often emphasize that succession is variable and unpredictable. Choi (2007) rejects the idea of restoring ecosystems to a historical condition as "past-oriented, static and idealistic" and recommends "multiple alternative goals and trajectories for unpredictable endpoints."

There may be philosophical objections to prairie reconstruction or remnant restoration. A goal of restoring tallgrass prairie to a pre-settlement state has been interpreted by some as an attempt to return to a more natural condition unaffected by humans. They feel that such a goal is inappropriate and misleading because it perpetuates the "wilderness myth" that the entire Western Hemisphere was in a natural condition, free from significant human influence, when encountered by Columbus (Callicott 2002). However, it is now generally accepted that American Indians did modify their environment and that much of the pre-settlement eastern tallgrass prairie persisted because of their burning practices. Assuming this condition in restoration planning reduces the necessity of the argument regarding the dichotomy of human influence and natural condition. The challenge that remains is to mimic landscape-scale processes of fire and herbivory.

Restoration of a degraded remnant involves a variety of considerations, including extent and type of disturbance or degradation, how to halt degradation, how to control invasive and aggressive woody species without harming natives, and whether to replace extirpated species. Similarly, each prairie reconstruction is the product of a unique set of circumstances of soil structure and temperature, existing vegetation, seeding mix, seeding rate and time, seed viability and germinability, as well as pre and postplanting weather conditions.

The range of variables for each type of restoration has contributed to the opinion that general methods cannot be formulated for widespread use. Consequently, much of the information regarding prairie restoration procedures and techniques has been anecdotal, derived from word-of-mouth experiences of practitioners. Although the anecdotal information is valuable, it is not always repeatable. Hobbs and Norton (1996) rather emphatically state, "What is clear is that restoration ecology has largely progressed on an ad-hoc, site- and situation-specific basis, with little development of general theory or principles that would allow the transfer of methodologies from one situation to another." I interpret their comment as a statement of fact rather than a criticism of past practice. However, it does clearly imply that information generated by practitioners should be tested in order to develop prairie restoration principles that can then be successfully applied by all practitioners. Development of restoration and reconstruction principles will avoid the perpetuation of mythical planting guidelines, untenable agency, seeding-time recommendations, and inappropriate seed mixes solely designed to produce flowering splashes.

Significant strides have been made in prairie reconstruction and restoration, but much remains to be done. Most of the early prairie reconstructions focused on prairie plants with little or no consideration of animals. Anderson (2009) suggests that a more comprehensive community view of prairie restoration is emerging and includes restoration and research efforts involving various organisms, including invertebrates, birds, small mammals, large herbivores, burrowing animals, fungi, bacteria, and mycorrhizal fungi. Prairie reconstruction and remnant restoration provide a means to test ecological theories. Prairie restoration practitioners and restoration ecologists should work together to develop and test principles and practices of prairie reconstruction and restoration. The knowledge gained will be invaluable to future generations as they try to meet the anticipated need for increased restoration of the tallgrass prairie resulting from ecosystem damage and alteration due to increases in human population and technological capability. We have a responsibility to both present society and future generations to restore and maintain functioning prairie ecosystems.

A valuable tallgrass prairie ecosystem is vanishing and will be gone unless our society assumes more responsibility for its retention. The best hope for retaining the historical tallgrass prairie ecosystem is to restore degraded remnants and reconstruct new patches of high-quality, species-appropriate prairie. This should include some landscape-scale reconstructions/restorations of thousands of acres in extent. Remnant prairie restoration and prairie reconstruction will continue to be done in a landscape that is being increasingly modified by humans. Thus, restorationists will need to provide the impetus, expertise, and materials to recreate and maintain this historical ecosystem in an alien landscape.

Though recreating the original prairie may not be possible, these restorations and reconstructions provide an opportunity to participate actively in the recovery of a degraded, damaged, or destroyed ecosystem. In the process, we can learn more about this recovering ecosystem. However, we must curb our arrogance and avoid creating the impression that reconstructing prairies can replace the need to preserve prairie remnants. Like Schulenburg, we need to be reminded that we can't provide all the elements. Certainly, we cannot recreate the diverse collection of plants, animals, and microbes that persist in prairie remnants. Schramm's (1992) goal of reconstructing a facsimile of original prairie is appropriate, as reconstructions enable ecological processes of soil building, increased water interception and infiltration, habitat improvement, and increased biodiversity.

We should be motivated to restore tallgrass prairie because we realize that something valuable is being lost. We restore prairie remnants and reconstruct prairies because we value the tallgrass prairie ecosystem and the organisms that inhabit it, knowing all the while that we can never achieve a complete replication of the historical landscape, and knowing that our responsibility is perpetual (Jordan 2003).

In spite of the many challenges associated with restoring the tallgrass prairie ecosystem, the ecological, economic, educational, and cultural rewards are well worth the effort. It is a part of our biological and cultural heritage, and it should be available for people to visit, experience, and understand.

Furthermore, there is a certain redemptive value in spending time on a prairie. My friend Arnold Webster commented as we walked off Cedar Hills Sand Prairie late one morning, "After spending an hour or so on the prairie, I can begin to feel the cussedness run out of me." Retaining and increasing this national treasure is a wise investment for the future.

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