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Assessing Wetland Functional Condition Change in Agricultural Landscapes

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Executive Summary

For more than a century, wetlands on private lands have been modified for growing crops, raising livestock, harvesting timber, building of infrastructure to support development expansion, and, in recent times, aquaculture. The individual and societal benefits provided by wetlands are well documented, and because of these benefits Federal, State and local government agencies and public and private organizations have worked together over the past several decades to reverse the decline of wetland acreage. More recently, the need to protect and restore wetland functions has been recognized. Studies investigating the relationship between historical and current activities in the surrounding landscape and wetland condition have concluded that the level of wetland functioning is determined in part by this relationship. If wetlands are to provide benefits at an optimum level, efforts to manage activities affecting wetlands must also include those beyond the wetland boundary.

The U.S. Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS) identified *healthy and productive wetlands sustaining watersheds and wildlife* as one of its national Strategic Plan objectives (USDA, NRCS 1997). To address this objective, NRCS initiated a National Wetlands Functional Assessment Pilot to determine whether NRCS was achieving a net increase in wetland function on agricultural lands. The pilot also provided the means to determine how the agency could achieve a more comprehensive approach to assess and track wetland functional condition over time. A workgroup was assembled from within NRCS to conduct a study in geographic locations historically and currently important to agricultural activities. Restoration of wetlands through USDA programs and NRCS technical assistance in these regions is intended to increase the wetland base that has experienced some of the greatest losses in the conterminous United States because of agricultural activities.

Three geographic regions were selected for sampling: the Northern Prairie Pothole Region (NPPR); the Central and Lower Mississippi Alluvial Valley (CMV and LMAV); and the High Plains (HP). Wetlands that are dominant in each of these regions and also make up the majority of wetlands restored were selected. Approximately 15 sites that have been targeted for restoration through USDA programs or have benefited from NRCS technical assistance were selected, as were approximately 15 *reference* wetlands that serve, together with the restoration sites, to provide a snapshot of wetland functional condition in each of the three regions.

To sample wetland functions directly and scientifically would be cost prohibitive and time consuming. Instead, the pilot workgroup elected to use a method of assessing wetland functions that requires wetlands first be classified based on water source, hydrodynamics, and geomorphic location, and then sampled using models developed from a reference wetland dataset specifically for that class. The method is commonly referred to as the hydrogeomorphic method or

HGM (Brinson 1993; Smith et al. 1995), and is used by NRCS to implement “Swampbuster” provisions of the 1985 Farm Bill, as amended. It is also identified in National Conservation Practice Standard Nos. 656, 657, and 658 as a method to conduct pre- and post-functional assessments on lands targeted for wetland restoration, creation, or enhancement by NRCS. Wetlands having the same hydrogeomorphic characteristics exhibit similar functions, and therefore can be placed in the same hydrogeomorphic class. Hydrogeomorphic classes can further be characterized by regional subclasses, which are described by large-scale factors such as homogenous climate and geology. The subclasses identified for this study are temporary and seasonal prairie pothole wetlands that are dominated by ground water recharge processes in the NPPR; forested, low-gradient riverine wetlands, often referred to as bottomland hardwood wetlands, of the CMV and LMAV; and playa wetlands, a prominent wetland type occupying depressions of varying sizes on the HP.

The HGM functional models are used to assess wetland *functional capacity* for an individual wetland at any point in time. Wetland functional capacity is used to assess the relative condition of a wetland to perform a suite of functions characteristic of an HGM subclass. Functions typically fall into four general categories: Hydrology, Biogeochemistry, Native Plant Community, and Fish/Wildlife Habitat. Wetlands with a high functional capacity for all functions exhibit *sustainable functional capacity*. A *functional capacity index* (FCI) is developed for each function, and ranges from zero (unrestorable) to one (highest sustainable functional capacity).

Restoration and reference sites were sampled once during 1998. FCI values calculated during 1998 for the restoration and reference sites represent current conditions (T1). Functional capacities for conditions before restoration (T0) at the restoration sites were estimated based on the FCI values calculated for agriculturally altered sites sampled from the reference wetland population, except for the LMAV where they were independently estimated. A net change in functional capacity was determined for each site (T1 – T0). Mean FCI values were calculated for T0, T1, and the net change for each function. Median FCI values were calculated for current conditions for each function to compare restoration and reference sites. Summing of FCI values across functions is not appropriate, nor is summing of functions across subclasses. HGM models are designed specifically for a particular wetland subclass. Because of the small sample size, the results presented are not statistically significant.

The results of this pilot indicate that there is a modest relative increase in mean functional capacity for wetland restoration sites on agricultural lands within the three geographic regions sampled. More importantly, the results indicate that the relative condition of many of the restoration sites is not at the highest sustainable functional capacity. Several possible reasons for this include vegetation structure, historic, and current land use activities in the surrounding landscape, lack of appropriate restoration techniques, landowner preferences for establishing a wetland subclass other than the one fitting the landscape, and modification of the restoration site to address adjacent landowner concerns with hydrologic restoration. However, the results do provide insight into the challenges faced by NRCS to restore wetlands at sustainable conditions. The following recommendations are discussed as a multifaceted approach to monitor wetland functional conditions:

1) Implement broad-scale wetland assessments
2) Implement site-specific wetland assessments
3) Add additional wetland elements to the National Resources Inventory
4) Identify wetland functional subclasses before restoration
5) Implement a geospatial wetland restoration strategy
6) Develop *Ecological Site Descriptions* for wetlands

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Acronyms and Abbreviations:

AR	Arkansas
BLH	Bottomland Hardwood Wetlands
CFR	Code of Federal Regulations
ChE	Cholinesterase
CMV	Central Mississippi Valley
CRP	Conservation Reserve Program
EMAP	Environmental Monitoring and Assessment Program
FCI	Functional Capacity Index
FSA	Farm Service Agency
GIS	Geographic Information System
HGM	Hydrogeomorphic method
HP	High Plains
HUC	Hydrologic Unit Code
KS	Kansas
LA	Louisiana
LMAV	Lower Mississippi Alluvial Valley
MAV	Mississippi Alluvial Valley
MLRA	Major Land Resource Area
MO	Missouri
MS	Mississippi
ND	North Dakota
NPPR	Northern Prairie Pothole Region
NPR	Northern Prairie Region
NRCS	Natural Resources Conservation Service
NRI	National Resources Inventory
PPR	Prairie Pothole Region
RAD	Resource Assessment Division
SCS	Soil Conservation Service
SHP	Southern High Plains
T0	Time Zero, Prior to restoration
T1	Time One, After restoration initiated
T2	Time Two, Year 2000
T5	Time Five, Year 2003
USDA	United States Department of Agriculture
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
WPA	Waterfowl Production Areas
WRP	Wetland Reserve Program

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Introduction

In 1893, a 47-mile-long levee was constructed along the Mississippi River from the Arkansas state line to New Madrid, Missouri. By 1910 the Federal Government had assumed maintenance of the levee to ensure that it would prevent flooding of the Mississippi River bottomland hardwood wetlands that had been converted to productive agricultural land. Federal funds were now used to support river commissions to build extensive levees systems up and down the Mississippi River. The reclamation of the fertile bottomlands of the river had begun under the Federal mandate to expand agriculture and commerce.

Between 1899 and 1919, more than 250 drainage districts were formed to undertake the draining of the wetlands in the lower Mississippi River basin. More than 2,900 miles of drainage ditches were constructed to affect more than 1.7 million acres of *worthless swamp land* along the Mississippi River in the southeastern counties of Missouri (Hidinger 1919; Missouri Bureau of Labor Statistics 1910). Only one-tenth of the *virgin* bottomland hardwood forest remained by 1919 in the lower Mississippi River drainageway. The major part of it was in cultivation and the remaining cutover bottoms actively going into cultivation at the time (Hidinger 1919).

The 1893 Missouri law opened the door to reclamation of *idle* bottomland. This was one of many state and federal initiatives within the past 200 years specifically developed to target the conversion of U.S. wetlands. Between 1937 and 1977, an estimated 2.7 million acres of bottomland hardwood wetland was lost through construction of drainage ditches and levees in the Mississippi Alluvial Valley (Taylor et al. 1990).

In the Prairie Pothole Region, drainage of the most productive waterfowl breeding habitat in this country had caused conservationists in the 1930's to demand that the drained potholes and other wetlands throughout the United States be restored (Page et al. 1938). Losses of prairie pothole wetlands were estimated in the 1970's at more than 4 million acres. Most of these losses are attributed to agricultural drainage and conversion to cropland (Tiner 1984). The historical extent of playa wetlands throughout the High Plains is unknown. However, current estimates indicate there are between 25,000 – 30,000 playa wetlands on the Southern High Plains, an area of intensive agricultural use since the 1920's (Luo et al. 1997; Bolen et al. 1989; Osterkamp and Wood 1987; Guthery and Bryant 1982).

In 1997, the U.S. Department of Agriculture (USDA), Natural Resources Conservation Service (NRCS) identified *healthy and productive wetlands sustaining watersheds and wildlife* as one of its National Strategic Plan objectives (USDA, NRCS 1997). To achieve that objective, NRCS identified *a net increase in wetland functions on agricultural land* by the year 2000 as a Strategic Plan Performance Measure. This effort was initiated in March 1998 with establishment of an interim National Wetlands Functional Assessment Pilot Workgroup.

The Workgroup identified several key geographic areas where wetland restoration on agricultural lands has significantly involved NRCS funds and technical assistance. Working from a draft strategy prepared by the NRCS Resource Assessment Division, the Workgroup decided to use available NRCS *interim* or *draft* functional assessment models. The models had been developed as part of the application of the wetland functional assessment tool, *An Approach for Assessing Wetland Functions Using Hydrogeomorphic Classes* (HGM), *Reference Wetlands and Functional Indices* (Smith et al. 1995) to implement wetland conservation provisions of the 1985 Farm Bill and subsequent amendments. The HGM wetland functional assessment approach was developed by representatives from several Federal agencies, including NRCS.

The HGM approach to wetland functional assessment is based on the premise that wetlands can be categorized into one of seven classes based on hydrogeomorphic characteristics (landscape position, hydrologic sources, and hydrodynamics). The classes and examples of each are shown in table 1. Wetlands that have similar hydrogeomorphic characteristics exhibit similar functions, and therefore can be grouped into the same hydrogeomorphic class (Brinson 1993).

Hydrogeomorphic classes can be further classified into HGM regional subclasses. Regional subclasses are described by homogenous climate, geology, and other large-scale factors involved in developing wetland functions at some defined geographic scale (Smith et al. 1995). A descriptions of HGM subclasses are provided in Appendix A: Description of Study Area.

The HGM approach involves the development of functional assessment models, which are used to assess wetland *functional capacity* for an individual wetland at any point in time. Because of the difficulty and cost of measuring wetland functions in absolute terms, these models provide a more efficient means of assessing the relative capacity of a wetland to perform a suite of functions characteristic of an HGM subclass. The functional capacity of a wetland is one way of quantifying wetland health or condition. The condition of a wetland is determined by the degree to which the wetland's hydrological, biogeochemical, and biotic characteristics have been altered by anthropogenic activities within the wetland as well as within the landscape. Wetlands with a high functional capacity for all functions exhibit sustainable functional capacity. As defined by Smith et al. (1995), wetlands and landscapes that have not been impacted by long-term anthropogenic alterations are considered sustainable if *structural components and physical, chemical, and biological processes in the wetland and surrounding landscape reach the dynamic equilibrium necessary to achieve the highest sustainable functional capacity*.

Therefore, those wetlands and landscapes that have undergone the least amount of anthropogenic alteration are those that exhibit sustainable functional capacity. For purposes of this document, the terms *sustainable, sustainable functional capacity, sustainable functional condition, and highest sustainable functional capacity* refer to the definition proposed by Smith et al. (1995) and are interchangeable in meaning.

In many cases, the landscapes in which restorations are occurring have changed considerably during historic times. In the Prairie Pothole Region, regional ground water levels are lower than historic levels because of agricultural drainage (Euliss and Mushet 1999; Galatowitsch and van der Valk 1994). Agricultural activities within temporary wetlands have resulted in lower diversity of plant species and greater percentages of unvegetated bottom in the wet meadow communities (Kantrud and Newton 1996). Fragmentation and drainage of prairie potholes have resulted in replacement of wetland complexes with isolated wetlands. Often the isolated wetlands remaining in the landscape are the larger semipermanent wetlands that were historically too costly to drain (Galatowitsch and van der Valk 1994). Restoration of prairie potholes that relies on seed banks or propagule dispersal from nearby wetlands to revegetate sites will likely result in compositional differences and fewer species in restored wetlands because of impacts from intensive, long-term cropping (Galatowitsch and van der Valk 1996; Weinhold and van der Valk 1989).

Table 1. Wetland classes categorized by the hydrogeomorphic classification (Brinson 1993)

Class	Example
Depressional	Wetlands that occupy a concave feature in the landscape such as prairie potholes
Lacustrine Fringe	Freshwater marshes along a lake or pond shoreline
Tidal Fringe	Regularly flooded salt marshes, mangrove swamps
Slope	Wetlands along floodplain side slopes where ground water flow is discharged along the slope surface because of bedrock, a confining layer or other surficial feature
Riverine	Wetlands along a surface water system, such as a river or creek, where the hydrologic flow is unidirectional, such as bottomland hardwood wetlands
Mineral Flats	Wetlands on a stream interfluve with a predominantly mineral soil
Organic Flats	Wetlands dominated by the vertical accretion of organic matter on interfluves

Extensive levee construction along the Mississippi River for 100 years has resulted in:

- altering hydroperiods,
- clearing of bottomland hardwood wetlands for agriculture and timber production,
- contaminating wetland water columns and sediments with agricultural runoff, and
- extensive fragmenting of the forested matrix historically characterizing such systems (Harris and Gosselink 1990; Taylor et al. 1990; Odum and Larson 1980; Cairns et al. 1980).

The landscape of the Southern High Plains in which the majority of playa wetlands are embedded (Bolen et al. 1989) is characterized by intensive irrigated agriculture and grazing (Haukos and Smith 1993; Nelson et al. 1983). Such land uses have resulted in modifying playa basins by:

- lowering of surface water caused by digging of pits to hold irrigation water;
- increasing the length and depth, as well as the timing, of water on the playas from the addition of irrigation tailwater;
- contaminating playa soils and surface water from feedlot runoff; and
- removing native vegetation through direct cropping and grazing of the playa (Bolen et al. 1989).

Such changes in the landscape are a challenge to restoration, as it involves attempting to return a sustainable landscape feature within a matrix from which it did not develop. Whether sustainable wetland ecosystems have been restored is the subject of this document.

The results presented here address the outcome of applying the available interim and draft HGM models for three hydrogeomorphic regional subclasses. This assists NRCS in measuring their contribution toward restoring wetland functions on agricultural lands. The numeric values calculated for the functions of the wetlands assessed represent as much of the range of variability in restoring wetland functions as could be supported with the funds provided. However, it is imperative for readers and NRCS decisionmakers to understand that the numeric values represent a small sample size of the wetland types assessed. Therefore values should not be used to establish a functional capacity numerical baseline. Rather, the results of the pilot should be used to help formulate a cost-effective and ecologically meaningful mechanism to assess and track wetland functional capacity in different landscapes over time.

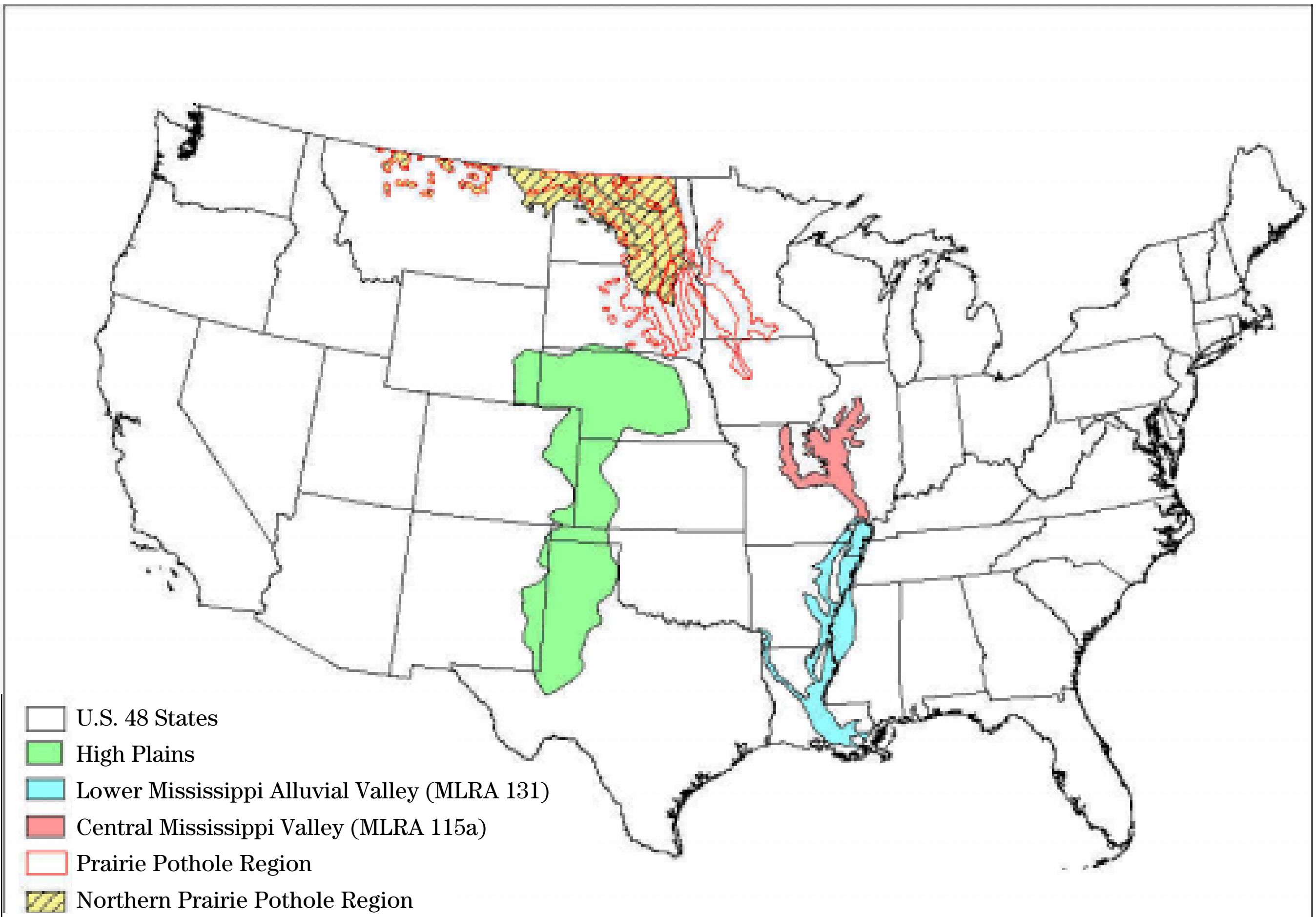
Study Area

Hydrogeomorphic regional subclasses were initially identified within five geographic areas by the Workgroup: the Central and Lower Mississippi Valley, the High Plains, Northern Prairie Pothole Region, New England Coastal Plain, and the Delmarva Peninsula.

Because of funding limitations, only the first three geographic areas were sampled during Fiscal Year 1998. Sampling of the Delmarva Peninsula was funded during Fiscal Year 1999. The three geographic regions selected for assessing wetland functions were the Northern Prairie Pothole Region (NPPR); the Central and Lower Mississippi Alluvial Valley (CLMV) composed of the Dissected Till Plains of the Central Lowland Province and the Mississippi Alluvial Valley within the Coastal Plain Province (Fenneman 1938); and the High Plains (HP) (fig. 1). For each geographic area, a specific wetland HGM subclass was selected.

Selection was a function of predominance of wetland subclass impacted historically and currently by agricultural activities, as well as the subclass targeted most frequently for restoration by NRCS via technical assistance or USDA program dollars. Appendix A provides a description of the three geographic areas sampled.

Figure 1 Geographic areas selected for assessing wetland functional condition*



* Boundary of NPPR is approximate. Information used to depict geographic extent includes Major Land Resource Areas, Land Resource Regions, and physiographic regions. USGS, Northern Prairie Wildlife Research Center provided the Prairie Pothole Region coverage.

Methods

Site selection

Approximately 15 (range: 5) sites targeted for restoration (referred to hereafter as *restoration sites*) as a result of NRCS technical assistance or program funds were selected from each HGM subclass. However, because of the extent of the Central and Lower Mississippi Alluvial Valley (CLMV), regional physiographic differences and differences in local edaphic, vegetation, and restoration practices, the region was divided into two sampling areas. The *upper* portion, hereafter referred to as the Central Mississippi Valley (CMV), included Missouri (MO) and Illinois (IL; one site), and the *lower* portion, known as the Lower Mississippi Alluvial Valley (LMAV), included Arkansas (AR), Louisiana (LA), and Mississippi (MS).

Restoration sites were selected within the geographic range of the subclass based on several factors. They were chosen to represent a range of ages (time since restoration initiated), geographic extent of the subclass (where travel funds permitted), and a qualitative range of restored conditions (those that *appeared* to be functioning as well as those where physical and functional conditions were noticeably less than expected or planned). Restoration sites ranged in mean age from 2 years for the LMAV, 4.5 years in the CMV, 6.8 for the NPPR, and 10 years for the HP sites. Restoration sites were defined as areas that included former wetlands and where NRCS provided technical assistance in accordance with National Conservation Practice Standard No. 657. Also included were sites where programmatic funds were used to enroll a landowner in a USDA conservation program that directly or indirectly targeted wetland restoration (i.e., Wetland Reserve Program and Conservation Reserve Program). No attempt was made to qualify a site as *restored,* as the NRCS Workgroup agreed that restoration is a continuing process that requires long-term monitoring until the site becomes self-sustaining. In addition, restoration sites funded under the Wetland Reserve Program may also include some degree of enhancement (USDA, NRCS 1996) as well as inclusion of an upland buffer that also often requires restorative treatment.

In addition to the wetlands undergoing restoration, approximately 15 (maximum 20, minimum 15) *culturally altered* and relatively unaltered wetlands (i.e., reference sites) were also assessed for each of the three HGM subclasses. Culturally altered wetlands include naturally occurring wetlands that have been modified anthropogenically (hydrologically, chemically, and biologically) to some extent but not so degraded that they no longer exhibit wetland ecosystem functions. Those relatively unaltered wetlands are characterized by having intact hydrologic, chemical, and biological processes resulting in sustainable functional capacity. Many unaltered wetlands are relicts of wetland systems that existed before European colonization. A total of 118 restoration and reference sites were assessed for the three HGM subclasses.

Reference sites were selected in a stratified random fashion from the same HGM subclass as the restoration sites. Site selection factors included the variability of wetland condition of reference sites currently on the landscape, the range of anthropogenically altered conditions present in each subclass, and the variability associated with geographic locations within the sampled regions. Because of the small sample size, the reference sites were not selected as paired sites with the sample restoration sites, but were proximate to the sample restoration sites. The purpose for sampling reference sites was to provide a snapshot of wetland condition on the landscape as a whole, but comprised of two subpopulations—restoration and reference sites. The method also provides a template for monitoring changes in wetland functional capacity at a landscape level.

Wetland functional assessment procedure

Wetland functions were assessed using HGM models (Smith et al. 1995) developed for each of the following subclasses: temporary and seasonal prairie pothole wetlands of the NPPR; forested, low-gradient riverine wetlands in the CMV and LMAV; and playa wetlands of the HP. Two of the three subclass models applied were developed as interim models by NRCS State Offices for application of the Swampbuster provisions of the 1985 Farm Bill, as amended (i.e., playa wetlands and the forested, low-gradient riverine wetlands). The interim models used to assess sites in the

CMV and LMAV were adapted by NRCS from those developed for a Regional Guidebook addressing mature forested, low-gradient riverine wetlands in western Kentucky (Ainslie et al. 1999). The HGM models used to assess the temporary and seasonal prairie pothole wetlands of the NPPR are contained in a draft regional guidebook (Lee et al. 1997), also used to apply “Swampbuster.” A list of the functions assessed for each subclass is presented in table 2.

Sites were assessed once during the 1998 growing season by NRCS biologists and other professionals having expertise in wetland ecosystems. The HGM approach was chosen as the functional assessment method, as it provides a quantitative index (functional capacity index; FCI) that can be used to compare change in wetland functional condition among wetlands of the same subclass. Current site conditions were used to assess the FCI for all 53 restoration sites for the three subclasses and represent conditions after restoration was initiated (T1). Functional capacity index values for conditions before restoration, T0, were estimated based on the FCI values calculated for agriculturally altered sites included in the reference sites data set. The exception to this protocol was for the LMAV, where T0 FCI values were independently estimated as the reference sites data set included only mature forested, low-gradient riverine wetlands. Projected FCI values were calculated for an additional 5 years beyond T1 (T2 through T5). Time periods T1 through T5 roughly correspond to calendar years 1999 through 2003. The future estimated FCI values were determined as part of the National Wetland Functional Assessment Pilot to determine precision in estimating FCI for future years as part of a long-term monitoring effort. Only T1 and T0 FCI values are reported in this document.

The HGM models used are comprised of wetland functions that generally equate to vegetation, hydrol-

Table 2. Functions assessed for three wetland hydrogeomorphic subclasses

Subclass	Function
Seasonal and temporary prairie potholes (NPPR)	Static surface water storage
	Dynamic surface water storage
	Nutrient cycling
	Removal of imported elements and compounds
	Retention of particulates
	Provide environment for characteristic plant community
	Habitat structure within wetland
	Habitat interspersion and connectivity among wetlands
Forested, low-gradient riverine wetlands (CMV and LMAV)	Temporary storage of surface water
	Retention and retarding the movement of ground water
	Cycling of nutrients
	Removal and sequestration of elements and compounds
	Retention of particulates
	Organic carbon export
	Provide environment for native plant community
	Promote wildlife habitat
Playa wetlands (HP)	Maintain characteristic static or dynamic storage, soil moisture, and ground water interactions
	Elemental cycling and retention of particulates
	Plant community
	Faunal habitat, food webs, and habitat interspersion

ogy, wildlife, and biogeochemical functions. Each function is constructed of two or more variables that are either directly measured or indirectly measured using a surrogate for the variable. A subindex score is determined for each variable. An FCI for each function is derived from the mathematical relationship among the variables and the subindex scores calculated for each variable. The FCI ranges from 0.00 to 1.00. An FCI of 1.00 represents the highest sustainable functioning capacity for any given function in a wetland (sustainable functional condition). FCI values are determined independently for each function, and functions are not interchangeable among wetland subclasses. For example, while a hydrologic function may be measured for temporary and seasonal prairie pothole wetlands and forested, low-gradient riverine wetlands, the specific variables measured will likely differ because the differences in geomorphic setting and the hydrodynamics of each subclass. Variables for the hydrologic function of each subclass are therefore also different, as is the scoring of the variables to determine the subindex score (Hauer and Smith 1998). A table listing each function by subclass, the equation used to calculate the functional capacity index for each function, and a description of the variables used in each functional equation appear in appendix B.

Statistical analysis

Because of the small sample size, statistical tests for differences among sites and between time periods (before and following restoration, T0 and T1 respectively) within a subclass were not conducted. Descriptive statistics (mean, maximum, minimum, and variance of FCI values) were calculated for each restored data set for each subclass. Because of the variability encountered in the reference sites, median values were calculated for both the reference sites and restored wetlands to compare condition of reference and restoration sites at a broad scale for each subclass. The wetland median FCI values of the reference sites were calculated for current conditions only, and were compared to the T1 FCI median values calculated for the restoration sites. Where the data allowed, analysis of age or vegetation stage was also conducted for the wetland subclasses. In addition, because of differences in restoration techniques in the Mississippi Valley, an analysis was conducted comparing the results in the CMV to those in the LMAV.

Results

Temporary and seasonal prairie pothole wetlands, Northern Prairie Pothole Region

Restoration sites

Eight wetland ecosystem functions were measured for 13 restoration sites (table 2). The change in mean FCI between T0 and T1 for each function is shown in table 3. Restoration site FCI values for each function at T0 and T1 are in appendix C, figures 1 – 8. The change in mean FCI differed among functions, with the *Retention of particulates* function exhibiting the greatest mean FCI change. The *Provide environment for characteristic plant community* function had the smallest mean FCI change between T0 and T1. In general, the functional capacity of restored temporary and seasonal prairie pothole wetlands sampled in the NPPR increased overall.

Functional capacity of the restoration sites was markedly different within and between sites and among functions. Before restoration, three restoration sites had calculated site FCI's of 0.00 for the *Static water storage* function compared to all 13 restoration sites having FCI's of 0.00 at T0 for the *Dynamic water storage* function (app. C, figs. 1 and 2). At T1, the 13 restoration sites exhibited some functional capacity increase in the *Static water storage* function, but an FCI of 0.00 was calculated for the *Dynamic surface water storage* function for four of the 13 restoration sites after restoration was initiated.

Changes in mean FCI values varied for the three biogeochemical functions measured (table 3). There was a comparable increase of the FCI values for the *Nutrient cycling and removal of imported elements and compounds* functions (0.41 and 0.44 mean FCI increase, respectively). However, the change in FCI was greatest for the *Retention of particulates* function (0.63 mean FCI increase) (table 3).

The mean FCI value at T0 for the *Provide environment for characteristic plant community* function was 0.28, and at T1 it was 0.68, an increase of 0.40 (table 3). The mean FCI values for the two wildlife habitat functions measured also increased. The mean FCI value for the *Habitat structure within wetland*

Table 3. Change in mean FCI (T1 – T0) for each function within a subclass

Subclass	Function	T0	T1	Mean change
Temporary and seasonal prairie pothole wetlands, NPPR	Static surface water storage	0.20	0.74	0.54
	Dynamic surface water storage	0.00	0.57	0.57
	Nutrient cycling	0.23	0.64	0.41
	Removal of imported elements and compounds	0.25	0.70	0.45
	Retention of particulates	0.10	0.73	0.63
	Provide environment for characteristic plant community	0.28	0.68	0.40
	Habitat structure within wetland	0.15	0.71	0.56
	Habitat interspersion and connectivity among wetlands	0.31	0.78	0.47
Forested, low-gradient riverine wetlands, LMAV (AR, LA, MS)	Temporary storage of surface water	0.59	0.59	0.00
	Retention and retarding the movement of ground water	0.41	0.41	0.00
	Cycling of nutrients	0.33	0.46	0.13
	Removal and sequestration of elements and compounds	0.73	0.73	0.00
	Retention of particulates	0.59	0.59	0.00
	Organic carbon export	0.31	0.40	0.09
	Provide environment for native plant	0.25	0.37	0.12
	Promote wildlife habitat community	0.53	0.56	0.03
Forested, low-gradient riverine wetlands, CMV (MO)	Temporary storage of surface water	0.37	0.38	0.01
	Retention and retarding the movement of ground water	0.65	0.74	0.09
	Cycling of nutrients	0.26	0.41	0.15
	Removal and sequestration of elements and compounds	0.55	0.61	0.06
	Retention of particulates	0.37	0.38	0.01
	Organic carbon export	0.21	0.35	0.14
	Provide environment for native plant community	0.29	0.43	0.14
	Promote wildlife habitat	0.39	0.45	0.06
Playa wetlands, High Plains (KS)	Maintain characteristic static or dynamic storage, soil moisture, and ground water interactions	0.60	0.81	0.21
	Elemental cycling and retention of particulates	0.43	0.73	0.30
	Plant community	0.37	0.69	0.32
	Faunal habitat, food webs, and habitat interspersion	0.23	0.68	0.45

function increased from 0.15 at T0 to 0.71 at T1. Site FCI values, however, ranged from 0.06 to 0.25 at T0 and 0.31 to 0.84 at T1 (app. C, fig. 7). The habitat function addressing the spatial relationship among the wetland under assessment and adjacent wetlands within the complex, *Habitat interspersion and connectivity among wetlands*, increased in mean FCI from 0.31 to 0.78, an increase in mean FCI of 0.47 (table 3). Site FCI values ranged from 0.18 to 0.53 at T0. At T1, site FCI values ranged from a low of 0.42 to a high of 0.96 (app. C, fig. 8).

Reference sites vs. restoration sites

Fifteen reference sites were assessed using the same functions assessed for the restoration sites (table 2). The median FCI for temporary and seasonal prairie pothole wetland reference sites sampled in the NPPR was compared to that for the restored wetlands for each function. Results of this comparison indicate that the reference sites wetlands exhibited a higher median FCI value than the restoration sites for five of the eight functions assessed (table 4). The differences do not appear significant, except for the *Removal of imported elements and compounds* function. Only one reference site sampled exhibited a sustainable FCI for seven of the eight functions assessed (FCI = 1.00).

Forested, low-gradient riverine wetlands, LMAV (AR, LA, MS)

Restoration sites

Mean FCI values were calculated for eight ecosystem functions (table 2) for 15 restoration sites in Arkansas, Louisiana, and Mississippi that fall within the LMAV. The change in mean FCI between T0 and T1 for each function is shown in table 3. Site FCI values at T0 and T1 are in appendix C, figures 9 to 16 for each function. Neither of the functions involving hydrologic dynamics and modification of offsite and onsite hydrology changed before or after restoration was initiated. FCI site values ranged from a minimum of 0.27 to a maximum of 0.74 for the *Temporary storage of surface water* function (app. C, fig. 9). The mean FCI at T0 and T1 was 0.59, with this function remaining relatively unchanged before and after restoration was initiated (table 3). A site FCI of 1.00, the highest sustainable functional index value attainable in the model, was assessed for the *Subsurface water storage* function at 2 of the 15 sites at T0 and T1 (app. C, fig. 10). The remaining 13 sites have T0 and T1 FCI values of 0.32.

Four functional models intended to measure the capacity of a wetland to perform various biogeochemical processes were used to derive FCI values. A slight increase in mean FCI was calculated between T0 and T1 for the *Nutrient cycling* function, with a change in mean FCI from 0.33 to 0.46. There was no change in mean FCI (0.73) calculated for the *Removal and sequestration of elements* function before and after restoration. Similarly, there was no change between T0 and T1 for the *Retention of particulates* function with mean FCI values of 0.59 at T0 and T1 (table 3). There were differences, however, among site FCI values (max 0.74; min 0.27; app. C, fig. 13). Mean FCI at T0 was 0.31 and 0.40 at T1 for the *Organic carbon export* function, a net change of 0.09 (table 3). There was also variation among site FCI values for this function, with minimum values at T0 and T1 of 0.26 and 0.29, respectively, and maximum values ranging from 0.32 to 0.73, respectively (app. C, fig. 14).

The mean FCI calculated for the *Maintenance of native plant communities* function at T0 was 0.25 and at T1 it was 0.37, a net change of 0.12 (table 3). The change in FCI varied among sites, with some sites showing no change in the calculated FCI between T0 and T1 (app. C, fig. 15). The function, *Maintenance of habitat support*, showed little change in mean FCI between T0 and T1, 0.53 and 0.56, respectively (table 3). Site FCI values for this function ranged from minimum values of 0.51 and 0.52 at T0 and T1 to maximum values of 0.56 and 0.60 at T0 and T1, respectively (app. C, fig. 16).

Reference sites vs. restoration sites

All eight ecosystem functions were assessed and FCI values calculated for 15 reference sites. The reference sites data set in this portion of the LMAV consisted of mature bottomland hardwood (BLH) wetlands, with the average age approximated at 60 to 65 years. Generally, the median FCI values for each function assessed were greater for the reference sites than those calculated for the restoration sites (table 4). The exception was for the *Subsurface water storage* function, where there was no difference between the reference and restoration site median FCI values.

Table 4. Comparison between reference and restoration site median FCI values for functions sampled in three wetland hydrogeomorphic subclasses

Subclass	Function	Reference sites	Restored sites
Temporary and seasonal prairie pothole wetlands, NPPR	Static surface water storage	0.82	0.79
	Dynamic surface water storage	0.75	0.79
	Nutrient cycling	0.58	0.68
	Removal of imported elements and compounds	0.80	0.69
	Retention of particulates	0.81	0.75
	Provide for characteristic plant community	0.78	0.71
	Habitat structure within wetland	0.76	0.75
	Habitat interspersion and connectivity among wetlands	0.58	0.79
Forested, low-gradient riverine wetlands, LMAV (AR, LA, MS)	Temporary storage of surface water	0.94	0.69
	Retention and retarding the movement of ground water	0.32	0.32
	Cycling of nutrients	0.78	0.48
	Removal and sequestration of elements and compounds	0.86	0.74
	Retention of particulates	0.94	0.69
	Organic carbon export	0.98	0.38
	Provide environment for native plant community	0.71	0.35
	Promote wildlife habitat	0.62	0.56
Forested, low-gradient riverine wetlands, CMV (MO, IL)	Temporary storage of surface water	0.42	0.42
	Retention and retarding the movement of ground water	1.00	1.00
	Cycling of nutrients	0.50	0.44
	Removal and sequestration of elements and compounds	0.80	0.74
	Retention of particulates	0.42	0.42
	Organic carbon export	0.27	0.32
	Provide environment for native plant community	0.50	0.43
	Promote wildlife habitat	0.56	0.40
Playa wetlands, the High Plains (KS)	Maintain characteristics static or dynamic storage, soil moisture, and ground water interactions	0.75	0.80
	Elemental cycling and retention of particulates	0.73	0.73
	Plant community	0.86	0.68
	Faunal habitat, food webs, and habitat interspersion	0.58	0.70

Forested, low-gradient riverine wetlands, CMV (MO, IL)

Restoration sites

The same HGM models applied in the LMAV were applied in the CMV (table 2). Site FCI values at T0 and T1 for each function are in appendix C, figures 17 to 24. A slight increase in mean FCI was calculated for the *Temporary storage of surface water* function (table 3). Site FCI values did not change between T0 and T1 for 13 of the 15 restoration sites (app. C, fig. 17). There was an increase in the mean FCI for the *Subsurface water retention* function (table 3), with the increase derived from 2 of the 15 restoration sites (app. C, fig. 18). Site FCI values for this function remained the same at T0 and T1 for the 13 remaining restoration sites.

Change in functional capacity for the four biogeochemical functions assessed varied by function and among sites. The mean FCI increased for the *Nutrient cycling* function from 0.26 to 0.41 (table 3), with all sites except one showing some increase in FCI from T0 (app. C, fig. 19). An increase in mean FCI was calculated for the *Removal and sequestration of elements* function (table 3). Four of the 15 restoration sites have an increase in site FCI and the FCI values for the remaining sites showed no change in site FCI for this function between T0 and T1 (app. C, fig. 20). A slight increase in mean FCI was calculated for the *Retention of particulates* function, from 0.37 to 0.38 (table 3). Two of the 15 restoration sites showed an increase in site FCI, and the remaining 13 sites showed no change in site FCI (app. C, fig. 21). Site FCI values for the *Organic carbon export* function also varied, ranging from 0.06 to 0.32 at T0 and 0.06 to 0.74 at T1. Ten of the 13 sites showed an increase in site FCI (app. C, fig. 22). The mean FCI for this function was 0.21 at T0 and 0.35 at T1, a net increase of 0.14 (table 3). The *Organic carbon export* and *Nutrient cycling* functions showed the greatest change in FCI of the four biogeochemical functions as well as the remaining functions for restoration sites in this wetland subclass.

An increase in mean FCI was calculated for the *Native plant community support* function (table 3). Thirteen of the 15 restoration sites had a site FCI increase, with FCI values ranging from 0.21 to 0.46 at T0 and from 0.27 to 0.62 at T1 (app. C, fig. 23). Mean FCI increased from 0.39 to 0.45 for the *Wildlife habitat* function (table 3), with all but one site showing a slight increase in site FCI values (app. C, fig. 24).

Reference sites vs. restoration sites

Similar to the sampling of the reference sites in the LMAV, 14 sites were selected in Missouri and one in Illinois within the CMV in close proximity to the restoration sites. The sites used to construct the reference site data set for the CMV consisted of mature BLH wetlands, early succession BLH wetlands dominated by shrub or herbaceous vegetation, and farmland and prior-converted wetlands under cultivation. The results show that the median FCI for the reference sites was higher than that for the restoration sites for four of the eight functions in the CMV (table 4). Three of the functions exhibited the same median FCI for restoration and reference sites. In addition, the median FCI value was lower for the *Export of organic carbon* function for reference sites compared to the restoration sites. However, this value changes when the reference sites are classified by vegetation structure. The mature reference sites exhibit the highest median FCI value (see discussions below and tables 4 and 6.

These results are different than those in the LMAV (table 4). Additional analysis shows that generally there is a greater difference between the reference site and restoration site median FCI values in the LMAV than between those in the CMV, with the notable exception of the *Maintenance of wildlife habitat* function (table 5). The difference may reflect site age, restoration techniques, selection of reference sites and vegetation structure, connectivity to other sites, lack of model sensitivity, or different interpretation of the same models by the two sampling teams.

To determine whether the disparity in median functional capacity index values between the CMV and LMAV could be further explained by differences in vegetation successional stage, the CMV reference sites data set was divided into three groups: Agriculturally altered wetlands, Early successional wetlands, and Mature bottomland hardwood wetlands. The median FCI values for each function were calculated for each group and qualitatively compared to the median FCI value for the restoration sites for the eight functions. The comparison indicates that the median FCI values for the restoration sites were more similar to those calculated for the early successional sites for three of the eight functions (*Nutrient cy-*

cling, Native plant community support, and *Wildlife habitat support*) (table 6). Median FCI values for restoration sites and agriculturally altered sites were the same for the *Temporary storage of surface water*, *Subsurface water retention and retardation*, *Removal and sequestration of elements,* and *Retention of particulate* functions. The median FCI values for the *Export of organic carbon* function was similar but not the same between the restoration and agriculturally altered sites.

A comparison was also made between the median FCI values for the mature bottomland hardwood reference sites wetlands in the CMV and LMAV (table 7). The median FCI values were lower for three of the four biogeochemical functions assessed for the mature BLH reference sites in the CMV than in the LMAV. The lower median FCI for the *Cycling of nutrients* function may be, in part, the result of younger stand age in the Missouri and Illinois sites. Average basal area (i.e., cross-sectional area of trunks measured at a standard height), an often-used surrogate of forest maturity, for the sites in Missouri and Illinois was 8.24 square meters. The average basal area for AR, LA, and MS was 13.39 square meters. Median FCI values for the *Native plant community support* and *Wildlife habitat support* functions were somewhat higher, however, for the Missouri/Illinois mature BLH reference sites, indicating that nonbiotic or landscape spatial variables scored higher for these functions in the CMV than in the LMAV. The CMV mature reference sites show a sustainable median FCI for the

Table 5. Comparison of differences in median FCI values between the CMV (MO) and LMAV (AR, LA, MS) restoration and reference sites

Function	CMV	LMAV
Temporary storage of surface water	0.00	0.25
Retention and retarding the movement of ground water	0.00	0.00
Cycling of nutrients	0.06	0.30
Removal and sequestration of elements and compounds	0.06	0.12
Retention of particulates	0.00	0.25
Organic carbon export	*	*
Provide environment for native plant community	0.07	0.35
Promote wildlife habitatat	0.16	0.06

* Because the median FCI for the restoration sites was greater than for the reference sites in the CMV, a direct comparison for this function is not appropriate.

Table 6. Comparison of median FCI values among vegetation stages for reference and restoration sites, CMV (MO).

Function	Agriculturally altered sites	Early successional	Mature forested	Restoration sites
Temporary storage of surface water	0.42	0.32	0.56	0.42
Retention and retarding the movement of ground water	1.00	0.66	1.00	1.00
Cycling of nutrients	0.10	0.56	0.58	0.44
Removal and sequestration of elements and compounds	0.74	0.39	0.88	0.74
Retention of particulates	0.42	0.32	0.56	0.42
Organic carbon export	0.25	0.18	0.87	0.32
Provide environment for native plant community	0.32	0.51	0.85	0.43
Promote wildlife habitat	0.17	0.63	0.70	0.40

ground water storage function (table 7) that may indicate a significant difference between the medium FCI values for this function.

Playa Wetlands, the High Plains (KS)

Restoration sites

Six wetland ecosystem functions were assessed for 10 restoration sites within the HP of Kansas (table 2). The change in mean FCI between T0 and T1 for each function is shown in table 3. Site FCI values at T0 and T1 are shown in appendix C, figures 25 – 28. Functional capacity increased for all four groups of functions (table 3). Mean FCI increased from 0.60 at T0 to 0.81 at T1 for the *Maintains characteristic hydrologic regime* function, a net increase of 0.21 (table 3). Site FCI values ranged from 0.55 to 0.70 at T0 and from 0.70 to 1.00 at T1 (app. C, fig. 25). Site FCI values ranged from 0.33 to 0.60 at T0 for the two biogeochemical functions assessed (*Maintains elemental cycling* and *Retention of particulates*).

At T1, site FCI values ranged from 0.63 to 0.93 (app. C, fig. 27). The mean site FCI change was 0.30 for the two functions (table 3). The mean FCI increased from 0.37 to 0.69 for the *Maintains characteristic plant community* function, the second greatest increase in mean FCI of the four functional groups assessed for this wetland subclass (table 3). The *Wildlife* functional group, addressing within wetland habitat structure, food web support, and habitat interspersion and connectivity among wetlands, showed the greatest increase in mean FCI values, increasing from 0.23 to 0.68 for a net FCI change of 0.45 (table 3). It was also the functional group with the lowest mean FCI value calculated after restoration was initiated.

Reference sites vs. restoration sites

Twenty reference sites were assessed using the same functions assessed for the restoration sites. The twenty sites were comprised of 10 farmed playa wetlands and 10 nonfarmed playa basins. When all reference sites are combined, only the plant community function median FCI is lower for the restoration sites (table 4). The median FCI value for the biogeochemical functions is the same.

When the reference sites data set, however, is separated into farmed and nonfarmed and is compared to the restoration sites, the median FCI values for the hydrologic regime and the plant community functions were similar between the nonfarmed playa wetlands and the restoration sites (table 8). For the remaining two functional groups, the median FCI values for the restoration sites were intermediate between the nonfarmed playa wetlands and the agriculturally altered playa wetlands (table 8).

Table 7. Comparison of median FCI values between CMV (MO) and LMAV (AR, LA, MS) mature forested, low-gradient riverine reference sites.

Function	CMV*	LMAV**
Temporary storage of surface water	0.56	0.94
Retention and retarding the movement of ground water	1.00	0.32
Cycling of nutrients	0.58	0.78
Removal and sequestration of elements and compounds	0.88	0.86
Retention of particulates	0.56	0.94
Organic carbon export	0.87	0.98
Provide environment for native plant community	0.85	0.71
Promote wildlife habitat	0.70	0.62

* Median FCI values calculated from 6 reference sites.
** Median FCI values calculated from 15 reference sites.

Discussion

The results from this pilot demonstrate that, using the available interim and regional guidebook HGM models as the assessment methods, there was an increase in wetland functional capacity index values for most functions assessed at the sample restoration sites for the three wetland subclasses. However, the results also indicate that the mean functional capacity index for each subclass function lies well below that identified as the highest sustainable functional capacity (i.e., 1.00). Proportioning individual site FCI values at T1 for each subclass function also shows that a small percentage of sites achieved an FCI of 1.00 for just one function (table 9).

Although some sites may eventually achieve a sustainable functional capacity for all functions characteristic of a subclass, others may never achieve this level over the short or long term. Several possible reasons for this—vegetation structure of restoration site; degree, type and frequency of landscape alterations; and HGM model refinement and validation—are discussed below. Other reasons such as restoration techniques applied and landowner management goals for the site also play a role. The type of restoration techniques and degree to which they were applied undoubtedly affect functional capacity levels.

This study was not designed nor intended to evaluate effectiveness of site design or implementation. However, data derived from this study indicates an evaluation of successful restoration techniques appropriate to a wetland subclass is critically necessary to achieve and maintain sustainable functional conditions. Periodic evaluation of restoration techniques is a necessary component to successful restoration implementation and should be conducted routinely as required by National Conservation Practice No. 657 (Wetland Restoration operation and maintenance standards).

The Conservation Practice also requires a pre- and post-assessment of the target restoration site, and use of the HGM or a similar assessment procedure. However, the HGM models or other similar assessment method used must be developed in such a way that historic as well as current landscape conditions are assessed. An assessment of the extent to which anthropogenic alterations have changed the landscape since historic times can determine whether sustainable functional capacity is even achievable or can provide a reference to require special restoration techniques to address alterations caused by historic or current land use practices.

Landowner preferences as to the type of wetland subclass that eventually results and the concerns of adjacent landowners with hydrologic restoration are also factors that influence recovery to a sustainable functional condition. One or both of those factors can negate successful functional restoration of specific wetland subclasses and result in a costly project with a high maintenance requirement. A better understanding of the wetland subclass, and what societal values a landowner may receive through restoration of that subclass, is a necessary precursor to achieving successful restoration of any wetland subclass.

Table 8. Comparison of median functional capacity index values among restoration sites, reference sites, and agriculturally altered playa wetlands, the High Plains (KS)

Function	Agriculturally altered playas	Nonfarmed playas	Restoration sites
Maintain characteristic static or dynamic storage, soil moisture, and ground water interactions	0.66	0.85	0.80
Elemental cycling and retention of particulates	0.48	0.97	0.73
Plant community	0.48	0.96	0.68
Faunal habitat, food webs and habitat interspersion	0.31	0.75	0.70

Table 9. Percentage of restoration sites grouped by functional capacity index value categories at T1

Subclass	Function	Percent sites with functional capacity index at T1 = 1.00	T1 ≥ 0.75 < 1.00	T1 ≥ 0.50 < 0.75	T1 ≥ 0.25 < 0.50	T1 ≥ 0.10 < 0.25	T1 < 0.10
Temporary and seasonal prairie pot-holes, NPPR	Static surface water storage	0.0	69.2	23.1	7.7	0.0	0.0
	Dynamic surface water storage	0.0	61.5	7.7	0.0	0.0	30.8
	Nutrient cycling	0.0	23.1	61.5	15.4	0.0	0.0
	Removal of imported elements and compounds	0.0	38.5	53.8	7.7	0.0	0.0
	Retention of particulates	0.0	76.9	15.4	0.0	7.7	0.0
	Provide for characteristic plant community	0.0	30.8	69.2	0.0	0.0	0.0
	Habitat structure within wetland	0.0	61.5	31.8	7.7	0.0	0.0
	Habitat interspersion and connectivity among wetlands	0.0	76.9	15.4	7.7	0.0	0.0
Forested, low-gradient riverine wet-lands, LMAV	Temporary storage of surface water	0.0	0.0	73.3	26.7	0.0	0.0
	Retention and retarding the move-ment of ground water	13.3	0.0	0.0	86.7	0.0	0.0
	Cycling of nutrients	0.0	0.0	0.0	100.0	0.0	0.0
	Removal and sequestration of elements and compounds	0.0	0.0	100.0	0.0	0.0	0.0
	Retention of particulates	0.0	0.0	73.3	26.7	0.0	0.0
	Organic carbon export	0.0	0.0	13.3	86.7	0.0	0.0
	Provide environment for native plant community	0.0	0.0	20.0	60.0	20.0	0.0
	Promote wildlife habitat	0.0	0.0	100.0	0.0	0.0	0.0
Forested, low-gradient riverine wet-lands, CMV	Temporary storage of surface water	0.0	0.0	26.7	46.7	26.7	0.0
	Retention and retarding the movement of ground water	60.0	0.0	6.7	26.7	6.7	0.0
	Cycling of nutrients	0.0	0.0	20.0	60.0	20.0	0.0
	Removal and sequestration of elements and compounds	0.0	46.7	20.0	6.7	0.0	26.6
	Retention of particulates	0.0	0.0	26.7	46.7	26.7	0.0
	Organic carbon export	0.0	0.0	26.7	33.3	33.3	6.7
	Provide environment for native Plant community	0.0	0.0	26.7	73.3	0.0	0.0
	Promote wildlife habitat	0.0	0.0	33.3	66.7	0.0	0.0
Playa wet-lands, High Plains	Maintain characteristic static or dynamic storage, soil moisture, and ground water interactions	100.0	70.0	20.0	0.0	0.0	0.0
	Elemental cycling and retention of particulates	0.0	20.0	80.0	0.0	0.0	0.0
	Plant community	100.0	100.0	80.0	0.0	0.0	0.0
	Faunal habitat, food webs, and habitat interspersion	0.0	20.0	70.0	100.0	0.0	0.0

Restoration site vegetation structure

Kusler and Kentula (1989) hypothesized that restored wetlands will functionally rebound at a relatively fast rate, and certainly faster than created wetlands. In addition, herbaceous wetlands, such as prairie pothole and playa wetlands, may exhibit a faster rate of functional change than sites targeted for forested wetland restoration. Comparative data on rate of functional capacity restoration relative to the dominant wetland vegetation life form is lacking for this study.

Data from the study, however, can be used to compare mean FCI values between wetlands of similar ages dominated by woody and herbaceous vegetation for the nutrient cycling function. This function is common to all subclasses sampled. An analysis indicates that at T1, the mean FCI values are similar for herbaceous (NPPR) and woody (LMAV, MO) restoration sites which are 1 to 2 years of age (table 10). However, the herbaceous restoration sites that are from 5 to 8 years of age at T1 have a mean FCI value of 0.71 compared to a mean of 0.39 for woody restoration sites which are 5 to 6 years of age at T1.

Other variables, such as soil structure, influence the calculation of the FCI for this biogeochemical function. This example illustrates, however, that vegetation structure is an important consideration when monitoring changes in functional condition for different wetland subclasses. Wetlands dominated by woody vegetation (forested and scrub-shrub wetlands) will often take more time to yield high FCI levels.

The wetland restoration strategy in the LMAV is to return cleared agricultural lands to what were historically mature forested wetlands. As such, many of the HGM model variables measured are indicators of mature forested wetland systems, not the young (those that have a mean age of 4.5 years in Missouri and 2 years in Arkansas, Louisiana, and Mississippi) sapling- and herbaceous-dominated wetlands that comprise the restoration wetlands. It will take many years—perhaps 60 or more years—for conditions similar to mature forested, low-gradient riverine wetlands to develop.

Site index values for several tree species common to the forested, low-gradient riverine restoration sites are shown in table 11. These values show the potential productivity of soil characteristic of these wetland systems to support the growth of these species and, theoretically, the restoration of forested, low-gradient riverine wetland systems. However, the ability of the soil to support these species, and eventually the wetland systems, depends in large measure on the degree to which soil structural and chemical characteristics have been modified through cropping practices.

Although the current absence of mature forested wetlands results in less than sustainable FCI values for the Central and Lower Mississippi Valley, the location of restoration sites funded by the Wetland Reserve Program (WRP) may be a contributing factor to the long-term conservation of forest-breeding landbirds in the region. The USGS, Biological Resources Division and the U.S. Fish and Wildlife Service, Lower Mississippi River Joint Venture, Vicksburg, Mississippi, have developed a geospatial model that identifies reforestation habitat priorities for forest-breeding landbirds in the Mississippi Alluvial Valley (MAV) (Twedt and Uihlein 1999). One of the objectives in developing the model was to “assess the performance of recent enrollments in the [WRP] with regard to their location relative to priority reforestation habitat.” Approximately 40 percent of WRP lands enrolled in the MAV that were digitally available at the time or could be digitally created in a short period for model use were included in the analysis.

Results showed that, with the limited spatial WRP data available at the time, the distribution of WRP acreage was in the higher reforestation priorities (Twedt and Uihlein 1999). Eighty-eight percent of WRP lands were above the average priority category for the MAV and 57 percent were distributed in the highest three reforestation priority categories. The high percent of land in the highest reforestation priorities is likely the result of the spatial juxtaposition of WRP land and adjacent existing forest (Twedt and Uihlein 1999). Although this information does not specifically address the forested, low-gradient riverine subclass, approximately 55 percent of WRP land in Mississippi, Louisiana, and Arkansas may be in this subclass.[1]

Data from the geospatial model shows that the greatest amount of acreage in Mississippi and Arkansas is

[1/] Estimate is not derived from documentation but based on estimation by NRCS.

Table 10. Comparison of mean FCI values over time for nutrient cycling function, herbaceous (NPPR) and forested (LMAV, MO/IL) restoration sites

Vegetation type	Age	T1	Age	T1
Herbaceous	1	0.50	7	0.68
(NPPR)	1	0.36	7	0.68
	2	0.30	8	0.78
			8	0.68
			8	0.71
			8	0.72
		0.39 (mean)		0.71 (mean)
Forested	1	0.45	5	0.54
(CMV-MO/IL)	2	0.36	5	0.22
	2	0.45	6	0.41
	2	0.44	6	0.29
	2	0.48	6	0.21
			6	0.24
			6	0.45
			6	0.40
			6	0.51
			6	0.64
		0.44 (mean)		0.39 (mean)

TABLE 11. Site index range for tree species common to soil of the forested, low-gradient riverine subclass, CMV and LMAV

Common name	Scientific name	Site Index Range (height in feet)
Eastern cottonwood	*Populus deltoides*	90-120
Tulip poplar	*Liriodendron tulipifera*	115
Sweet gum	*Liquidambar styraciflua*	90-110
Water oak	*Quercus nigra*	90-110
Texas red oak	*Quercus texana*	90-110
Shumard's oak	*Quercus shumardii*	105
Cherrybark oak	*Quercus pagoda*	90-100
Willow oak	*Quercus phellos*	90-100
Loblolly pine	*Pinus taeda*	90-95
Pin oak	*Quercus palustris*	80-100
Green ash	*Fraxinus pennsylvanica*	78-100
American sycamore	*Platanus occidentalis*	80

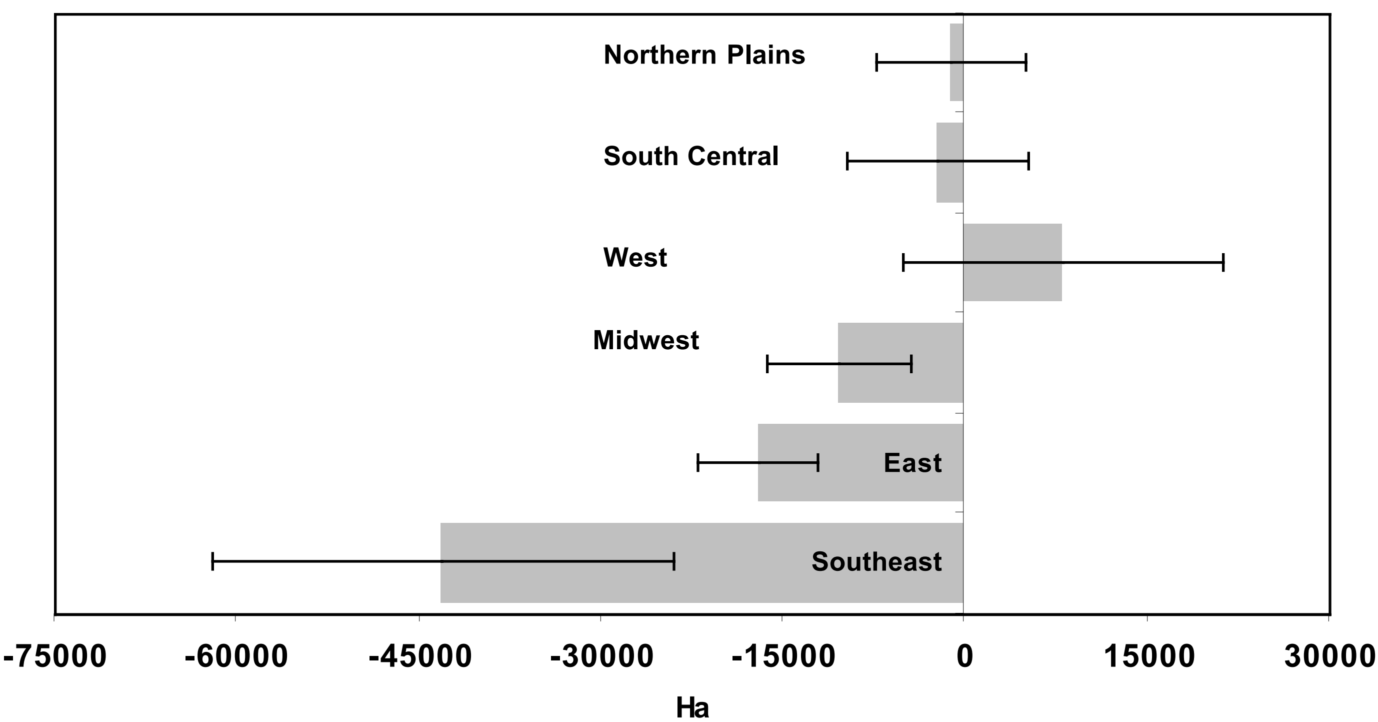
distributed in the Moderate Priority, Priority, and Moderately High Priority reforestation categories. Acreage in Louisiana is highest in the Priority, Moderately High Priority, and High Priority reforestation categories. The distribution of acreage in Missouri is greatest in the lower reforestation categories (Twedt and Uihlein 1999). Although the state acreage distributions are not limited to just WRP land, they do provide a broader context in which to evaluate restoration of forested wetland relative to breeding landbirds. An updated digital WRP layer could enhance the findings from this initial analysis and provide a more robust analysis of the contribution of WRP to breeding landbird conservation in the MAV. Additionally, collaboration with the U.S. Fish and Wildlife Service's MAV Joint Venture and the Biological Resources Division of the USGS could also involve developing a suite of geospatial models that identify multiple landscape features to assist in restoration of the MAV as a functioning landscape.

Landscape alteration

Landscape alteration is one of the most insidious factors responsible for change in wetland extent and function. The most current wetland trends, derived from the 1997 National Resources Inventory (NRI), show a net change in wetland extent of approximately 65,934 hectares during this period (USDA, NRCS 2000a). Gross acreage losses were highest in the Southeast, South Central, and Midwest NRCS administrative regions, although the overall net acreage change during this period shows that the Southeast, East, and Midwest are still experiencing the highest net acreage losses (fig. 2).[2]

2/ The net change in wetland extent for each region includes states and wetland subclasses other than those sampled. Arkansas, Louisiana, and Mississippi are included in the South Central and Southeast Regions, respectively. Missouri and Illinois are in the Midwest Region. The Northern Plains Region includes North Dakota and Kansas.)

Figure 2. Net change in wetland extent between 1992 and 1997 within NRCS administrative regions*



* See text for states associated with a particular region of interest to this study. Error bars represent the 95% confidence interval. Confidence intervals for the Northern Plains, West and South Central regions are wider than estimates because they include 0, hence the net change may not be different than 0 (USDA, NRCS 2000a).

These areas historically had extensive Palustrine wetlands, a type of wetland that includes the HGM wetland subclasses sampled. Palustrine wetlands continue to dominate the landscape, accounting for almost 95 percent of wetlands sampled by the 1997 NRI. Of the 42.8 million hectares of Palustrine wetlands in the conterminous United States, 10.3 million occur on cropland, pastureland, and rangeland (table 12). These wetlands have been and continue to be exposed to a range of anthropogenic activities that modify the landscape, resulting in significant effects on their ability to maintain or achieve a high sustainable functional condition.

Similar to other Palustrine wetlands, the three HGM subclasses sampled have been impacted by anthropogenic activities, particularly agricultural activities. Many of these wetland systems have been drained, ditched, chemically altered, or otherwise manipulated to provide suitable conditions to conduct traditional agricultural practices. The surrounding landscapes have been altered from a matrix of native forest or prairie to one of intensive agrosystems, with subsequent changes in hydrologic and nutrient pathways, increased erosion, introduction of contaminants, and fragmentation of the wetland-landscape interface. Such changes in the historic, pre-European landscape have resulted in wetland acreage losses, functional impairment, and degraded ecosystem conditions. The challenge to restore wetland ecosystems is to recognize that the landscapes in which these systems are embedded have been drastically altered and that restoration must involve not only changes to conditions within the wetland but also in land practices beyond the wetland itself. In many cases, restoration sites may never achieve their highest sustainable functional capacity because landscape alterations are likely to continue in the foreseeable future.

Table 12. Extent of palustrine wetlands in conterminous U.S. by selected broad land cover type (USDA, NRCS 2000b)

Land Cover Type	Palustrine Wetland Extent (million ha)
Cropland	4.1
Pastureland	3.1
Rangeland	3.1
Forest land	26.3

Temporary and seasonal prairie pothole wetlands, NPPR

The existing hydrologic models for temporary and seasonal prairie pothole wetlands in the NPPR mathematically emphasize the importance of surface water elevation in measuring functional capacity for any given wetland. The less-than-sustainable FCI values for the hydrology functions assessed indicate that the original hydrology has not returned following restoration activities. Although almost 62 percent of the prairie pothole restoration sites exceed the mean functional capacity value of 0.74, none of the sites exhibited a sustainable functional capacity value for the *Static surface water storage* function. The relatively lower mean functional capacity value of 0.57 for the *Dynamic surface water storage* function also indicates that hydrologic modifications have hampered hydrologic recovery to a sustainable functional capacity level.

The inability to achieve a sustainable functional capacity in the sampled restoration sites is due to the low scoring of the variable associated with wetland outlet elevation for both of these hydrology functions (ditches were plugged, but outlets were not restored to preexisting topography). The ability of the restoration sites to hold water under either static or dynamic surface water conditions was therefore adversely affected (Rodney O'Clair, NRCS, North Dakota State Office, and Michael Whited, NRCS, Wetlands Science Institute, personal communications).

Closely linked to the effect of outlet elevation on recovery of sustainable hydrologic conditions in restored wetlands is past and current land use. Although planted grasslands currently buffer all of the restoration sites sampled, the previous land use was agriculture. Cultivation in the surrounding uplands accelerates erosion and sediment deposition into the basin wetland (Gleason and Euliss 1998; Martin and Hartman 1986) as a result of increases in surface runoff characteristics (Euliss and Mushet 1996).

The increased sediment in the wetland has several potential impacts on the ability of the wetland to function at sustainable levels. These include
- decreased water storage volume,
- decreased seed bank and propagule viability,

- decreased survivability of invertebrates and invertebrate egg banks,
- buried organic substrates and microbes with inorganic soil, and
- shallower basins that can drastically alter plant community composition and wetland type (Euliss and Mushet 1999; Gleason and Euliss 1998).

Changes in the physical characteristics of the prairie potholes resulting from increased sediment may adversely affect ecosystem processes such as primary productivity, trophic structure support, and nutrient cycling and transformation (Gleason and Euliss 1998).

The ultimate result is that both restored and natural wetlands have a shortened existence on the landscape because of increased sediment deposition from tillage (Gleason and Euliss 1998). If past land use impacts are not considered in restoration design and implementation of temporary and seasonal prairie pothole wetland in the NPPR, then restoration of hydrologic functional capacity will be less than sustainable until such modifications are addressed and remedial restoration action is taken.

The functional loss from complete sedimentation of a restored or natural prairie pothole is obvious, but the effects on wetland functions from the gradual filling in over time are not currently well documented (Gleason and Euliss 1998). The prairie pothole wetland function, *Retention of particulates*, is intended to capture the natural filtering ability of a temporary or seasonal prairie wetland (Lee et al. 1997). The variables measured for the function are designed to reflect this natural filtering capacity at a sustainable functional condition. Any changes within the catchment or the wetland that would accelerate sediment deposition in the prairie pothole resulted in a low FCI value. The median FCI value of 0.75 was calculated at T1. This score indicates that, while most of the sites scored at a relatively moderate level (site FCI values ranged from a minimum of 0.67 to a maximum of 0.88), historic land use practices have likely resulted in less-than-sustainable functional conditions following restoration. None of the restoration sites exhibited a sustainable functional condition regardless of land cover (i.e., on CRP lands) or age.

Although the median FCI for the reference sites was 0.81, 40 percent of the reference sites had an FCI value between 0.04 and 0.15. These low site FCI values indicate that activities within the catchment or wetland have historically impaired or are currently impairing the condition of the wetland to a degree that they can no longer function at sustainable levels to remove particulates from the surrounding landscape.

The results indicate that for sustainable conditions to be achieved in wetlands targeted for restoration, techniques addressing past land use practices (e.g., establishment of properly maintained water control structures, removal of the deposited sediment) are imperatively needed (Galatowitsch and van der Valk 1994). Otherwise, degraded functional conditions remain, regardless of the number of acres counted as restored.

Wetlands in the NPPR that are individually restored within a catchment that is still primarily in agricultural production may continue to experience degradation. Certainly restored wetlands that return to production because of short-term contracts associated with the CRP are at high risk to return to degraded conditions. This is particularly true of temporary prairie pothole wetlands. Euliss and Mushet (1999) found lower diversity and number of aquatic invertebrates in temporary wetlands embedded in catchment basins within intensively farmed agricultural fields compared to those located in grasslands of U.S. Fish and Wildlife Service Waterfowl Production Areas or similar grassland habitats in the Prairie Pothole Region of North Dakota.

Temporary wetlands are considered the most vulnerable wetland type in the PPR, often cropped in all but the wettest years. Such wetlands are routinely tilled and exposed to direct applications of fertilizers, pesticides and herbicides. Temporary wetlands are critical to waterfowl as they are the first wetlands to thaw during the spring when physiological conditions of newly arrived waterfowl are at their lowest. Although drought years have historically limited the number of wetlands available for waterfowl (Grue et al. 1989), the initiation of drainage associated with agriculture during the late 1800's in the Prairie Pothole Region and the continued cropping of temporary wetlands can adversely limit the numbers of temporary wetlands necessary to satisfy the high caloric requirements of waterfowl during the early part of the breeding season.

A median FCI value of 0.69 was calculated for the restoration at T1 for the *Removal of imported ele-*

ments and compounds function. Although this does not reflect a sustainable functional level, the value nevertheless indicates that some degree of removal of elements, including pesticides, is occurring in the restored wetlands sampled. A higher median FCI, 0.80, was calculated for the reference sites. Much like the sediment-filtering capacity addressed above, the variables assessed for the elemental removal function are intended to reflect a gradient of disturbance activities that can either support a sustainable functional capacity or impair it.

Like the retention of particulates, wetlands are often managed as depositories of undesirable elements, with the expected result that they will transform or bury any adverse element without functional impairment. However, input of these materials from the surrounding landscape into wetlands (particularly when it is cropland), either in the water column or attached to sediments, can be potentially lethal to waterfowl and their food sources.

A 1987 study investigated the effects of aerial application of a pesticide, ethyl parathion, on mallard ducklings. The pesticide was applied to sunflower fields immediately adjacent to prairie pothole wetlands. Survival data on mallards that had broods and selected aquatic invertebrates was collected before and after a routine application of the pesticide. Comparable data was also collected for five fenced control wetlands embedded in dense nesting cover or native grassland. Survival was 32 to 65 percent of the ducklings in the control wetlands compared to 3.8 percent for the treated wetlands.

Brain cholinesterase (ChE) activity was severely depressed in all but one of the ducklings in the treated wetlands. Cholinesterase in brain and blood tissues is a reliable indicator of exposure to organophosphate pesticides and may be diagnostic of death resulting from exposure to such contaminants (Hill and Fleming 1982). Invertebrates in the treated wetlands were also negatively impacted. Amphipod survival was significantly reduced over a 25-day period following application (Grue et al. 1989).

In another study, Dieter et al. (1995) investigated the lethal and sublethal effects of phorate, an organophosphate pesticide, on mallard duckling survivability. Phorate is commonly used in fields and tilled wetland basins in the Prairie Pothole Region, and is toxic to birds (Dieter et al. 1995). Phorate can be found concentrated in wetland sediments, in the wetland water column, or directly in agricultural runoff from fields. The study documented the acute lethal exposure of phorate to mallard ducklings as well as the sublethal effects of inhibited brain and blood ChE in mallard ducklings that also results in altered behavior and survival. In addition, phorate can be lethal to invertebrates and absorbed by plant tissues, both major food items of mallard ducklings (Dieter et al. 1995). Because phorate is representative of many organophosphates used in the Prairie Pothole Region, the continued use of such pesticides within restored and natural wetland basins and in surrounding fields will likely continue to pose an adverse risk to mallard and other waterfowl young that use them.

The range of estimated site FCI values at T0 was lower for the *Habitat structure within wetland* function compared to that for the *Habitat interspersion and connectivity among wetlands* function for the NPPR sites. The difference between the mean FCI values calculated for conditions following the onset of restoration for both functions (0.71 within-wetland habitat structure function vs. 0.78 for the spatially explicit habitat function) does not appear to be significant. But the lower mean FCI value for within-wetland habitat structure, coupled with the lower estimated site FCI range for this function, may indicate that restoring the conditions necessary to support wildlife habitat within temporary and seasonal pothole wetlands is somewhat more difficult to achieve because of past land use of these wetland types.

A breakdown of estimated and calculated FCI values by wetland type (temporary and seasonal) for the restoration sites indicates that there are relatively small differences in the mean FCI values between wetland types at T0 for the within-wetland habitat or among-wetland interspersion and connectivity functions. The difference at T1 between the within-wetland habitat mean FCI value and the spatial configuration function mean FCI value assessed for temporary wetlands is somewhat greater (table 13). The mean FCI at T1 for the within-wetland habitat function is lower for temporary wetlands than the among-wetland interspersion and connectivity function mean FCI for temporary wetlands.

In addition, the within-wetland habitat mean FCI at T1 is lower for the temporary wetlands than for the seasonal wetlands. These results again suggest that past agricultural activities within temporary wetlands impair the ability of restored temporary wetlands to achieve sustainable functional capacity over time. The results may further suggest that the variables used to assess conditions of within-wetland habitat structure are more sensitive to past land use within the wetland or more easily capture its effects than the variables used to assess the spatial configuration of the wetland complex.

The variables included in the *Habitat interspersion and connectivity among wetlands* function are intended to capture the importance of the prairie wetland complex in the landscape, as well as the importance of maintaining habitat structure and connectivity of uplands surrounding wetland complexes. This is extremely critical for waterfowl, many of which require wetland complexes connected by upland grasslands for nesting (Galatowitsch and van der Valk 1994; Swanson and Duebbert 1989). Successful movement by hens and broods from the nest to wetland basins, to feed and escape upland predators, is dependent upon contiguous and dense upland herbaceous cover. The restoration of prairie pothole wetlands and the surrounding uplands via the Conservation Reserve Program may enhance the nesting success of waterfowl. In the NPPR of North Dakota and Minnesota, Kantrud (1993) evaluated the nest success of dabbling ducks from 1989 to 1991 on CRP land compared to that on U.S. Fish and Wildlife Service Waterfowl Production Areas (WPA). Kantrud found that waterfowl nest success was 23.1 percent during this period on CRP lands in areas of high wetland density compared to 8.2 percent on WPA lands of similar cover.

Table 13. Temporal and spatial comparison of mean functional capacity index values of habitat functions between temporary and seasonal prairie pothole restoration sites, NPPR

	Within-wetland habitat		Among-wetland interspersion and connectivity	
	Temporary	Seasonal	Temporary	Seasonal
T0	0.17	0.12	0.33	0.28
T1	0.68	0.75	0.78	0.78

The data suggest that the higher nest success rate on CRP lands may have been related to enhanced protection from predators, particularly for large dabbling duck species such as mallard and gadwall whose nests on CRP lands were located further from semi-permanent wetlands than on WPA sample sites. In addition to waterfowl, several studies have focused on the value of CRP fields to grassland birds (Johnson and Koford 1995; Johnson and Schwartz 1993) but spatial linkages between CRP upland and wetland have not been addressed.

Forested, low-gradient riverine wetlands, CMV and LMAV

Changes in historic hydroperiods affect the ability of restored sites to reach a sustainable functioning capacity. The less than sustainable functional capacity values calculated for the *Temporary storage of surface water* function before and after restoration was initiated on the forested, low-gradient riverine wetlands of the LMAV indicate that hydrologic modifications outside the wetland under restoration are still hampering sustainable functional capacity. Similarly, only 36 percent of the combined 30 restoration sites from the LMAV and CMV exhibited sustainable functional capacity values for the *Retention and retarding ground water* function. This indicates the importance of the linkage between offsite hydrologic modifications and the ability to restore sustainable functional capacity.

Although none of the restoration sites sampled in the CMV and LMAV involved construction of water control structures to restore wetland hydrology, this management technique is commonly used to restore surface water conditions to a wetland. However, construction of water control structures at restoration sites does not guarantee a return to sustainable hydrologic conditions. Placement of a water control structure can result in development of a wetland in the *Depressional* functional class, particularly if the structure is accompanied by removing the wetland soil to deepen the wetland below the original surface level. Also, management of the structure may not mimic naturally occurring hydroperiods. This can result in less than sustainable hydrologic functioning. Management to mimic natural hydrologic variability characteristic of the subclass can often be difficult, if not impossible, because of land practices (levee and ditch construction/maintenance) on and off site.

Hydrologic restoration is undoubtedly one of the most difficult functions to achieve in the CMV and LMAV because of the extensive network of drainage channels and levees. King and Allen (1996) suggest that where watershed-wide hydrologic restoration is not feasible, a complex of impoundments similar to greentree reservoirs may restore forested riverine wetlands dependent on periodic flooding. However, implementation and management of such systems require a long-term commitment to emulate natural hydrologic variability, as well as routine monitoring to ensure that the management is in fact restoring sustainable levels of all riverine wetland functions (King and Allen 1996).

Playa wetlands, the High Plains

As a result of the ephemeral nature of wetland playa surface water, the complexities of state water rights and public vs. private use of water in the High Plains, and the historic modifications of playas to support agricultural interests in the Plains, the continued existence of playas depends on their ability to provide sustainable functions that will accommodate a variety of societal uses.

The focus of most playa research has been on the use of playas for sources of irrigation water and feedlot waste retention. Also considered is their importance in recharge of the Ogallala aquifer, primarily on the Southern High Plains (SHP) (Scanlon et al. 1994; Wood and Sanford 1994; Lehman 1972). Effects of feedlot waste retention within playas have focused on potential contamination effects on the Ogallala. The almost impermeable clays of playa bottoms, along with the further sealing from organic feedlot waste material, have shown playas to be potentially effective long-term traps of such material (Stewart et al. 1994).

Stewart et al. (1994) found that most nitrate was removed in three playas used for long-term feedlot waste storage and treatment at a depth of 10 feet, with high accumulation occurring in the upper 1 foot of the playa clay. There was greater leaching of nitrate into the ground water above the playa clay where soil become coarser and more permeable. Presumably, nitrate is being denitrified within the saturated soil of the playa.

Haukos and Smith (1996) suggest, however, that nutrient cycling and transformation processes are poorly understood in playas and that further research is warranted. In addition, a more robust data set is needed to further determine the long-term viability of playas for animal feedlot waste storage and treatment (Stewart et al. 1994). Investigation of feedlot waste retention effects on biological systems is also warranted.

Similar to the temporary and seasonal prairie potholes of the NPPR, playas in cultivated watersheds of the SHP have greater sediment deposition than do playas in rangeland watersheds (Luo et al. 1997). Playas in cropland had 8.5 times more sediment accumulated than did those within rangeland watersheds. In particular, cropland watersheds that have medium-textured soil had significantly higher depths of sediment accumulation in playas than did cropland watersheds that have fine-textured soil, or rangeland with either soil texture type (Luo et al. 1997). Increasing inputs of sediment into playa wetlands will eventually obliterate their presence on the High Plains landscape. In the interim, the wet phases of the playa hydroperiod will become increasingly shorter. The result will be forms of increasingly intensive and expensive restoration and management to maintain wetland functions, particularly to support critical overwintering, migratory and breeding habitat for numerous species of waterfowl and shorebirds as well as the sandhill crane, *Grus canadensis* (Anderson and Smith 1998; Smith 1994).

Croplands properly treated and maintained with effective conservation practices, as well as those enrolled in the Conservation Reserve Program, exhibit decreased rates of sedimentation (Luo et al. 1997). However, lack of effectively maintained or implemented conservation practices and declines in amount of CRP land will ultimately reduce the functional longevity of playa wetlands on the High Plains.

Hydrogeomorphic method model refinement and validation

The hydrogeomophic method (HGM) approach to wetland functional assessment was selected for use in this effort for several reasons. The approach provides a recognized method of assessing wetland functional capacity (Smith et al. 1995). The method employed in HGM focuses on the functional capacity of wetlands and not on their societal values, thereby allowing a more scientific approach to wetland functional assessment (Brinson 1996). As stated earlier, the assessment provides a numerical score of the functional capacity for each function assessed in a wetland, thereby avoiding ambiguity involved in ratings of High, Moderate, and Low. In addition, NRCS has already invested in the development of interim HGM models as required in Swampbuster to evaluate whether agricultural impacts to wetlands are minimal and the ratio of compensatory mitigation required to offset impacts. And, most importantly, the method was selected because the approach directs the development of functional variables to include not only those within the wetland itself but also the characteristics of the landscape that can affect the relative functional condition of a wetland (Smith et al. 1995).

The models used to assess prairie pothole wetlands are assembled in a draft regional guidebook (Lee et al. 1997). An interagency team identified reference wetlands and collected data from them to develop the index used to score functional capacity for each function assessed. As discussed above, the landscape surrounding the prairie wetlands and the past and present land uses of the landscape exert a profound effect on the functional capacity of temporary and seasonal prairie pothole wetlands in the NPPR. However, validation of the models has not been completed to date although efforts to do so are underway (Ned Euliss, personal communication). Validation of the models could result in changes in the FCI values reported, as well as a change in the number of variables measured or the scoring of any variable for any given function.

Much like the prairie potholes to the north, the playa wetlands of the High Plains are embedded in a landscape dominated by agriculture. Playas have historically been used as sources for livestock watering and, as the only source of surface water, for irrigation water. The scientific knowledge of playas, however, is much less well known than their uses.

The HGM models used to assess the playa wetlands are adaptations of the prairie pothole models, but were developed without reference wetlands data, much like the models for the forested, low-gradient riverine wetlands. However, the forested wetlands of the LMAV have been studied for a longer time than the playa wetlands of the High Plains; hence there is a better understanding of their ecology and the potential effects from disturbances and alterations.

The playa HGM models were developed with the information available at the time. In addition, the models were developed to evaluate minimal effects to playa wetlands from agricultural activities per Swampbuster. Establishment of reference wetlands for the High Plains playa wetland subclass and the forested, low-gradient riverine wetland subclass in the CMV and LMAV would improve model outputs.

In addition, inclusion of functional variables that reflect different development stages (based on vegetation structure and/or other physical features) within the reference data set for both wetland subclasses would provide a finer-tuned index of functional capacity for each. Although lacking the benefit of reference data, the results generated from application of the interim HGM models still provide a documented measure of change in wetland functional condition for playa wetlands of the High Plains and forested, low-gradient riverine wetlands of the CMV and LMAV.

Recommendations

The results of the National Wetlands Functional Assessment Pilot indicate that several methods are available for NRCS to monitor change in wetland functional condition. However, factors such as scale of assessment, the cumulative effects of anthropogenic alterations on the functional condition of wetlands within a landscape, and the ability to capture and accurately report agency performance in wetlands restoration and management argue for a multifaceted approach to assess wetland functional change. Furthermore, the results suggest that NRCS should take additional steps that would assist field staff in attaining the highest functional condition possible on wetland restoration sites.

The first three recommendations address methods to assess and monitor change in wetland functional condition. The methods are listed individually, as each provides different types of information relative to wetland functional condition. Together, they provide a comprehensive assessment approach. Periodic evaluation of the methods is needed to determine their relevance to the NRCS mission and strategic plan goals. The last three recommendations focus on activities that would enhance wetland restoration activities at the field level. These recommendations would also better integrate the application of restoration activities in the field with assessing change in wetland functional condition at a national level.

1. Broad-scale assessment

The Resource Assessment Division (RAD), Soil Survey and Resource Assessment Deputy Area, should continue to develop and routinely assess broad-scale indicators of potential degradation to wetland ecosystem functioning, using GIS, remote sensing data, and modeling. Although the assessments could focus on a specific HGM subclass throughout a landscape, the purpose of such assessments would be to address degradation processes potentially affecting all wetlands on a landscape regardless of subclass (HUC-8, MLRA's, Bailey's Ecoregions). For example, research underway in the NPPR (USGS, Northern Prairie Wildlife Research Center) is seeking to identify multiple-scale indicators of wetland condition. Using these preliminary results, NRCS could begin to periodically assess several of these indicators, or others not yet identified through current research efforts. For temporary and seasonal prairie pothole wetlands, NRCS could potentially monitor the following three indicators of change resulting from landscape degradation processes identified by this research:

- net loss of and ratio of temporary wetlands on the landscape
- potential change in basin inorganic sediment levels
- shifts in vegetation zonal patterns

The ability to determine a reference condition for each of these will require coordination with the Northern Prairie Wildlife Research Center (USGS, BRD); Environmental Protection Agency, EMAP; Wetlands Science Institute, NRCS State Offices in the NPPR; and others involved in research affecting temporary and seasonal wetlands in the NPPR.

In lieu of a site-specific FCI derived from the HGM models, absolute measurements would be derived and tracked. Several methods will be required. For example, a net loss of temporary wetlands could be derived from new data elements added to the NRI or a geospatial wetland data set. Estimated number of temporary wetlands and the ratio of temporary: seasonal and temporary:semipermanent could be derived, based on historic published records or via construction of potential historic distribution (see below).

Potential changes in wetland inorganic sediment levels could be derived from modeling sediment yield from uplands into the basins under different cropping or grazing practices and crop or vegetation covers. A baseline level could be determined to calibrate the model from *in situ* sediment deposition in selected basins. Initial work by Kantrud and Newton (1996) indicates that several potential wetland vegetation indicators are linked to agricultural activities; examples are changes in zonation patterns and extent of unvegetated wetland surface. With appropriate ground-truthing to calibrate baseline conditions and remote sensing data, such indicators could provide another measure of the extent to which degradation processes affect functioning of wetland ecosystems.

2. Site-specific assessment

The broad-scale approach described above provides a means of monitoring potential degradation to wetland functioning. However, it does not provide a means to assess changes in wetland functional condition at a finer scale and for specific subclasses. This could be done for selected geographic regions periodically, using a site-specific approach, such as the HGM method. Periodically assessing wetland condition at a site-specific scale provides a mechanism to evaluate local perturbations to wetland ecosystem functioning that could not be easily detected or are not evident at a broad scale. Local-scale monitoring would also provide a numerical basis for tracking functional condition for specific functions through the continued use of the HGM method.

Although the Wetland Restoration training sponsored by NRCS does include monitoring of wetland function at a site-specific level, results from application of this training are not required to be reported. Evaluation and required reporting of functional condition at this scale could be entered into the *NRCS Performance and Results Measurement System* for specific subclasses, although some refinement of the reporting system is needed to accommodate such data. A subsample of wetland restoration sites funded by USDA programs and reference wetlands benefiting from other conservation practices would provide a more complete picture of NRCS wetland management activities than is currently available.

To initiate this level of assessment, an interdisciplinary workgroup of approximately 8 to 12 individuals should be established within NRCS, led by the Wetland Science Institute. The workgroup would coordinate and establish wetland reference data sets for the forested, low-gradient riverine wetlands of the CMV and LMAV and playa wetlands of the High Plains. The feasibility of including other wetland subclasses (e.g., Delmarva wet hardwood flats, slope wetlands of the western United States, salt marshes of the New England Coastal Plain) should also be explored.

The workgroup should focus on refining existing NRCS interim HGM models for only those functions that could be assessed over time using primarily remote sensing (satellite imagery, high-resolution aerial photography) and limited field data collection (although field collection will dominate initially to establish the reference data set to develop appropriate index scores). The workgroup should coordinate with other agencies and institutions to build upon ongoing studies and Agency expertise.

Although the HGM assessment procedure ultimately can provide a relatively useful assessment tool, there are critical limitations associated with current protocol to build the functional capacity index models. Often there is little, if any, information regarding hydrologic or biogeochemical processes, and the interagency team members must either rely on generic literature or use best professional judgement. Much energy is devoted to development of variables and models for these functions, yet the reality is that the models often represent what is believed to be occurring, not what is documented. To remedy this often tedious and costly approach, variable selection and measurement should be at as generic a scale as the interagency team feels comfortable, until such time that documentation allows refinement. In this regard, it is suggested that as reference data is collected, gaps in knowledge should be identified and appropriate steps be taken to work with the research community to cost-effectively address those gaps.

Once the reference wetlands are established and selected functional models developed and calibrated for the chosen subclasses, a statistically derived subsample of restoration and reference sites should be selected for assessment. FCI values for the selected functions assessed could then be reported. Periodic assessment of the wetlands could occur at a frequency tailored to the age of the wetland (for restoration sites), wetland subclass, and the functions to be assessed.

3. National resources inventory wetland data elements

The NRI provides a means of collecting data across a spectrum of wetland types (vs. HGM subclass). Information on data elements, collected from one NRI to the next, is used to derive status and trends data. For wetlands this has been primarily limited to acreage changes. However, authorizing legislation for the NRI includes statements about the quantity and quality of habitat for wildlife and fish, yet the NRI captures little about habitat quality, particularly with regard to wetlands. The following wetland elements are recommended to be added to future NRI efforts:

- Soil data for wetland sites to document the specific hydric criteria that are expressed at each sample point. This will greatly assist in more accurately identifying what is or is not a wetland for both acreage and functional assessment purposes.
- Quantitative measures of soil erosion or sediment delivery to wetland sites that occur in agricultural settings (both from USLE, and perhaps by measuring the sediment depth on subsamples).
- Vegetative transect data for subsamples of representative wetland sites (such as has occasionally been done under Swampbuster).
- Extant factors such as grazing, pesticide runoff, traffic, and dumping, which apparently degrade wetland quality or impede the achievement of functional potential.

4. Wetland functional subclass identification

Identification of functional subclasses is needed to assist field staff in determining the appropriate type of wetland for restoration based on geomorphic setting, hydrodynamics, and hydrologic source(s). Such guidance would provide several beneficial results including the ability for NRCS to reliably link its on-the-ground wetland restoration activities and national efforts to identify and track changes in wetland condition. Just as importantly, such a mechanism would enable NRCS and its partners to implement appropriate restoration techniques for specific wetland subclasses. This would improve the treatment effects of wetlands on the landscape, particularly when combined with other land treatment practices.

Although issues such as concerns of adjacent landowners and specific enhancement features desired by a landowner are also valid factors in designing restoration projects, understanding and identifying hydrogeomorphic features of a wetland subclass provide a sound basis for other proposed modifications. Such guidance should be incorporated into National Conservation Practice No. 657, Wetland Restoration. The Science and Technology and Resource Assessment and Soil Survey Deputy Areas and the Wetland Science Institute should coordinate this effort.

5. Geospatial wetland restoration strategy

In conjunction with identifying wetland subclasses, a more comprehensive restoration approach is also recommended. This is based on work underway in the NRCS New Hampshire State Office (Ammann 1999). The results from this pilot, as well as the current scientific literature, indicate that the ecological condition of the landscape in which wetlands are imbedded affects the functional condition of wetland ecosystems. If wetland restoration is to achieve highest sustainable conditions, part of the solution is to restore and protect as many of the historic native systems as possible, including wetlands, and do it so that functioning ecosystems emerge on the landscape.

In addition, information on the historic and current landscape and the stressors influencing wetland ecosystem functioning provide a way of determining where restoration activities are most needed and likely to succeed. The core of the ecosystem restoration process developed by Alan Ammann, PhD., in New Hampshire uses a Geographic Information System and existing data to

- identify the extent and type of native ecosystems historically present,
- identify the anthropogenic stressors that have altered that environment, and
- identify land ownership as well as protected areas.

The GIS products allow NRCS and its partners an efficient way to integrate many different data sets and develop a management/restoration strategy for the area. Potential management/restoration sites are then inventoried. Management and restoration techniques specific to the ecosystem targeted are then implemented. The sites are monitored to ensure that the activities implemented are successful, and identify any additional management that may be needed. The approach provides a sound basis for ecosystem restoration within a landscape context, and can also be used to help refine wetland subclass identification.

6. Ecological site descriptions

Ecological sites are the interpretive units for rangeland and forest land. Many wetlands exist on these land types and are managed for livestock production. Ecological sites provide a way to inventory, evaluate and manage range and forest lands. The information developed for ecological site descriptions – soils, hydrology, plant community composition and dynamics, disturbance events, anthropogenic alterations, and management interpretations – is similar to that developed for HGM wetland subclass profiles, although use of the information differs between the two. Ecological sites are described based on physical factors, particularly soils, that characterize a specific plant community and the amount of vegetation produced by that community. Because they are institutionalized within NRCS, development of ecological site descriptions for wetlands on range and forest lands would enhance the understanding of ecological site dynamics within a subclass and provide management scenarios resulting from different site characteristics. This information would be useful in designing and managing wetland restoration sites.

For wetlands that are not on rangelands or forest lands, as defined within the NRCS "National Range and Pasture Handbook" 190-VI, similar information is warranted and could be provided in a handbook format for field use. The Science and Technology and Resource Assessment and Soil Survey Deputy Areas and the Wetland Science Institute should initiate collaboration on both efforts.

Summary

Between 1992 and 1996, $274,000,000 was appropriated to the Natural Resources Conservation Service for the Wetland Reserve Program to restore some of the 1.7 million acres historically lost through agricultural activities in the Mississippi River drainage and other regions in the conterminous United States. As of February 2000 during the 20th signup for the USDA Conservation Reserve Program, approximately 63,199 hecartes of cropped wetland were accepted into the program (USDA, FSA 2000). Program contracts are accepted for a 10- to 15-year period, removing and conserving land unsuitable for agriculture and returning, if not all in perpetuity, some portion of it to native or managed grassland and, where feasible, prairie wetland. The PPR in North Dakota, South Dakota and Iowa comprise slightly more than 54% of the total cropped wetland acreage accepted for restoration in the 20th CRP signup (USDA, FSA 2000).

The continued decline of wetland losses—31,995 hectares per year during 1982–92 vs. 105,300 hectares per year from the mid-1970's to mid-1980's (Flather et al. 1999)—and the large acreage numbers accepted into USDA wetland restoration programs (269,325 hectares accepted into the Wetlands Reserve Program for restoration between 1992 and 1999; Flather et al. 1999) are often provided as the barometer of wetland ecosystem recovery. Continuing to focus only on wetland acreage, however, does not adequately address the functional capacity of or degree of impairment to wetland ecosystems. Applying comprehensive and integrated conservation treatments upon the landscape, including wetland restoration, argues for the need to document and quantify change in wetland functional condition.

Although the HGM wetland functional approach provides a standard method with which to monitor changes in functional capacity, a greater degree of confidence and precision in FCI values can be acquired through development of wetland reference data sets. Calibration of the playa and forested, low-gradient riverine wetland interim functional models that have reference wetland data would help to develop standards for assessing selected functions in these wetlands for continuing national assessment purposes. In addition, it is important to capture the stressors operating at multiple spatial scales within landscapes to assist in targeting functionally sustainable restoration sites. Identifying the stressors and the relative degree of degradation in a landscape will allow the development of appropriate surveys or monitoring efforts to gauge changes in wetland functional condition and proactively identify policy, program, and funding factors necessary to maintain sustainable wetland ecosystems.

The NRCS has shown a relative increase in wetland functional capacity index values on agricultural lands targeted for restoration within the temporary and seasonal prairie pothole wetland subclass of the Northern Prairie Pothole Region; the forested, low-gradient riverine wetland subclass of the Central and Lower Mississippi Valley; and the playa wetland subclass of the High Plains. The results also indicate that the current landscapes in which restorations are occurring do not reflect historic conditions and, as a result, a return to functional sustainability is thwarted. The results, however, provide a context in which NRCS can develop a refined, proactive approach to identify and quantify changes in wetland functional condition on private lands in the 21st century.

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Appendix A
Description of Study Area

Northern Prairie Pothole Region

The Northern Prairie Region (NPR) includes the Prairie Pothole Region (PPR), which extends from south-central Canada into the north-central United States and the Nebraska Sandhills (Lee et al. 1997). The U.S. portion of the PPR extends along the northern boundary of Montana, into northern and eastern North and South Dakota, western Minnesota, and south into the Des Moines lobe of north-central Iowa (Lee et al. 1997). The PPR can be further divided into the northern and southern PPR. The northern PPR (NPPR) is characterized by a cooler and less humid climate than the southern PPR and is dominated by small grain crops. The southern PPR has a higher amount of precipitation than the NPPR and is dominated by such row crops as soybeans and corn (Lee et al. 1997). Mixed- and tall-grass prairie are the potential natural vegetation of the PPR (USDA, NRCS 1981). The Nebraska Sandhills is a 52,000 square kilometers stabilized dune field dominated by irrigated agriculture and cattle ranching (Mitsch and Gosselink 1993). The study area is confined only to the NPPR (fig. 1).

The NPR is characterized by inter- and intra-annual fluctuations in seasonal mean temperature, humidity and precipitation. Periodic droughts are common because of variability in the regional climate. Precipitation measures than 11 inches of precipitation from April through September every 2 out of 10 years (Lee et al. 1997). The climatic variability of the NPR dominates hydrodynamics of the region, affecting all wetland functions.

Millions of prairie potholes dotted the glacial landscape of the PPR historically. Before the European settlement the area covered an estimated 80,000 square kilometers (Frayer et al. 1983). Conversion of the prairie landscape to agriculture eliminated many of these potholes, particularly those that dried in most years and could be tilled. An estimated 65 percent of the original wetland area in the PPR has been drained (Euliss and Mushet 1996). Much of the native prairie was also put to the plow, with only scattered remnants remaining by the beginning of the 20th century.

Several different hydrogeomorphic subclasses of prairie pothole wetlands exist in the NPPR, all of them within the Depressional class (Brinson 1993). Addressed in this study are temporary and seasonal wetlands (Stewart and Kantrud 1971). Temporary wetlands are dominated by wet-meadow vegetation. Seasonal wetlands have a center zone of shallow marsh surrounded by a zone of wet-meadow vegetation. However, fluctuating water levels as well as land use result in frequently alternating phases of the vegetation zones (Stewart and Kantrud 1971). The wetlands are hydrologically dominated by surface runoff from snowmelt and spring rains, with hydrologic losses due to evapotranspiration and ground water recharge (Lee et al. 1997). They are found on hummocky and undulating collapsed topography of the glaciated NPR, typically with nonintegrated drainage. Because of the closed basin topography and a significant source of wetland hydrology originating from the surrounding landscape, this wetland subclass is closely tied to the condition of the surrounding landscape (Lee et al. 1997).

Central and Lower Mississippi Valley

This portion of the study area is within two physiographic provinces, the Central Lowland Province and the Coastal Plain Province (Fenneman 1938). The Central Lowland Province includes the Dissected Till Plains Section. The Dissected Till Plains Section exists as the exposed Kansan glacial drift, a nearly flat till plain covered in loess that increases in depth close to the large rivers. The Kansan ice sheet is considered older than the Wisconsin, and therefore the landscape is more dissected than the adjacent Till Plains Section east of the Mississippi River.

The origin of the loess in the Dissected Till Plains Section is a result of glaciation. The loess was carried by water onto floodplains and then distributed over the till plain by wind. Bedrock immediately below the plain near the Mississippi River is comprised primarily of Mississippian age resistant limestones.

Paleozoic age sandstone and shale underlie the limestones, but are rarely exposed except in narrow areas near the Mississippi and Missouri Rivers (Fenneman 1938). The study area encompasses those portions of Missouri that drain into the Missouri and Mississippi River within the Dissected Till Plains Section; they will be referred to in this document as the Central Mississippi Valley (CMV). It is located above the confluence of the Ohio and Mississippi Rivers (fig. 1). Below the Mississippi-Ohio confluence lies the Mississippi Alluvial Plain Section within the Coastal Plain Province (Fenneman 1931). This area will be referred to as the Lower Mississippi Alluvial Valley (LMAV). The LMAV extends to the Gulf of Mexico, nearly 1,000 kilometers in length (Keeland et al. 1995) (fig. 1). The CMV and LMAV sample sites are respectively located in the Broadleaf Forest (Continental) Ecoregion Province, Central Till Plains, Oak-Hickory Section, and the Riverine Forest Ecoregion Province, Mississippi Alluvial Basin Section (Bailey 1997; McNab and Avers 1996).

Bottomland hardwood (BLH) wetlands have historically characterized the broad floodplains of these two physiographic regions. They represent the most dramatic wetland loss nationally (Keeland et al. 1995; Frayer 1991; Abernathy and Turner 1987; Hefner and Brown 1985). Historical acreage estimates put the extent of BLH wetland at approximately 10 million hectares (Roelle et al. 1990), with an estimated 8 million hectares accounted for in Arkansas, Louisiana, and Mississippi (Keeland et al. 1995). By 1937, approximately half of the BLH wetlands had been lost (Tiner 1984). An estimated less than 2 million hectares currently remain in the LMAV (The Nature Conservancy 1992).

Bottomland hardwoods are located along low-gradient waterways and, where flood waters are not excluded by levees, are often flooded on an annual basis. Where levees or drainage canals have been constructed, the frequency of flooding can be reduced considerably and even eliminated, drastically altering ecosystem and landscape functions. Bottomland hardwood wetlands are classified as forested, low-gradient riverine wetlands based on their geomorphic characteristics and hydrologic dynamics (Ainslie et al. 1999; Brinson 1993).

Bottomland hardwood wetlands found on the extensive floodplain of the mainstem Mississippi, as well as on the floodplains of the Missouri, Tensas, and other major tributaries in the Mississippi River basin, extend laterally across a topographic, edaphic, and hydrologic gradient. These areas are often referred to as zones for descriptive purposes and, in addition to characteristic vegetation, exhibit changes in redox potential, organic matter decomposition, and fish and wildlife use (Mitsch and Gosselink 1993; Wharton et al. 1982). However, the zones are tightly coupled because of the flow of energy in the form of water, sediment, nutrients, and organisms that maintain the bottomland hardwood wetlands. The zones characterized by temporarily and seasonally flooded wetlands (Zones IV and V, Wharton et al. 1982) occupy the somewhat higher topographic positions of these major floodplains.

Low-order tributaries of the Mississippi are also often dominated by temporarily- and seasonally-flooded bottomland hardwood wetlands. These are forested, low-gradient, temporarily, and seasonally flooded riverine wetlands. They are characterized typically by a mixed-deciduous hardwood canopy, often dominated by oak species (*Quercus* spp.) as well as maple (*Acer* spp.), ash (*Fraxinus* spp.), American elm (*Ulmus americana*), sweetgum (*Liquidambar styraciflua*), black gum (*Nyssa sylvatica*), and cottonwood (*Populus deltoides*), particularly in the northern portion of the LMAV (Mitsch and Gosselink 1993).

The forested, low-gradient riverine temporarily and seasonally flooded wetlands in the CMV and LMAV were selected for functional assessment. These wetlands are located in Major Land Resource Areas (MLRA) 131, the Southern Mississippi Valley Alluvium and 115, the Central Mississippi Valley Wooded Slopes (USDA, NRCS 1999). Extensive agricultural land in this region is planted to feed grains and hay for livestock in the CMV and soybeans, rice, cotton, wheat, and sugar cane constitute the primary crops in the LMAV. Combined, MLRA 131 and 115 are approximately 154,460 square kilometers in extent (USDA, SCS 1981).

The High Plains

The High Plains (HP) is a 5-million-year-old remnant landform of the Great Plains that resulted from regional uplift and subsequent erosion of the surrounding plains landscape (Trimble 1980). The landform extends from the southern edge of South Dakota south through two-thirds of Nebraska, clipping the southeastern corner of Wyoming, continuing through western Kansas, the eastern fringe of Colorado and New Mexico, and passing through the Oklahoma border into the Texas panhandle before ending at the Edwards Plateau (fig. 1). The High Plains is the central portion of the Great Plains and extends for more than 750 miles north to south (Trimble 1980).

The Southern High Plains (SHP), often referred to as the Llano Estacado, occupy an area approximately 82,000 square kilometers in extent (Bolen et al. 1989). The area is semiarid in the north and west and warm-temperate in the east and south (Bolen et al. 1989). Located south of the Canadian River in Texas and New Mexico, the SHP is characterized by intensive agricultural land use. Palacois (1981) estimated that 20,000 square kilometers of the SHP supported irrigated agriculture. The area lacks permanent surface waters but was historically peppered with playa wetlands, although no historical figure of their extent is available. The majority of the SHP lies within Major Land Resource Area 77, comprising approximately 127,000 square kilometers in size. Much of the SHP is in cultivation for wheat and grain sorghum, and has corn, soybeans, alfalfa and vegetable crops grown under irrigated conditions (USDA, SCS 1981).

Somewhat more numerous, but still fewer in number than in the Southern High Plains, playa wetlands extend through the Central High Plains generally north of the Canadian River to just north of the Arkansas River in Kansas and Colorado (Trimble 1980). A large proportion of the Northern High Plains is covered in sand dunes and loess, and the few playas that are in this region are confined to the eastern boundary of the High Plains north of the Arkansas River to the South Fork of the Republican River, Nebraska. A significant portion of Major Land Resource Area 72 comprises the Central and Northern High Plains, an area covering 77,220 square kilometers in size (USDA, NRCS 1981). This area is characterized by extensive fields of winter wheat, small grains, alfalfa, grain sorghum, and other hay crops. The High Plains aquifer provides irrigation water to grow sugar beets, corn, and grain sorghum (USDA, NRCS 1981).

Playa wetlands are hydrogeomorphically classified as Depressional (Brinson 1993). They are shallow, largely circular basins that have surface water present because of annual and seasonal precipitation events. High evapotranspiration, percolation, and regionally localized irrigation withdrawals account for hydrologic outputs (Haukos and Smith 1993). Annual species dominate the seed banks of playas, and successful species germinate and grow rapidly during relatively short and unpredictable precipitation events. This is likely the result of the highly variable fluctuating hydrologic regime present, that is, several submerged-drawdown cycles in a single growing season. In addition, vegetation structure can vary between playas because of the localized nature of the hydrologic inputs and losses (Haukos and Smith 1993). Basin surfaces are composed of clays (primarily the Randall soil) that form an almost impermeable layer between the basin floor and the underlying Ogallala Formation.

The Ogallala is the principal geologic unit of the High Plains and underlies approximately 80 percent of its extent (U.S. Geological Survey 1999). In recent years studies have identified the potential importance of playa wetlands to the recharge of the High Plains aquifer, particularly where they overlie the Ogallala Formation (Wood and Sanford 1994). Significant lowering of this regional ground water source for irrigated farming occurred after World War II (1945). By 1977, more than 70,000 wells tapped into the Ogallala (Bolen et al. 1989). The water table of the Ogallala decreased more than 15 meters between 1930 and 1980 (Weeks 1986, in Bolen et al. 1989). By 1990, an estimated 95 percent of water withdrawn from the High Plains aquifer was for irrigation (McGuire and Sharpe 1997).

Playa basins formed in loess, primarily from the Quaternary Blackwater Draw Formation (Holliday 1989). There are several mechanisms currently believed to be responsible for their formation, including deflation because of wind, fluvial erosion, and lacustrine deposition, salt dissolution, and resultant subsidence, dissolution of calcic soils and calcretes, and wildlife activities, particularly American bison (*Bison bison*) (Gustavson et al. 1994). While playa is Spanish for "beach," the Llano Estacado (meaning "staked plain") may be a corruption of llano estancado meaning "plain of many ponds" (Bolen et al. 1989). In 1541 the Spanish explorer Francisco Vasquez de Coronado first recorded the presence of "some ponds, round like plates . . ." (Bolen et al. 1989). Dendrochronological data indicate that his observations were made during an extremely severe drought that lasted for approximately 25 years (Weakley 1943).

Because of the lack of permanent surface water, playa basins provide the major source of surface water. Most playas and their watersheds are closed systems, lacking a surface water connection between them (Bolen et al. 1989). Playa wetlands are dominated by precipitation and runoff falling within the watershed. Evapotranspiration and seepage via cracks in the montmorillonitic clay into the Ogallala represent hydrologic losses from playa wetlands. Infiltration into adjacent permeable soil may also occur once the water table within the playa rises above the relatively impermeable clay soil lining the wetland surface (Wood and Osterkamp 1984).

The timing and duration of surface water varies, depending on the degree and type of alteration as well as on local weather patterns. Unaltered playas typically have surface water present from mid-spring through midwinter (Ward and Huddleston 1972). Irrigated playas, however, are flooded during the late winter and early spring (Guthery et al. 1982).

Appendix B
HGM Functional Models and Variables

Temporary and Seasonal Prairie Pothole Wetlands, NPPR
Forested, Low-Gradient Riverine Wetlands, CMV and LMAV
Playa Wetlands, the High Plains

Functional Capacity Index Models

Subclass	Function	Model
Temporary and seasonal Prairie Pothole wetlands, NPPR	Static surface water storage	$(V_{OUT} \times ((V_{SOURCE} + V_{UPUSE} + V_{SUBOUT})^{/3} + (V_{WETUSE} + V_{SED} + V_{PORE})^{/3})^{/2})^{1/2}$
	Dynamic surface water storage	If V_{PIT} is 1.0, then V_{PIT} is not used in the FCI calculation: $(V_{OUT} + (V_{SOURCE} + V_{UPUSE})^{/2} + (V_{PORE} + V_{WETUSE})^{/2})^{/3}$ If V_{OUT} is ≤ 0.5, or V_{PIT} is < 0.75, then FCI is 0.0. Otherwise, use: $[(V_{PIT} + V_{OUT})^{/2} + (V_{SOURCE} + V_{UPUSE})^{/2} + (V_{PORE} + V_{WETUSE})^{/2}]^{/3}$
	Nutrient cycling	$((V_{UPUSE} + V_{WETUSE} + V_{SED})^{/3} + (V_{PCOVER} + V_{DETRITUS})^{/2} + (V_{SOM} + V_{PORE})^{/2})^{/3}$
	Removal of imported elements and compounds	$[((V_{SOURCE} + V_{OUT} + V_{SUBOUT} + V_{PIT})^{/4}) \times ((V_{UPUSE} + V_{SED})^{/2} + (V_{PCOVER} + V_{WETUSE} + V_{DETRITUS})^{/3} + (V_{SOM} + V_{PORE})^{/2} + (V_{BDENSITY} + V_{VCONTINUITY} + V_{BWIDTH})^{/3})^{/4}]^{1/2}$
	Retention of particulates	If $V_{OUT} \le 0.5$, use: $[(V_{UPUSE} + V_{WETUSE} + V_{SED} + V_{OUT})^{/4} + (V_{BDENSITY} + V_{BCONTINUITY} + V_{BWIDTH})^{/3}]^{/2}$ If $V_{OUT} > 0.05$, use: $[(V_{UPUSE} + V_{SED})^{/2} + (V_{BDENSITY} + V_{BCONTINUITY} + V_{BWIDTH})^{/3}]^{/2}$
	Provide environment for characteristic plant community	$(V_{WETUSE} + V_{SED} + V_{OUT} + V_{PIT} + V_{SUBOUT} + V_{PRATIO} + V_{PCOVER} + V_{DETRITUS} + V_{SOM})^{/9}$
	Wetland habitat structure	$(V_{UPUSE} + V_{WETUSE} + V_{SED} + (V_{PRATIO} + V_{PCOVER})^{/2} + V_{DETRITUS} + V_{OUT} +$ $(V_{BWIDTH} + V_{BCONTINUITY} + V_{BCONDITION})^{/3})^{/7}$
	Habitat interspersion and connectivity	$[((V_{UPUSE} + V_{WETUSE} + V_{OUT})^{/3} \times ((V_{WDEN} + V_{WAREA} + V_{WPROXIMITY})^{/3})]^{1/2}$ Until reference standards are available for $V_{WPROXIMITY}$, use: $[((V_{UPUSE} + V_{WETUSE} + V_{OUT})^{/3} \times ((V_{WDEN} + V_{WAREA})^{/2})]^{1/2}$
Forested, low-gradient riverine wetlands, CMV, LMAV	Temporary storage of surface water	$[(V_{FREQ} \times V_{XSEC})^{1/2} \times (V_{ROUGH} + V_{SLOPE})^{/2}]$
	Retention and retarding the movement of ground water	$(V_{WTGRAD} \times V_{CONDUC})^{1/2}$
	Cycling of nutrients	$[V_{BTREE} + V_{SHRUB} + V_{HERB}^{/3}) + (V_{WD} + V_{DETRITUS})^{/2}]^{/2}$
	Removal and sequestration of elements and compounds	$[(V_{FREQ} \times (V_{SORPT} + V_{REDOX} + V_{DETRITUS} + V_{WD})^{/4}]^{1/2}$
	Retention of particulates	$[(V_{FREQ} \times V_{XSEC})^{1/2} \times (V_{ROUGH} + V_{SLOPE})^{/2}]^{1/2}$

Functional Capacity Index Models—Continued

Subclass	Function	Model
	Organic carbon export	$[(V_{LITTER} + V_{WD})^{/2} \text{ x } (V_{FREQ} \text{ x } V_{SURFCON})^{1/2}]^{1/2}$
	Provide environment for native plant community	$[(V_{COMP} + (V_{DTREE} + V_{BTREE})^{/2})^{/2} \text{ x } (V_{FREQ} + V_{POND} + V_{WTD} + V_{SOIL})^{/4}]^{1/2}$
	Provide wildlife habitat	$[(V_{FREQ} + V_{POND} + V_{MACRO})^{/3}) \text{ x } (V_{COMP} + V_{BTREE\,+} V_{DTREE} + V_{LOG} + V_{LITTER} + V_{SNAGS})^{/6}] \text{ x } (V_{SIZE} + V_{CONNECT} + V_{CORE})^{/3}]^{1/3}$
Playa wetlands, High Plains	Maintains characteristic hydrologic regime	$(V_{MOD} + V_{SED} + V_{SOADD} + V_{SORED} + V_{UPUSE} + V_{WETUSE})^{/6}$
	Maintains elemental cycling	$[((V_{BUFFCON} + V_{BUFFWID})^{/2}) + V_{MOD} + V_{PDEN} + V_{PORE} + V_{SED} + V_{WETUSE}]^{/6}$
	Retains particulates	$[((V_{SED} + V_{UPUSE})^{/2}) \text{ x } V_{MOD}]^{1/2}$
	Maintains characteristic plan community	$(V_{CANOPY} + V_{MICRO} + V_{MOD} + V_{PDEN} + V_{PRATIO} + V_{SED} + V_{WETUSE})^{/7}$
	Maintain habitat structure within wetland	$[((V_{BUFFCON} + V_{BUFFWID})^{/2} + V_{CANOPY} + V_{PDEN} + V_{PRATIO} + ((V_{SED} + V_{UPUSE} + V_{WETUSE})^{/3})]^{/5}$
	Maintain food web	$[((V_{BUFFCON} + V_{BUFFWID})^{/2} + V_{DETRITUS} + V_{LANDSP} + V_{PRATIO} + V_{SED} + V_{UPUSE} + V_{WETUSE}]^{/7}$
	Maintains habitat interspersion and connectivity among wetlands	$[[((V_{BUFFCON} + V_{BUFFWID})^{/2}) + V_{CANOPY} + V_{PDEN} + V_{PRATIO} + ((V_{SED} + V_{UPUSE} + V_{WETUSE})^{/3})]^{/5} + V_{LANDSP} + V_{WDEN}]^{/3}$

Subclass range	Variable	Description	Variable Index
Temporary and seasonal	$V_{BCONTINUITY}$	Grassland buffer continuity	0.0 – 1.0
Prairie Pothole	$V_{BDENSITY}$	Grassland buffer density	0.0 – 1.0
wetlands, NPPR	V_{BWIDTH}	Grassland buffer width	0.0 – 1.0
	$V_{DETRITUS}$	Detritus	0.0 – 1.0
	V_{OUT}	Wetland outlet	0.0 – 1.0
	V_{PCOVER}	Plant cover	0.0 – 1.0
	V_{PIT}	Excavation	0.0 – 1.0
	V_{PORE}	Soil porosity	0.0 – 1.0
	V_{PRATIO}	Ratio of native to nonnative plant species	0.0 – 1.0
	$V_{PROXIMITY}$	Proximity to other wetlands	0.0 – 1.0
	V_{SED}	Sediment delivery to wetland	0.0 – 1.0
	V_{SOM}	Soil organic matter	0.0 – 1.0
	V_{SOURCE}	Source area of flow interception	0.0 – 1.0
	V_{SUBOUT}	Constructed subsurface/surface outlet	0.0 – 1.0
	V_{UPUSE}	Upland land use	0.0 – 1.0
	V_{WDEN}	Density of wetlands in the landscape	0.0 – 1.0
	$V_{WETAREA}$	Wetland diversity in the landscape	0.0 – 1.0
	V_{WETUSE}	Wetland land use	0.0 – 1.0
Forested, low-gradient	V_{BTREE}	Basal area of trees	0.0 – 1.0
riverine wetlands,	V_{COMP}	Plant species composition	0.0 – 1.0
CMV and LMAV	V_{CONDUC}	Saturated hydraulic conductivity	0.0 – 1.0
	$V_{CONNECT}$	Connectivity to adjacenthabitats	0.0 – 1.0
	V_{CORE}	Interior core area	0.0 – 1.0
	V_{CWD}	Coarse woody debris	0.0 – 1.0
	$V_{DETRITUS}$	Primary detrital component	0.0 – 1.0
	V_{DTREE}	Tree density	0.0 – 1.0
	V_{FREQ}	Frequency of overbank flow	0.0 – 1.0
	V_{HERB}	Herbaceous cover	0.0 – 1.0
	V_{LITTER}	Surfaces for microbial activity	0.0 – 1.0
	V_{LOG}	Logs	0.0 – 1.0
	V_{MACRO}	Macrotopographic relief	0.1 – 1.0
	V_{POND}	Extent of ponding	0.0 – 1.0
	V_{REDOX}	Presence of redox soil features	0.0 (Absent) 1.0 (Present)
	V_{ROUGH}	Floodplain roughness (Mannings coefficient)	0.1 – 1.0
	V_{SHRUB}	Density of shrubs	0.0 – 1.0
	V_{SIZE}	Size of the wetland of which the wetland assessment area is part	0.0 – 1.0

Description of Model Variables*

Subclass Index range	Variable	Description	Variable
	V_{SLOPE}	Flood plain slope	0.1 – 1.0
	V_{SNAGS}	Density of standing dead trees	0.0 – 1.0
	V_{SOIL}	Presence of characteristic soil	0.0 – 1.0
	V_{SORPT}	Sorptive properties of soils	0.0 – 1.0
	$V_{SURFCON}$	Surface hydraulic connections	0.0 – 1.0
	V_{WD}	Woody debris	0.0 – 1.0
	V_{WTD}	Depth of water table	0.5 – 1.0
	V_{WTGRAD}	Water table gradient	0.0 – 1.0
	V_{XSEC}	Floodplain:channel width ratio	0.1 – 1.0
Playa wetlands,	$V_{BUFFCON}$	Buffer zone continuity	0.0 – 1.0
High Plains	$V_{BUFFWID}$	Buffer zone width	0.0 – 1.0
	V_{CANOPY}	Plant community canopy	0.0 – 1.0
	$V_{DETRITUS}$	Detritus	0.0 – 1.0
	V_{LANDSP}	Landscape condition	0.0 – 1.0
	V_{MICRO}	Wetland microtopography	0.25 – 1.0
	V_{MOD}	Excavation or qther modification to wetland basin	0.1 – 1.0
	V_{PDEN}	Wetland plant density	0.25 – 1.0
	V_{PORE}	Soil quality within 50 cm of wetland soil surface	0.0 – 1.0
	V_{PRATIO}	Ratio of native to non-native plants	0.0 – 1.0
	V_{SED}	Sediment pelivered to petland	0.0 – 1.0
	V_{SORED}	Source prea flow interception	0.0 – 1.0
	V_{SOADD}	Source area flow addition	0.0 – 1.0
	V_{UPUSE}	Upland land use	0.0 – 1.0
	V_{WDEN}	Wetland density	0.0 – 1.0
	V_{WETUSE}	Wetland land use	0.0 – 1.0

* For more information on the three subclass models applied, contact:

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CMV – Chris Hamilton, NRCS, Missouri State Office, 573/876-9416, email chris.hamilton@mo.usda.gov;
LMAV – Sam Davis or Delaney Johnson, Mississippi State Office, 601/634-7996
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Playa Wetlands, High Plains:
Robert Schiffner, NRCS, Kansas State Office – Dodge City Area Office, 316/227-3431,
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Appendix C
Restoration Site FCI Values at T0 and T1

Figures 1 through 28 of appendix C show the Funtional Capacity Index (FCI) values for each restoration site by subclass and function. The **y** axis is labeled FCI and is the calculated Functional Capacity Index for T0 and T1. The **x** axis is the site label. The key for the site label is as follows:

Region ID:

NPPR	Northern Prairie Pothole Region
LMAV	Lower Mississippi Alluvial Valley
CMV	Central Mississippi Valley
HP	High Plains

State abbreviation: (used only for LMV and HP)

AR	Arkansas
KS	Kansas
LA	Louisiana
MO	Missouri
MS	Mississippi

Sequential site ID number:
First two digits, beginning with 01

Site age:
Last two digits in site label