

Floodplain Forests and Water Quality in the Upper Mississippi River System

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Photo Courtesy of Lewis and Clark Community College

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Executive Summary

Management and protection of water resources has been an important mission of federal, state, and local governments since the passing of the Clean Water act in 1972. The Nation's focus on water resources has in turn resulted in a large number of scientific studies. Many have focused on conservation agriculture and related practices, such as riparian forested buffer strips, conservation tillage, filter strips, wetland construction, and other farming techniques designed to prevent sediments and nutrients from entering into surface water. However, the links between floodplain forests and forest management to water quality are less well known, in part due to a lack of resources for these types of studies.

The Mississippi River is the largest riverine ecosystem in North American and the third largest in the world. The Upper Mississippi River System (UMRS) floodplain ecosystem is a mixture of bottomland forest, grasslands, wetlands, island, backwaters, and side channels. These ecosystems support more than 300 species of birds, 57 species of mammals, 45 species of amphibians and reptiles, 150 species of fish, and nearly 50 species of mussels. Floodplain forests are an essential component of the UMRS and support a suite of ecosystem services such as reducing eutrophication, sediment abatement, carbon (C) sequestration, wildlife habitat, and flood water storage. However, forest cover along this large river-floodplain ecosystem has significantly declined from historical levels, and many of today's floodplain forests suffer from chronic ecological stressors that have led to further declines in health and diversity, including reductions in regeneration capacity. The lack of natural regeneration in mature forests leads to large, open canopy gaps that provide a window of opportunity for invasive species like reed canary grass (*Phalaris arundinacea*). Once established, reed canary grass (RCG) outcompetes native plants, and its reproduction tactics, dense root mass, and adaptability make it a dangerous nuisance to native ecosystems including floodplain forests in the UMRS. The long term impacts of vegetation dynamics such as these on large river ecosystem functions, habitat integrity, and water quality are often not well understood.

Studies have found that altered hydrology has led to further declines in forest health and diversity in many areas, and induced shifts in forest composition towards more flood tolerant tree species. Restoration efforts have been met with limited success and low survival rates of young

tree plantings. This is often due to a combination of changes in hydrology and inundation frequency, soil complexes, and competition from invasive species like RCG. Over time, cumulative effects include a gradual thinning of the forest canopy, loss of bottomland forest, reduced biodiversity, and an increase in the size and extent of RCG meadows throughout the upper reaches.

The full impact of forest loss is not yet fully understood, but research shows that forests have a large impact on the soils and waters within their system. Floodplain forests are not only a result of a wet habitat, but they are drivers of the terrestrial water cycle. Using their deep-reaching roots, they maintain evapotranspiration rates that recharge precipitation even in drought conditions. Plants, including trees, and their associated microbial communities help reduce soil erosion and promote nutrient cycling. The presence of trees in the landscape also has a beneficial impact on soil hydrology. Evidence from several studies suggests that floodplain forests have the potential to more effectively mitigate flood risk compared to grasslands and agriculture fields. Just as vegetation affects watershed and floodplain hydrology, it also affects the erosion and deposition of sediments transported by water through the watershed. Deforestation clearly results in soil erosion and a corresponding increase in sediment loads to rivers and streams. Conversely, when grassland pastures are converted to a mix of broadleaf trees, long term infiltrations rates are significantly increased and surface runoff is reduced.

Nutrient loading and sediment deposition from non-point sources are the primary drivers of poor water quality in rivers and streams. Floodplain forests have the capacity to remove nitrates from river systems through the process of denitrification and have been shown to act as important nitrate sinks in river systems and riparian zones. In floodplains, denitrification rates can be at least four times higher than in river channels. Floodplain forests also have the capacity to absorb chemicals like organic pesticides that run off agricultural fields into our water systems. It should be noted that different plant species in riparian zones and vegetated floodplains have unique physiological processes, and thus vary in their capabilities to take up and degrade herbicides and pesticides. These results emphasize the importance of species diversity among floodplain plant communities in the removal of anthropogenic pollutants.

High biodiversity in an ecosystem like a forest helps to stabilize the system and allow it to function under varying conditions. Restoration efforts can be highly successful in reestablishing system function and services, but goals should include preventing ecosystem degradation, biodiversity loss, and invasion by non-native species. Although restored forests and wetlands may take years or decades to stabilize, with sufficient effort and monitoring, functions and services can be renewed to approximately 90% of undisturbed systems in as little as 10 years.

Major Findings

- Floodplain forest cover in the UMRS is dramatically reduced from historical levels due to basin-wide changes in land use and development. Altered hydrology and additional ecological stressors have led to further declines in forest health and diversity in many areas, and induced shifts in forest composition towards more flood tolerant tree species.
- Restoration efforts in UMRS floodplains are often met with limited success and low survival rates among tree plantings. This is often attributable to: discrete flood events or other disturbances; an incomplete understanding of the relationships between ecological tolerances of tree species and elevation, hydrology, inundation frequency, and alluvial soil complexes; competition from invasive species; and in many cases a combination of these factors.
- Reed canary grass (RCG) is one of the most damaging invasive plant species in the UMRS, especially in the northerly river reaches. An increase in the size and extent of RCG meadows in floodplain areas throughout the upper reaches has contributed to a gradual thinning of the forest canopy, loss of bottomland forest, and reduced biodiversity.
- The potential impacts of climate change on UMRS floodplain forests are not yet well known, but changes in temperature and precipitation patterns, longer growing seasons, higher atmospheric CO₂ levels, and increased flood and disturbance frequencies all have the potential to lead to increased ecological stresses and impact biodiversity in Mississippi River floodplain ecosystems.
- High biodiversity helps to stabilize an ecosystem and maintain its functions and services. Through time, species have complementary effects on each other to support different ecosystem functions. High biodiversity of plants and their associated microbial communities is positively associated with reducing soil erosion and promoting nutrient cycling.

- Floodplain forests not only occupy wet habitats, but they can be drivers of the terrestrial water cycle. They maintain evapotranspiration rates to recharge precipitation even in drought conditions using their deep-reaching roots. For example, in the Mississippi River Basin over 50% of precipitation may be generated by evapotranspiration.
- Restoration efforts can be highly successful in reestablishing system functions and services, but goals should also include preventing ecosystem degradation and invasion by non-native species.
- Although restored forests and wetlands may take years or decades to stabilize, with sufficient effort and monitoring, functions and services can be renewed to approximately 90% of undisturbed systems in as little as 10 years.
- Elevated infiltration and evapotranspiration rates in forested riparian areas compared to grasslands reduce the overall volume of runoff water reaching streams and thus attenuate the intensity of runoff events.
- Past research has found a significant difference between how nutrients are removed by grass vs. forested buffer strips. Forested buffer strips have been found to be more efficient at filtering out nitrates compared to crops or grasses, and lowland forested buffer strips have higher nitrate uptake rates for groundwater and surface runoff in headwater systems.

Research Recommendations

- Development of a system-wide, georeferenced, data-driven model clarifying the functional relationships, tolerance ranges, and interactions between vegetation communities and hydrological regimes, micro-elevation, and soil properties in the UMRS.
- Development of a system-wide GIS-driven effort to map RCG dominated areas in the UMRS and identify areas that are likely to transition from floodplain forest cover to RCG based on underlying forest community characteristics and dynamics, elevation, and/or soil properties.
- Additional research into the role of UMRS floodplain forests and vegetation communities on nutrient and carbon sequestration and fluxes, including the impact of widespread invasive species like RCG on river-floodplain nutrient dynamics, is highly recommended. This would provide much-needed information to comprehensive assessments of large river nutrient loads and dynamics at basin-level scales relevant to issues such as Gulf Coast hypoxia.

- Development of additional site-level experimental research projects in the UMRS focused on topics related to: improving guidelines for floodplain forest restoration methods and techniques; interactions between floodplain forests and invasive species; the effects of floodplain vegetation dynamics on biogeophysical processes; and links between vegetation composition and structure, nutrient and carbon fluxes and sequestration, and water quality at multiple scales.
- Few studies have investigated the capabilities of woody perennials to remove chemical herbicides and pesticides. While the uptake and removal of these chemicals by grasses and other annual crops has been fairly well studied, additional investigation of the ability of floodplain forest tree species in the UMRS to improve water quality through phytoremediation is needed.

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Introduction

Management and protection of water resources has been an important mission of federal, state, and local governments since the passing of the Clean Water act in 1972. The US Environmental Protection Agency (USEPA) has led efforts to protect water quality, the US Department of Agriculture (USDA) has been instrumental in addressing nonpoint source pollution from the agricultural sector, and the US Army Corps of Engineers (USACE) has led efforts to control flooding and maintain navigation infrastructure. More recently, the USACE has also focused on balancing flood control and navigation needs with maintaining and restoring ecosystem services in important waterways like the Upper Mississippi River System (UMRS).

The Mississippi River is the largest riverine ecosystem in North America and third largest in the world. The UMRS, which includes the combined 500-year floodplains of the Upper Mississippi, Illinois, and navigable portions of the Kaskaskia, Minnesota, Black and St. Croix Rivers, covers approximately 2.6 million acres. The UMRS floodplain ecosystem contains a mix of bottomland forests, grasslands, islands, backwaters, side channels and wetlands – all of which support more than 300 species of birds, 57 species of mammals, 45 species of amphibians and reptiles, 150 species of fish, and nearly 50 species of mussels. It is a migratory flyway for more than 40 percent of North America’s migratory waterfowl and shorebirds, and a globally important flyway for 60 percent of all bird species in North America (USACE 2004).

The nation’s focus on water resources has resulted in a large number of scientific studies conducted by universities, federal research agencies, private foundations, and other groups interested in protecting and managing our water resources. The focus of many of these studies has been to evaluate the impacts of agricultural practices on water resources, especially water quality. Consequently, most scientific studies have evaluated various conservation agriculture and related techniques, such as riparian forested buffer strips, conservation tillage, filter strips, wetland construction, and other farming techniques designed to prevent sediments and nutrients from eroding into surface water. Consequently, scientific data linking floodplain forests and forest management actions to water quality and flood attenuation is more limited because it has generally not received the same level of federal funding as other methods for protecting and managing water resources.

Floodplain forests are an essential component of the UMRS, and support a suite of ecosystem services such as reducing eutrophication, sediment abatement, carbon sequestration, wildlife habitat, and flood water storage. However, floodplain forest cover along this large river-floodplain ecosystem has significantly declined from historical levels, and many of today's floodplain forests suffer from chronic ecological stressors that have led to further declines in health and diversity, including reductions in regeneration capacity. For example, levee systems prevent flooding and protect developed and agricultural land, but levees also disconnect protected portions of the floodplain and the remnant ecological communities that occupy them from the natural river fluctuations they are adapted to. The lock and dam system is essential for transportation along the Upper Mississippi River, but it has also dramatically altered hydrology within the river system. Locks and dams elevate the water table in impounded reaches and make soils in some lower-lying areas that once promoted floodplain forests too waterlogged for natural tree regeneration to replace canopy trees that senesce and die.

Lack of natural regeneration in mature floodplain forests can in turn lead to large, open canopy gaps that provide a window of opportunity for invasive species that require high light environments to become established, such as reed canary grass (*Phalaris arundinacea*). Reed canary grass (RCG) is a perennial cool season grass that historically was grown as a forage crop (Read and Ashford 1968, Kroth et al. 1976) or a conservation grass (Rice and Pinkerton 1993). RCG can reproduce vegetatively through rhizomes leading to fast and total colonization of open areas. Once established, RCG outcompetes native plants, and its reproduction tactics, dense root mass, and adaptability make it a dangerous nuisance to native ecosystems including floodplain forests in the UMRS. The long term impacts of vegetation dynamics such as these on large river ecosystem functions, habitat integrity, and water quality are often not well understood.

Based on an exhaustive literature review, the following report provides an overview of floodplain forest habitats, trends and associated ecosystem services in the UMRS, and discusses the relationship between floodplain forests and water quality by focusing on two key areas: 1) interactions between floodplain forests and nutrient and sediment fluxes in rivers and streams, and 2) the effects of forest restoration and management activities on floodplain ecosystems, hydrology, and flood attenuation.

Floodplain Forests and Vegetation Dynamics

Background and Historical Trends

Land cover in the Upper Mississippi River (UMR) basin is primarily agriculture, and the majority of forestland occurs in the northern (Minnesota and Wisconsin) and southern (southwestern Illinois and southeastern Missouri) parts of the basin. However, a considerable amount of forestland in the central portions of the basin is associated with river and stream corridors, including floodplains and tributaries of the UMRS.

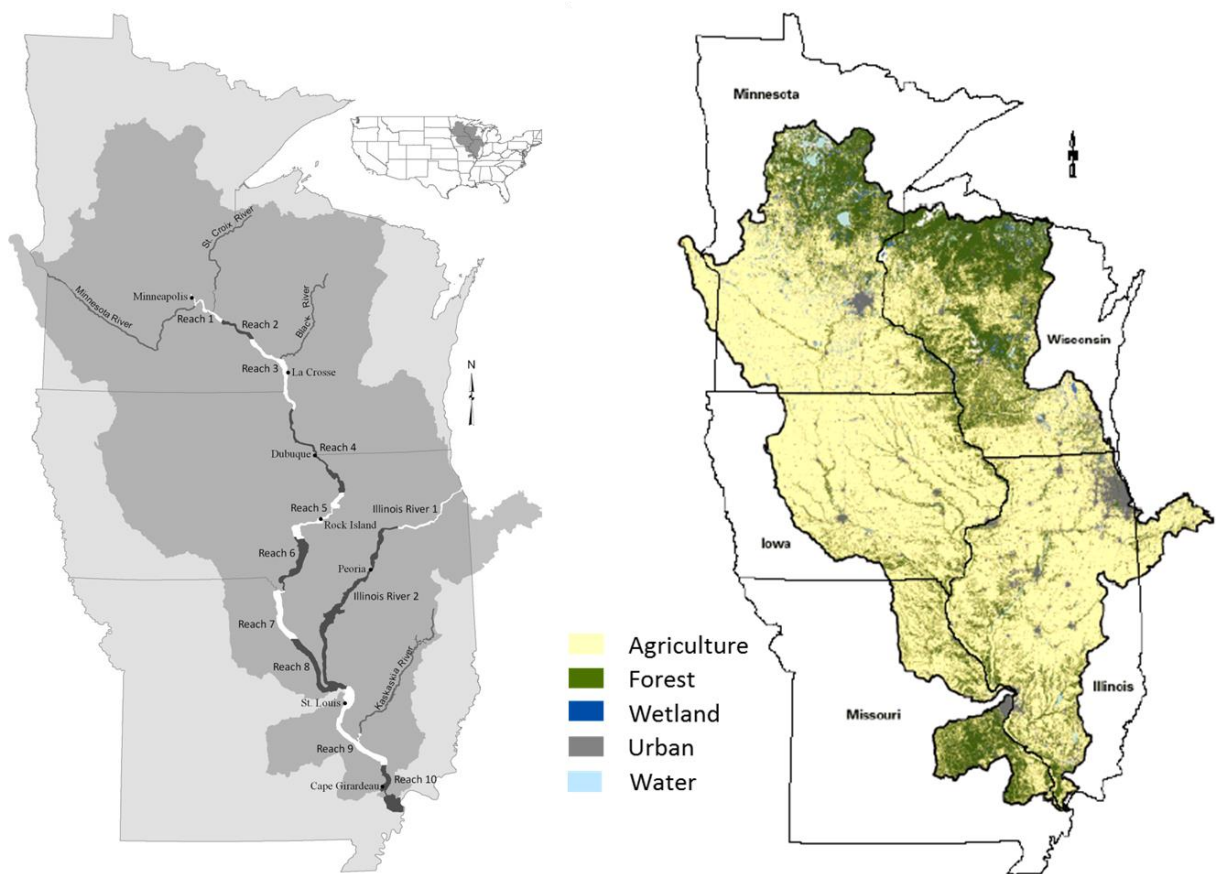


Figure 1. The UMRS and UMR Basin land cover. *From: (Guyon et al. 2012).*

The UMRS floodplain ecosystem is complex, spatially and temporally dynamic, and interspersed with a variety of habitat types differentiated by an interacting combination of environmental factors and gradients such as hydrology, soils, geomorphology, elevation, biological succession, and disturbance (Guyon et al. 2012). Forests, grasslands, wet meadows, and shrublands often combine to form an interconnected mosaic of terrestrial and aquatic habitats within larger floodplain ecosystems and even smaller project-scale management units.

The historic UMRS ecosystem exhibited natural gradients in habitat among river reaches. Northern river reaches were more forested and were composed of mixed silver maple forests, river channels, seasonally flooded backwaters, floodplain lakes, marsh, and prairie. Beginning around the northern Iowa border and along the lower Illinois River, floodplain plant communities were dominated by riparian forests that graded through oak savanna to prairie at higher elevations. Below the Kaskaskia River, the floodplain was heavily forested with species characteristic of southern bottomland hardwood communities including baldcypress (*Taxodium distichum*), Nuttall oak (*Quercus texana*) and cherrybark oak (*Quercus pagoda*) (Theiling et al. 2000, USACE 2004).

Prior to widespread European settlement of the Upper Midwest region, the UMRS ecosystem supported a diverse landscape of tallgrass prairie, wetlands, savannas, and forests (USGS 1999). However, European settlement brought many changes to the landscape and associated river channels. The rivers provided efficient transportation and were the focal point of commerce and colonization. As the economy and population grew in the Midwest, so did the demand to use rivers for transportation. The U.S. Government became involved in Mississippi River navigation in 1824 when the Army Corps of Engineers was tasked with removing logs and other obstructions from the river channels to ease constraints on steamboat travel. Some of the more important actions and impacts following that were: the 4 ½ ft and 6 ft Channel Projects in 1878 and 1907, respectively; the 9 ft Channel Navigation Project (authorized in 1930); impoundment and river regulation; increased commercial navigation traffic; continued resource exploitation (e.g., logging, hunting, commercial fishing and clamming); increased industrialization of agriculture; urban development; expansion of levee systems; water quality degradation; and more recently exotic species introductions (USACE 2004).

Over the past 150 years, these impacts have resulted in a contemporary landscape that is more than 80 percent developed (USGS 1999). Wetland drainage, agricultural field tiles, road ditches, channelized streams, and urban storm water runoff accelerate flow to main stem rivers (Demissie and Kahn 1993). Modern hydrology is highly altered, with increased frequency and amplitude of changes in river discharge in some river reaches, and dams and reservoirs in the basin and river regulation in the main stem also modify river flows (USGS 1999). The contemporary landscape also delivers large amounts of sediment, nutrients, and contaminants to the river (Bhowmik and Demissie 1989, Meade 1995, USGS 1999, WestConsultants 2000, WestConsultants 2000, USFWS 2006).

Although river impoundment flooded considerable forested area in northern reaches, large portions of forest remain relatively intact in National Wildlife Refuge areas. In other river reaches, most natural floodplain communities have been replaced by agriculture. Channel dynamics and water level fluctuations that support diverse, productive floodplain communities have been altered throughout the UMRS.

Floodplain forests are important to the biological integrity of the UMRS. As stated in the report entitled *Ecological Status and Trends of the Upper Mississippi River System 1998* (USGS 1999), "The ecosystem as a whole benefits from floodplain forests. Besides serving as a rich habitat for wildlife and fish during floods; the forests reduce soil erosion, improve water quality and provide a scenic and recreational landscape" (Urich et al. 2002).

Historical and present day disturbances have contributed to long-term changes in the composition and structure of UMRS floodplain forests (USGS 1999). Historic surveys reveal a higher proportion of oaks and other mast trees in the forest community than exist today (USGS 1999). Impacts of river floodplain development include forest loss and water gain in northern reaches, and forest and grassland losses in the rest of the UMRS (USACE 2004). Modern UMRS forests represent only a small portion of pre-European settlement floodplain forests in some reaches. Floodplain forest coverage has been significantly reduced from pre-settlement levels due to timber harvesting, conversion to agriculture, and other land use changes (Yin et al. 1997,

Kruse and Groninger 2003), and contemporary floodplain forests are affected by altered hydrological regimes associated with river impoundment.

The dramatic loss of forested and prairie land cover throughout the majority of reaches is immediately discernable. For example, forests covered 56 percent of the landscape at the confluence of the Illinois and Mississippi Rivers in 1817. By 1975, these forests were reduced to 35 percent of the landscape (Nelson et al. 1995). Similarly, floodplain forests covered 71.4 percent of the landscape in a 63- mile-long portion of the unimpounded reach in 1809, but by 1989 covered only 18.3 percent of the same landscape (Yin and Nelson 1995). A corresponding loss of tree species diversity has been identified as a management concern in the UMRS (Allen 1997, Guyon et al. 2012), and a general lack of successful tree regeneration, particularly the hard mast component, has also been documented in many areas (Battaglia et al. 2002, Battaglia et al. 2008). Further losses in floodplain forest habitat and diversity over time are likely unless active forest management can reverse this trend (Urich et al. 2002, Guyon et al. 2012).

In addition to landscape-level changes in land cover/land use, alterations in hydrological regimes and the isolation of large portions of the floodplain behind mainline levees have resulted in significant compositional shifts in floodplain forest communities. Many mast-producing species such as oaks and hickories have declined in importance, while silver maple has dramatically increased in importance throughout the UMRS (Figure 2). Species importance values combine measures of relative density, relative frequency, and/or relative dominance into a single metric and indicate the overall abundance of a species in an ecological community. Historically, forested floodplain habitats have been significantly affected over time by human use and modifications to the Mississippi and Illinois Rivers and their floodplains. There was a significant reduction in floodplain forest due to timber harvesting, conversion to agriculture and creation of drainage districts. In addition, there was a significant reduction in floodplain forest acreage during lock and dam construction and river impoundment. While the remaining forests and grasslands may appear to casual observers to be natural and pristine, many of the important processes that determine their growth and survival are now artificially managed and are not representative of pre-settlement conditions. Furthermore, excessive sedimentation and nutrient inputs from upland sources continue to degrade water quality and floodplain habitats.

Environmental regulations and conservation incentive programs in place since the 1970s have resulted in significant improvements in water quality and upland habitat. Most surface waters are now in compliance with established standards, and many sensitive species such as freshwater mussels and mayflies have shown a recurrence even in previously degraded zones below cities. However, non-point source pollution, such as nutrient runoff from agricultural fields, continues to be a problem throughout the system.

Effective habitat protection, management, and restoration have been critical elements in maintaining river-floodplain habitats in the UMRS. Improved land use has substantially reduced erosion and sedimentation in many streams. Setting aside marginal lands has been beneficial for wildlife, and management activities have demonstrated the effectiveness of restoration methods in establishing natural habitats. These types of experiences clearly demonstrate the resilience of rivers like the Mississippi and Illinois and the positive return on investments in environmental restoration (USACE 2004). However, there are still a great many problems and a demonstrable need to continue to advance restoration efforts.

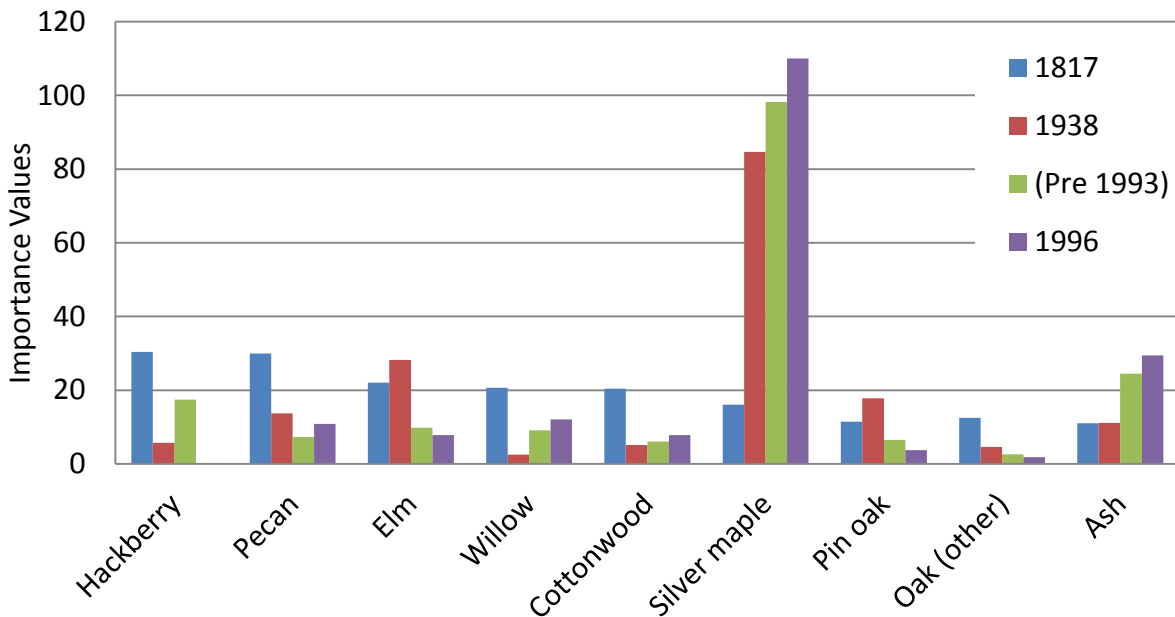


Figure 2. Tree species importance values at the confluence of the Illinois and Mississippi Rivers. Importance values are equal to the sum of relative density and relative dominance on a scale of 0-200. Adapted from: (Nelson and Sparks 1998)

Contemporary Floodplain Forests

Today, contiguous forest cover in the UMRS is primarily confined to a relatively narrow band on the riverward side of agricultural levees, particularly in the lower reaches (USACE 2004). Above Pool 13, which runs from Clinton to Bellevue on the Iowa side of the river, the floodplain is generally narrower in extent and the levee system is not as pronounced. Natural channel dynamics and water level fluctuations have also been altered throughout the UMRS, further reducing the natural diversity and productivity of floodplain ecosystems (Theiling et al. 2000). Species composition of the remaining forest has also become less diverse, due in part to the altered hydrology resulting from river impoundment, a corresponding loss of the seasonal “flood pulse” (Junk et al. 1989), and the effects of periodic severe flooding, particularly the flood of 1993. This reduction in species diversity is especially evident in the decline of mast-producing species such as oaks and hickories (USGS 1999). Bank erosion has also affected floodplain forests to some degree (USACE 2004), and diseases, insects and invasive plant species also continue to negatively impact UMRS floodplain forests throughout the system.

Land cover/land use in the UMRS is generally defined by the General Wetland Vegetation Classification System (GWVCS), a 31-class system developed and used by the USACE Upper Mississippi River Restoration Environmental Management Program. It was developed from year 2000 color infrared aerial photography and was designed primarily for use in systemic level studies. A full description of the development of the GWVCS and all 31 land use/land cover types it encompasses can be found in the General Classification Handbook for Floodplain Vegetation in Large River Systems (Dieck and Robinson 2004). The following are brief descriptions of a few of the terrestrial UMRS vegetation types most relevant to this report:

Floodplain Forest (FF) – Floodplain Forest represents areas on islands, near the shoreline, or around lakes, ponds, and backwaters that are more than 10 percent vegetated with seasonally flooded forests. These forests are predominantly silver maple (*Acer saccharinum*), but also include elm (*Ulmus americana*), cottonwood (*Populus deltoides*), black willow (*Salix nigra*), and river birch (*Betula nigra*). This general class is typically found growing at or near the water table where it becomes inundated from spring flooding and high water events.

Lowland Forest (LF) – Lowland Forest represents areas along the riverbanks and within the floodplain that are drier than floodplain forest sites and are more than 10 percent vegetated with temporarily flooded forests. Common trees include pecan (*Carya illinoensis*), hickory (*Carya spp.*), river birch, sycamore (*Platanus occidentalis*), and oak (*Quercus spp.*). This general class is most common in southern reaches of the Upper Mississippi and Illinois River Systems and is typically found growing on moist, well-drained soils.

Wet Meadow (WM) – Wet Meadow represents lowland areas that are more than 10 percent vegetated with perennial grasses and forbs. Common vegetation includes reed canary grass (*Phalaris arundinacea*), rice cut-grass (*Leersia oryzoides*), and goldenrod (*Solidago spp.*). This general class may have small inclusions of woody vegetation, sedges, or emergent vegetation, such as smartweed (*Polygonum spp.*) or purple loosestrife (*Lythrum salicaria*). It is typically found growing on saturated soils and is often considered the transition zone between aquatic communities and uplands.

A healthy, functioning floodplain forest requires a diversity of forest structural components including tree species, age and size classes, canopy heights, and understory composition (Urich et al. 2002). However, changes in flood frequency, duration, and depth resulting from river impoundment and channelization have reduced diversity within remaining UMR forests in all four major river reaches (Yin and Nelson 1995). Much of the current floodplain forest is between 50 and 70 years old, consisting of mostly three or four flood and shade tolerant species, but heavily dominated by silver maple. With sustained high water levels, little germination takes place and seedlings are unable to survive frequent floods. The closed canopy of these even-aged forests also prevents the re-establishment of other species that are shade intolerant such as cottonwood, black willow, and river birch. Hard mast species, such as oaks, have significantly declined and now occur on less than 10% of the floodplain (Urich et al. 2002).

It is expected that significant canopy die-off will occur in many locations throughout the UMRS within about 50-70 years due to the mature, even-aged condition of much of the forest resource (USGS 1999). This will likely result in open conditions and promote undesirable species such as RCG that make it difficult for floodplain forest trees to regenerate. Large scale die-off from floods or other disturbances could also result in a similar conversion of vegetation cover. In

addition to the wildlife habitat it provides, closed canopy forest limits the establishment and expansion of the invasive RCG through shading.

Upper Mississippi River community-level vegetation types and floodplain forest tree species are distributed along ecological gradients defined mostly by their ability to survive various levels of inundation (Urich et al. 2002, Romano 2010, De Jager et al. 2016). Lower lying areas typically support the most flood-tolerant species, including willows, cottonwood, silver maple, and green ash (*Fraxinus pennsylvanica*). Trees located on higher elevations along ridges or terraces have less tolerance to flooding and high water tables. These include species like oaks and hickories that occupy formerly high points of land in the floodplain but are no longer able to reproduce successfully because of inundation and/or permanently elevated water tables. Some floodplain studies that have shown the importance of slight changes in elevation that correspond to hydrological differences (Allen et al. 2001, Battaglia and Sharitz 2006, De Jager et al. 2012), and report strong turnover of species occurs with only slight changes in elevation (Battaglia et al. 2002). Several studies have looked at how the physiological tolerance limits of trees effects their distribution across the floodplain landscape but more work remains to refine those relationships, due in large part to the differences between present day hydrological regimes and historical baselines (Romano 2010, De Jager et al. 2012).

Floodplain Forest Restoration

As already noted, a general decline in tree species diversity, particularly the hard mast component, is a management concern in UMRS floodplain forests (Guyon et al. 2012), and a general lack of desirable tree regeneration has also been documented (Battaglia et al. 2002). Many floodplain forests tend to be dominated by large mature trees and this could be a response to the periodic flood disturbances that preferentially lead to higher mortality in younger, smaller-sized cohorts (USGS 1999, Cosgriff et al. 2007). Efforts to restore UMRS floodplain forests are ongoing, but have been met with limited success in many locations. In some cases, this is also attributable to the frequent flood disturbances associated with large river floodplain habitats. At any rate, tree plantings in floodplains have often met with mixed results (Stanturf et al. 2001, Dey et al. 2003), and refining methods to assess site quality and suitability for plantings would benefit restoration efforts in the UMRS.

A major factor to overcome during restoration efforts is altered hydrology due to impoundment, levees, upland development, and climate change which is impacting forest distribution and structure system-wide (Yin et al. 1997, Knutson and Klaas 1998, Yin 1998, Ryan et al. 2008, Prasad et al. 2009, Romano 2010, Guyon et al. 2012) . Upper Mississippi River forests were affected by changes in surface and groundwater levels when the river was transformed from a free-flowing, floodplain river to a shallow draft navigation system. The navigation system includes 37 lock and dams of variable size and length depending on their location in the system. Dams have variable degrees of impact throughout the system. They inundate a large proportion of the floodplain in the northern river reaches and less in the south (Theiling and Nestler 2010). Regardless of their location in the system, however, they increase water levels upstream from each dam, alter surface and groundwater stage-discharge relationships, and also reduce water level variation (Theiling and Burant 2013). Many UMR forests have transitioned to a low diversity riverfront forest community found in the 0-2 year flood frequency zone since the navigation system became operable. Hydrology is an important driver, but there may be other factors like soil composition that also influence the structure, composition and distribution of floodplain forest communities (De Jager et al. 2012).

Invasive Species

Invasive plant species also have significant impacts on UMRS floodplain forests by suppressing regeneration and out-competing the native vegetation for water, sunlight, nutrients, and space. There are a large amount of invasive plant species in the UMRS and the number continues to grow, but river managers have identified a select number of invasive and/or weedy species of special concern. These include reed canary grass (*Phalaris arundinacea*), johnsongrass (*Sorghum halepense*), European buckthorn (*Rhamnus cathartica*), various species of honeysuckle (*Lonicera spp.*), white mulberry (*Morus alba*), black locust (*Robinia pseudoacacia*), garlic mustard (*Alliaria petiolata*), Japanese knotweed (*Polygonum cuspidatum*), oriental bittersweet (*Celastrus orbiculata*), Japanese hops (*Humulus japonicus*), crown vetch (*Coronilla varia*), bur cucumber (*Sicyos angulatus*), and trumpet creeper (*Campsis radicans*) (Guyon et al. 2012).

Of these species, RCG is likely the most damaging of all the invasive plant species in the UMRS floodplain forest at this time, especially in the mid- to upper impounded reaches (Pools 1-18). This grass has high light requirements, can establish itself quickly in floodplain forest openings and along edges, and often forms dense monocultures. Dense growth can out-compete tree seedlings and prevent germination of native species, resulting in gradual thinning of the forest canopy, loss of bottomland forest, and an increase in the size and extent of RCG meadows. In addition to providing wildlife habitat, closed canopy forests limit the establishment and expansion of RCG through shading. Partial forest canopies have the potential to provide high quality habitat, but this type of habitat is very difficult to maintain in areas where invasives such as RCG are present (Guyon et al. 2012).

The Reno Bottoms area in upper Pool 9, just south of La Crosse, WI, provides an excellent case study on the combined impacts of altered hydrology and invasive species on UMRS floodplain forest ecosystems. Flat topography, higher groundwater levels caused by river impoundment, increased frequency and duration of floods, and increased competition from RCG have all adversely affected forest cover and regeneration. Reed canary grass (RCG) is widespread in much of the area, and significant overstory tree mortality has been observed in multiple locations throughout, likely due the compounded impacts of the chronic ecosystem stressors mentioned above. The current forest is composed mainly of a few highly flood tolerant species such as silver maple, many of which are now mature and may soon be approaching the end of their life span. As overstory trees die off, younger trees are generally missing throughout the area where RCG competition is particularly problematic.

Resource managers and others initiated a hydrogeomorphic model (HGM) of the Reno Bottoms area in 2009 as a preliminary step in the development of a restoration plan to address these resource management issues. Hydrogeomorphic models can be a valuable tool in identifying ecosystem restoration options and providing management recommendations at a variety of spatial scales in large river floodplain systems such as the UMRS. The HGM process generally includes three stages: 1) determining historical conditions and ecological processes of an area from historical information including geological, hydrological, and botanical maps and data; 2) determining ecosystem alterations by comparing historical versus current landscapes; and 3)

identifying options and approaches to restore specific habitats and ecological conditions (Heitmeyer 2007, Heitmeyer 2008, Theiling et al. 2012).

The Reno Bottoms HGM project was designed to address a lock and dam 8 embankment modification project as well as floodplain forest restoration options across the larger Reno Bottoms area. Key findings included: the development of an HGM “matrix” of potential historic habitat distribution related to geomorphic, soil, elevation, and flood frequency conditions; a summary of major changes that occurred in the Reno Bottoms ecosystem; and projected impacts to floodplain forest distribution based on potential lock and dam embankment modification alternatives (Heitmeyer 2009).

Outputs of the HGM project indicated that historic floodplain forest communities generally occurred above elevations of 623 feet in the 2-5 year flood frequency zone. Wet meadow, shrub/scrub, and marsh habitat occurred at lower elevations, and some prairie habitat occurred on higher elevation tributary fans above 628 feet. Hydrologic changes resulted in wetter conditions throughout Reno Bottoms, with more frequent and prolonged flooding and saturated soil conditions during the growing season. This resulted in expansions of marsh, wet meadow and aquatic habitats, and corresponding losses of large areas of floodplain forest. Of particular consequence, the distribution of floodplain forests shifted up by about 2 feet in elevation from historic pre-lock and dam periods and now generally occurs above the 625 ft elevation contour line. Options for restoring floodplain forest communities in the Reno Bottoms area are still being explored by management agencies such as the US Army Corps of Engineers and US Fish and Wildlife Service, among others.

There have been a handful of more recent studies of the impacts of RCG, hydrology, soils, wildlife and their interactions on floodplain forest vegetation communities and associated ecosystem functions in the UMRS. De Jager et al. (2013) found that herbivory can interact with local flooding regimes in rivers to delay recruitment of some tree species, cause shifts in successional trajectories, and leave forests vulnerable to invasion by RCG. De Jager et al. (2015) found that the highest nitrification rates in floodplain settings were found in low-lying areas and during times immediately following inundation. In addition, they found that restoration of forest

cover in areas invaded by RCG appears likely to restore abiotic soil properties and nitrification dynamics. De Jager et al. (2012) found evidence for a threshold along an elevation gradient of the UMRS floodplain, corresponding with flood durations lasting 40% of the growing season. At lower elevation sites, forest soils and vegetation were strongly correlated with flooding, and soils were dominated by silt plus clay with high organic matter and forests by a few highly flood tolerant species. Kreiling et al. (2015) found that seasonal dynamics in floodplain nutrient availability are driven more by inundation than by differences in vegetation community types. They also found that RCG has the potential to increase availability of some nutrients, and that restoration of forest cover may promote the recovery of nutrient availability to levels observed in mature reference forests. Finally, Thomsen et al. (2012) found that RCG exerted strong competitive effects on the establishment and early growth of tree seedlings, that areas treated with herbicides and site scarification had greater establishment of wetland herbs and tree seedlings, and that deer browsing can limit tree seedling height growth in floodplain forest restoration sites.

The three UMRS Army Corps of Engineers Districts, headquartered in St. Paul, MN, Rock Island, IL, and St. Louis, MO have employed a number of forest restoration measures in specific recognition of the management problems posed by RCG. These include planting larger root production method (RPM®) trees that already extend above the height of RCG at the time of planting, using tree mats and tubes to reduce root competition and limit damage by voles and other rodents, planting cuttings or bare root stock where applicable, scarifying sites prior to planting, and/or using herbicides. These techniques have resulted in varying degrees of success and are continually being refined under adaptive management protocols (Guyon et al. 2012).

Additional Considerations

With specific regard to forest resources system-wide, significant canopy die-off is anticipated in many locations within about 50 years due to the mature, even-aged condition of the majority of UMRS floodplain forests (USGS 1999). This has the potential to create open sunny conditions that will promote undesirable species such as RCG, and make it difficult for floodplain forest trees to regenerate. Without active management, some of the expected changes over the next 50 years include (*adapted from Urich et al. (2002) and Guyon et al. (2012)*):

- A reduction in cottonwood and willow. These are typically pioneer species that become established on newly accreted islands or exposed substrates. They require open sunlight and will not regenerate in the shaded understory of an established forest.
- A more open forest canopy. Much of the current floodplain forest is closed canopy, where trees are spaced close enough together to create a continuous layer of upper tree crowns. As these trees age, die off and fall to the ground, openings will be created. If conditions are not suitable for regeneration of trees, these canopy gaps will likely be invaded by herbaceous vegetation (e.g., RCG) and remain in an open condition for many years. Even if conditions are suitable for tree regeneration, maple and ash may continue to dominate.
- Continued loss of forest in the lower parts of pools. Gradual loss of islands to erosion will also result in less overall forest area and fewer trees.
- Conversion from forest to other vegetation types in mid-pools. As a result of dam construction and water level control, the water table is higher in islands and shorelines located within the lower and middle portions of each pool. This creates site conditions that may be less suitable for forest, but better for other species, such as RCG. Thus, the trend may be a gradual replacement of forest species with herbaceous vegetation.
- Fewer mast trees. Mast trees, such as oaks and hickories, are generally less tolerant of flooding and saturated soil conditions than other floodplain tree species. They also produce a heavy seed, which is not as widely dispersed as the lighter, wind-carried seeds of cottonwood, willow, maple, and ash. These factors will likely contribute to a continued reduction of mast trees within these floodplains.
- An increase in shade tolerant species. Box elder and mulberry are highly tolerant of shade. Since much of the current forest is dense canopy, it is likely that these two species will increase through natural establishment in the understory of existing maple stands. Although there is some habitat value associated with them, box elder and mulberry are generally not considered as desirable as other floodplain tree species.

The use of tree plantings is of course a major component of any active forest management program with restoration goals, and there are even a variety of silvicultural options such as interplantings that may have added benefits to bird and wildlife species. Such efforts have been studied in the Lower Mississippi Alluvial Valley (LMAV) (Gardiner et al. 2004), and results look promising for the UMRS as well. The basic approach was to allow a young cottonwood

stand to be established, often through natural regeneration, and then interplant it with desirable oak species. Although the biomass of oak saplings interplanted beneath cottonwood was reduced relative to open-grown oaks, height growth was comparable. Furthermore, planting or encouraging the natural regeneration of fast growing tree species like cottonwood in conjunction with mast-producing species has been shown to encourage rapid avian colonization in the Lower Mississippi Alluvial Valley, and may therefore be preferred over monotypic plantings of oaks in areas where the establishment and promotion of forested avian habitat is a management goal (Twedt and Portwood 1997, Wilson and Twedt 2005).

Although utilizing cottonwood as a successional “nurse” species has not been proven to enhance oak survivorship, it has been shown to not be detrimental to survivorship or growth either, and there are other ecological benefits recognized in the literature. For example, Twedt et al. (2002) found that young cottonwood stands supported greater bird species diversity than young stands planted with oak species. The authors attributed this to the rapid assimilation of forest structure by fast growing cottonwood stands. Slower growing oak plantings generally supported avian species more characteristic of grasslands. Wilson and Twedt (2003) found that bottomland hardwoods stands and cottonwood stands supported different communities of spring migratory songbirds, suggesting that a mosaic of forest types including cottonwood stands may benefit overall bird species diversity at a landscape scale. Hamel (2003) found that winter bird species communities contained twice as many species in young cottonwood stands than in young oak stands, and also attributed this finding to vegetation structure. Although all of these studies took place in the LMAV, they likely have applicability in the UMRS as well.

Wildlife Habitat

Upper Mississippi River System floodplain ecosystems cover approximately 2.6 million acres and support more than 300 species of birds, 57 species of mammals, 45 species of amphibians and reptiles, 150 species of fish, and nearly 50 species of mussels. The forest and grassland ecosystems of the UMRS are especially important habitat for many species of birds during migration and nesting, and they provide critical habitat for a number of rare and declining species in addition to federal and state-listed threatened and endangered species. It is a migratory flyway for more than 40 percent of North America’s migratory waterfowl and shorebirds, and a

globally important flyway for 60 percent of all bird species in North America (USACE 2004). A 261-mile portion of the Upper Mississippi River was designated a Globally Important Bird Area in 1998 because it harbors significant numbers of waterfowl, raptors, wading birds and song birds. It is also important habitat for 286 State-listed or candidate species and 36 Federally-listed or candidate species of rare, threatened, or endangered plants and animals endemic to the Upper Mississippi River Basin (USACE 2004).

Some bird species such as bald eagles, great blue herons, great egrets, and cerulean warblers favor taller trees such as cottonwood and swamp white oak for roosting and nesting habitat (Urich et al. 2002). However, studies have shown that only a minor amount of natural cottonwood and oak regeneration is occurring on the floodplain (Yin et al. 1997, USGS 1999). Some of these birds now utilize silver maple as a substitute to tall trees, yet future widespread occurrence of even silver maple may also be in question in some areas due to competition with RCG (Urich et al. 2002). Tall tree habitat will likely continue to diminish without active management promoting growth of these types of trees.

Other bird and wildlife species require large, contiguous closed canopy forests to maintain viable populations. These species, such as the red-shouldered hawk, are referred to as area-sensitive. However, loss of forested habitat has resulted in the fragmentation of forests into smaller, disconnected patches, and loss of healthy floodplain forests results in the loss of wildlife habitat. For example, areas with large blocks of interior forest meet the needs of area-sensitive bird species, including red shouldered hawks, cerulean warblers, Acadian flycatchers, prothonotary warblers, veerys, wood thrushes, pileated woodpeckers, and eastern wood peewees (Knutson et al. 1996). Some floodplain bird species may respond more to forest width than edge vs. interior habitat or habitat patch size (Kirsch and Gray 2009), and the concept of forest interior-dependent species may be less applicable in situations where forest “patches” are surrounded by a mosaic of other natural habitats rather than row crops, but it is generally agreed that floodplain forests support a greater number of bird species than other Upper Mississippi River habitats (USGS 1999).

In some areas, silver maple forests continue to occur in large contiguous blocks of habitat. However, if current low levels of natural tree regeneration are not reversed, even these forests may become more fragmented by RCG and other herbaceous vegetation and thus less suitable for species with these life history requirements (Urich et al. 2002). In general, conditions for UMR floodplain birds will deteriorate if floodplain forests continue to decline, become more open-canopied, and disappear from the landscape (Knutson et al. 1996). The conservation of the floodplain forest bird community therefore depends on system-wide efforts to restore degraded floodplains, maintain wide forested corridors, and provide hydrologic conditions that promote the natural regeneration of a high diversity of tree species including oaks, silver maple, ash, cottonwood, sycamore, and sweetgum (Knutson et al. 1996).

One notable feature of the breeding bird community in Upper Mississippi River floodplain forests is the dominance of the community by birds that breed here and winter elsewhere, and many neotropical and short distance migrant birds that use Upper Mississippi River floodplain forests and associated habitats are of management concern nationally, regionally, or for certain Upper Mississippi River States (Guyon et al. 2012). The abundance of cavity nesters in UMR floodplain forests indicates the ecological significance of standing dead trees. The size and abundance of snags, dead trees and live trees with large dead limbs in the UMR floodplains versus the uplands are caused by differences in tree species and hydrological regimes among other factors.

Natural forest stand dynamics and different types of forest restoration methods can influence forest structure, which in turn can influence bird and wildlife populations. For example, Twedt et al. (2002) found that young cottonwood stands supported greater bird species diversity than young stands planted with oak species. Fast growing cottonwood stands quickly established forest structure, whereas slower growing oak plantings generally supported avian species more characteristic of grasslands. Hamel (2003) found that winter bird species communities contained twice as many species in young cottonwood stands than in young oak stands, and attributed this finding to vegetation structure. Wilson and Twedt (2003) found that bottomland hardwood and cottonwood stands supported different communities of spring migratory songbirds, which

suggests that a mosaic of forest types may increase overall bird species diversity at a landscape level. However, all of these studies took place in the LMAV.

Bottomland forests along the UMR support migrating and nesting populations of bald eagles (*Haliaeetus leucocephalus*), ospreys (*Pandion haliaetus*), red-shouldered hawks, and other raptors. Although the bald eagle was de-listed from the Endangered Species Act in 2007, it is still protected under the Bald and Golden Eagle Protection Act and the Migratory Bird Treaty Act (USFWS 2007). The red-shouldered hawk is listed as endangered in Iowa, threatened in Wisconsin, and of special concern in Minnesota. The UMR floodplain contains a considerable amount of forested habitat and is thus important for maintaining red-shouldered hawk populations in these States and providing a corridor for linking the habitats of northern and southern populations.

The Upper Mississippi River is an important nesting and feeding area for great blue herons, double crested cormorants and great egrets because extensive bottomland forests and diverse aquatic areas provide suitable nesting and foraging habitat. Herons require large mature trees like cottonwood and swamp white oak for nesting, and these trees are natural components of the mature silver maple dominated forests of the Upper Mississippi River floodplain (Knutson and Klaas 1998, Yin 1998, Urich et al. 2002).

Waterfowl are likely the most visible and certainly the most economically important group of bird species on the river system. Two species of forest nesting waterfowl can be found on the Upper Mississippi River – the wood duck and hooded merganser. Both of these species nest in large cavities in trees over or near water. Wood ducks are omnivorous but a large part of their diet consists of acorns, seeds and berries. Hooded mergansers are primarily piscivorous, supplementing their diet with crustaceans and aquatic insects (Guyon et al. 2012).

Floodplain forests of the UMR also provide habitat for many mammals, amphibians, and reptiles. There are over 50 species of mammals on the UMR, of which at least 28 are associated with forest habitats. Terrestrial mammals such as the white-tailed deer, red fox, gray fox, coyote, squirrels, raccoon, and opossum are found in abundance, primarily inhabiting the river's

floodplain and islands. Bobcat and black bear are occasionally observed in the upper reaches of the Upper Mississippi River, primarily above Pool 11. Aquatic mammals, such as the river otter, beaver, mink, and muskrat are commonly observed along the riverbanks and/or backwaters. A few species rely on cavities in the floodplain forests for shelter and the flying insects that are produced in and along the river for food (Urich et al. 2002). Indiana (*Myotis sodalists*) and Northern long-eared bats (*Myotis septentrionalis*) are both currently listed on the Federal Endangered Species Act (USFWS 2015) and roost in bottomland and floodplain habitats. On average, they prefer large trees with a dbh of 37-39cm, and Indiana bat maternity colonies require large snag trees that are beginning to sluff their bark in order to raise their pups. These mothers will use several different trees, moving their offspring to new locations as conditions vary. In some cases live shagbark hickories are also used if a suitable snag cannot be found. One of the important factors for suitable roosts is the proximity of the roost to the forest. Snags that are in an open area like a swamp or field are not likely to be used by these species. Roosts are usually in a small opening in the forest or within 50m of the forest edge (Carter and Feldhamer 2005). White-tailed deer occur throughout the floodplain forest, and can influence tree regeneration. There are also at least 40 species of reptiles and amphibians on the UMR floodplain, and about 22 of these species are associated with floodplain forest habitats (Urich et al. 2002).

Climate Change

The potential long-term impacts of climate change on floodplain forests in the Upper Mississippi River System are not well known at this time, but some inferences can be made based on predicted changes to temperature and precipitation patterns in the Upper Mississippi River Basin. Warmer temperatures, longer growing seasons, and increased atmospheric CO₂ levels all have the potential to increase productivity in forested ecosystems (Ryan et al. 2008). However, climate change may also affect the frequency of natural disturbances such as fires, floods, insect outbreaks, ice storms, and windstorms (CCSP 2008). Some climate models link projected increases in precipitation over the Upper Mississippi River Basin to increased runoff, but considerable uncertainty remains (Lettenmaier et al. 2008). Increased rates of precipitation and associated runoff could lead to increased stresses on river floodplain ecosystems. In addition, climate change has the potential to affect biodiversity in the UMRS through changes in growing

season length, species distributions and phenology, and other components of ecosystem function (Janetos et al. 2008).

The U.S. Forest Service has completed a significant amount of work mapping the potential response of tree and bird species in the eastern United States to various climate change scenarios (Prasad et al. 2009). Results of these analyses are available via the Climate Change Tree and Bird Atlases, which are maintained by the U.S. Forest Service and available at:

<http://www.fs.fed.us/nrs/atlas/>.

Relevant federal initiatives responding to the potential risks posed by climate change include the U.S. Fish and Wildlife Service's Climate Change Strategic Plan (USFWS 2010) and the U.S. Forest Service's Strategic Framework for Responding to Climate Change (USFS 2008). Both plans emphasize mitigation, adaptation, and advancing efforts to share knowledge and build collaborative partnerships as key strategies to address climate change.

Floodplain Soils

Floodplain soils were formed by erosion and deposition processes associated with the lateral movement of rivers and streams, and therefore, their physical and chemical properties are a reflection of those processes. Rivers and streams carry loads of suspended sediments that are deposited in the floodplain during flooding events. When floodwater exceeds bank height, the floodplain is inundated with sediment-laden water. Course sediments settle out within the river channel or near the banks where the water is more turbulent and has more energy. Finer sediments settle out further away from the river banks as the floodwater slows and loses energy.

For this report, soils data from the Illinois portion of the Mississippi River floodplain were consolidated and summarized in order to present a representative sample of the entire Upper Mississippi River floodplain. Although soils data were obtained for the entire Upper Mississippi River floodplain, data from other states are still being delineated and summarized. Soils data were obtained from the gridded Soil Survey Geographic Database (gSSURGO) available online through the USDA Geospatial Data Gateway (<https://gdg.sc.egov.usda.gov>) (verified on

07/07/2016). Floodplain dimensions were delineated using 10 m resolution data from the National Elevation Dataset, also available through the Geospatial Data Gateway. ArcGIS was used to extract soil attributes for soils within the Illinois Mississippi River floodplain. Soils data were then grouped according to textural class (Table 1), drainage class (Table 2), and flooding frequency (Table 3).

Table 1 shows that half of the soils in the Mississippi River floodplain in Illinois have a silt loam texture, and when combined with finer textured soils (silty clay loam, clay loam, and silty clay), the total percentage of fine-textured soils is 73%. With such a fine texture, it is not surprising to find that over 55% of the soils in the Mississippi River floodplain in Illinois are classified as somewhat poorly drained, poorly drained, or very poorly drained (Table 2) and 40% of the soils are occasionally or frequently flooded (Table 3).

Table 1. Frequency of the various soil textural classes in the Mississippi River floodplain in Illinois. Textures are listed in order of increasing coarseness.

Textural Class	Area (Ac)	% of Total Floodplain
Silty clay	183,603	5.8%
Clay loam	57,300	1.8%
Silty clay loam	495,715	15.7%
Silt loam	1,573,736	49.7%
Loam	246,069	7.8%
Sandy loam / Fine sandy loam	120,706	3.8%
Loamy fine sand	9,956	0.3%
Fine sand	85,113	2.7%
Sand	132,801	4.2%
Others / undefined	260,088	8.2%
Total for all textural classes	3,165,087	100.0%

Table 2. Size of the various drainage classes in the Mississippi River floodplain in Illinois.

Drainage Class	Area (Ac)	% of Total Floodplain
Excessively Drained	111,103	3.5%
Somewhat Well Drained	15,845	0.5%
Well Drained	809,200	25.6%
Moderately Well Drained	242,631	7.7%
Somewhat Poorly Drained	788,262	24.9%
Poorly Drained	872,794	27.6%
Very Poorly Drained	85,887	2.7%
No classification	239,363	7.6%
Total for all Classes	3,165,087	100.0%

Table 3. Size of areas exhibiting various levels of flooding frequency in the Mississippi River floodplain in Illinois.

Flooding Frequency	Area (Ac)	% of Total Floodplain
Frequent	610,478	19.3%
Occasional	689,072	21.8%
Rare	208,873	6.6%
None	1,331,280	42.1%
No classification	325,384	10.3%
Total for all Classes	3,165,087	100.0%

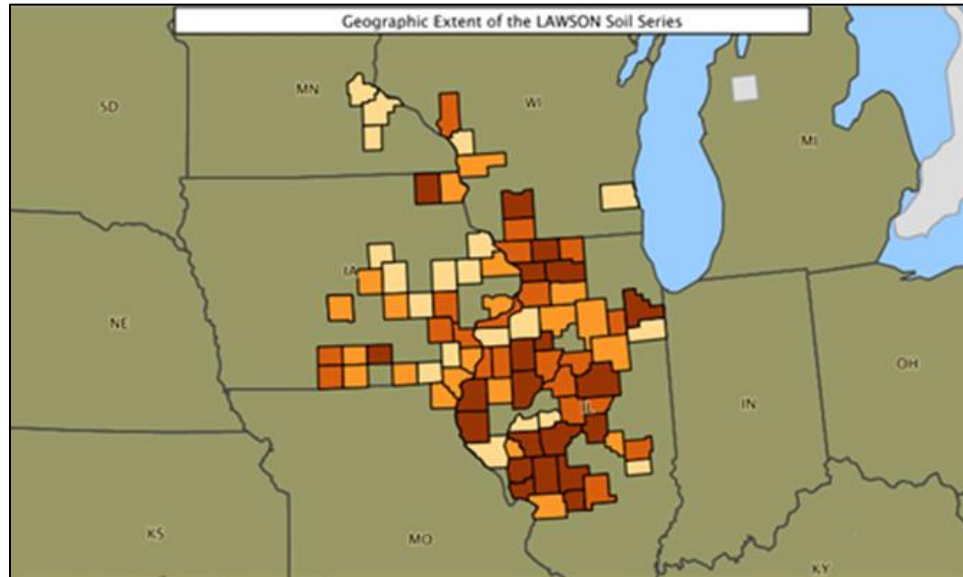


Figure 3. Spatial distribution of the Lawson soil series within the Upper Mississippi River watershed. Although the Orion series is not found exclusively in the Mississippi River floodplain, it is typical of floodplain soils.

While it is not possible to examine every soil type in the Mississippi River floodplain, the Lawson soil series represents a typical soil found in the Mississippi River floodplain.

Figure 3 shows the geographic extent of the Lawson Soil series. Although the Lawson series is not found exclusively in the Mississippi River floodplain, it is typical of floodplain soils and is found in the floodplains of many tributaries of the Mississippi River, including the Illinois and Des Moines rivers. The Lawson soil series consists of very deep, somewhat poorly drained soils formed in silty alluvium. These soils occur on flood plains and upland drainageways where slopes range from 0 to 5 percent. Table 4 lists the description of a typical Lawson soil profile. Features that are common in floodplain soils include a relatively dark, thick, fine-textured A horizon that forms directly on the parent material C horizon. The A horizon typically has a high organic matter content. The C horizon is also typically fine-textured, but may also contain courser sand and gravel materials. When the Lawson soil exists in an undisturbed condition, natural vegetation consists of scattered silver maple, white ash, American elm trees, tall prairie grasses, and forbs. For agricultural purposes, it is frequently used for forage production, with RCG being one of those forages (Martina and von Ende 2008). Cultivated areas of Lawson soil, and similar floodplain soils, produce good crop yields where excess water is not a problem. The Lawson soil series is typical of soils that are susceptible to RCG infestations, but

there is still a need to elucidate the actual soil characteristics that contribute to RCG infestations Bernthal and Willis (2004).

Table 4. A physical description of a typical Lawson Soil Series profile.

TAXONOMIC CLASS: Fine-silty, mixed, superactive, mesic Aquic Cumulic Hapludolls
TYPICAL PEDON: Lawson silt loam, on a 1 percent slope, in an uncultivated field, at an elevation of about 265 meters above mean sea level. (Colors are for moist soil unless otherwise stated.)
A1 --0 to 30 centimeters; very dark brown (10YR 2/2) silt loam, grayish brown (10YR 5/2) dry; weak fine subangular blocky structure parting to moderate medium and fine granular; friable; many roots; neutral; clear wavy boundary.
A2 --30 to 48 centimeters; black (10YR 2/1) and very dark brown (10YR 2/2) silt loam, dark grayish brown (10YR 4/2) dry; weak fine subangular blocky structure parting to moderate medium and fine granular; friable; many roots; neutral; clear wavy boundary.
A3 --48 to 76 centimeters; black (10YR 2/1) and very dark brown (10YR 2/2) silt loam, grayish brown (10YR 5/2) dry; moderate fine subangular blocky structure; friable; neutral; clear wavy boundary. (Combined thickness of the A horizon is 60 to 90 centimeters.)
C1 --76 to 102 centimeters; very dark gray (10YR 3/1) and black (10YR 2/1) silty clay loam stratified with thin lenses of silt loam and loam; moderate medium angular and subangular blocky structure; firm; neutral; clear wavy boundary. (0 to 39 centimeters thick)
C2 --102 to 152 centimeters; dark grayish brown (10YR 4/2) silt loam interlayered with thin lenses of loam and sandy loam; massive with a few thin, coarse-textured strata; friable; common fine prominent yellowish brown (10YR 5/6) masses of iron accumulation in the matrix; neutral.

Forest Impacts on Floodplain Hydrology

The way that water moves over the surface of a landscape and through the subsurface is affected by the predominant type of vegetation covering the landscape. Any changes to the land cover within a watershed will have an impact on how water moves through that landscape. On a global scale, it is clear that human impacts, such as deforestation for agricultural purposes, results in flashier flood events, accelerated streambank erosion, and the delivery of greater amounts of sediments into streams, rivers, and ultimately oceans (Turner and Rabalais 2003, Walling 2006).

Evidence shows that the presence of trees in the landscape provides beneficial impacts on soil hydrological processes. Kellner and Hubbart (2016) monitored soil moisture content for three years within a historic agricultural field and a 100-year old remnant bottomland hardwood forest and found that the forest soil profile contained more water during wet periods due to the presence of preferential flow paths into the subsoil. Similarly, Marshall et al. (2009) found surface runoff from an area with a 10-year old stand of broadleaf trees was less than that from adjacent grassland and attribute it to a higher rate of saturated hydraulic conductivity in the forest soils resulting in more rapid infiltration of water into the soil profile. The evidence clearly suggests that floodplain forests have the potential to more effectively mitigate flood risk, as compared to grassland/agricultural sites. Furthermore, the benefits of converting grassland to forest, especially grazed grassland, can occur quickly. When a grassland pasture was converted to a mix of broadleaf trees, surface runoff was reduced by 78% during the first year, and by 5 years later, infiltrations rates in the forested plots were significantly increased (Marshall et al. 2014).

Hydrologic models are commonly used to explore the impact of various land use practices on runoff water quantity and quality. Although different models are used by researchers, they are all based on well-defined biogeophysical processes with corresponding mathematical algorithms that describe the movement of water through a watershed based on multiple parameters such as precipitation, landscape characteristics (i.e., slope), soil types, land cover (i.e., vegetation), evapotranspiration, rooting depth, and others. Models are first calibrated using actual daily streamflow data from a specific delineated watershed, and longer periods of continuous streamflow data result in stronger calibrations. Approximately half of the daily streamflow data,

generally four or more years, is used to calibrate the model, and the remaining streamflow data is used to verify that the model accurately predicts runoff volumes. Based on the output from a hydrological model calibrated with 8 years of data and validated with 6 years of data, Viola et al. (2014) found that reforesting 20 to 50 percent of the total area of a watershed area that had been converted to grass resulted in 1.7 to 3.7 percent reductions in water yield (i.e., precipitation minus evapotranspiration). Reforestation resulted in reduced minimum and maximum streamflows. Conversely, conversion of forest to grassland resulted in increased streamflow discharge. It is clear that forested watersheds provide greater protection from flashy runoff events and flooding. Ouyang et al. (2013) used a similar modelling approach (six years of calibration data and 5 years of validation) in the Lower Yazoo River in the Lower Mississippi River Basin to show that floodplain reforestation in the area confined by river levees would result in reduced flow discharges and lower sediment loads.

Regardless of whether the evidence comes from field measurements or models, it is clear that forested riparian areas provide greater benefits to floodplain hydrological functions than grasslands or agricultural lands. Increased infiltration in forests can reduce the overall volume of runoff water as well as attenuate the intensity of runoff events. On large rivers, forests are better at protecting streambanks from erosion than are grasslands (Lyons et al. 2000).

Forest Impacts on Sediment Loads in Rivers and Streams

Sediment loads in streams and rivers is directly impacted by how water moves through the landscape. Consequently, human activities in the landscape have short and long term impacts on sediment transport. Just as vegetation affects watershed and floodplain hydrology, it also affects the erosion and deposition of sediments transported by water through the watershed. Humans began to significantly modify the Mississippi River watershed upon the arrival of European settlers beginning in the 18th century (Schoenholtz et al. 2001). Since early settlers were primarily farmers, most of the landscape modification in the floodplains consisted of removing trees and shrubs to create tillable agricultural fields. Turner and Rabalais (2003) looked at the historical record of sediments transported through the Mississippi River watershed for the past 200 years and found that sediment loads were clearly increased by the clearing of forested

landscapes. Walling (2006) examined current trends in the sediment loads of major rivers around the planet and found multiple examples, such as the Magdalena River in Colombia, where increasing sediment loads are attributable to deforestation of the landscape combined with agricultural intensification.

Deforestation of native forests clearly results in the erosion of soils and an increase in sediment loads in rivers and streams. This occurred not only in the Mississippi River basin, but in the watersheds of large rivers around the planet, and it continues to be a problem in many developing countries. Consequently, the focus of many conservation programs is to reduce sediment transport through the watershed, and the most effective strategy to accomplish this is to convert highly erodible croplands to permanent vegetation, especially in the riparian zone around the river.

The options for creating permanently vegetated riparian zones around rivers and streams include the use of grasses, trees, or a combination of the two. Vegetated riparian zones help reduce the sediment loads in rivers and streams through two mechanisms: 1) streambank stabilization; and 2) filtration of sediments from surface runoff. Evidence that permanently vegetated surfaces can effectively protect water quality is obvious when considering the reverse process, i.e., the conversion of permanently vegetated land to cropland. Schilling et al. (2015) found that conversion of adjacent forest and grass riparian zones to a corn/soybean crop rotation resulted in large increases in groundwater nitrate concentrations. Their data did not suggest whether forests or grasslands are more effective at removing sediments from runoff water, but in a riparian vegetation strip along the Adour River in Southwestern France, Brunet and Astin (2008) found that larger quantities of sediments were deposited in wooded areas than in non-wooded sections. Using calibrated models, Ouyang et al. (2013) determined that reforestation in the riparian zone of the Lower Yazoo River in the Lower Mississippi River Basin would result in reduced sediment loads. In contrast, Lyons et al. (2000) concluded that well maintained grassy riparian zones can be effective at preventing stream bank erosion and trapping suspended sediments in watersheds <250 km² and with streams <10 m wide. However, forested riparian zones were more effective at stabilizing severely eroding stream banks on wider channels in larger watersheds.

Water Quality Parameters

Nutrient loading and sediment deposition from non-point sources are the primary drivers of poor water quality in rivers and streams (Turner and Rabalais 2003). Nutrient dynamics in floodplains have a strong effect on the ecology of their associated rivers given the high level of connectivity during flooding events. Many consider floodplain forests in the UMR to be ecological sinks for nutrients and sediments that enter the system through run-off and erosion (Pinay et al. 2000). The manner in which nutrients are processed and assimilated by river and floodplain biota depends on several factors and on the nutrient of interest (Baldwin and Mitchell 2000). Nitrogen dynamics are dependent on bacterial communities, temperature, oxygen, and discharge. Phosphorus dynamics are strongly tied to sediment transport and erosion processes in the river, as well as to biotic uptake through primary production. Availability of both nitrogen and phosphorus to terrestrial vegetation and macrophytes depends on the flooding regime and degree of soil desiccation between inundation events (Baldwin and Mitchell 2000). It is possible for the floodplain to act as a source of nutrient and sediment pollution to the river. The frequency and duration of flooding events, along with the landscape type of the area flooded, can affect whether the floodplain acts as a nutrient source or sink.

Nitrogen

Excessive loads of nitrogen (as nitrate) can cause devastating effects on aquatic ecosystems through eutrophication. Floodplain forests have the capacity to remove nitrates from river systems through the process of denitrification that is performed by facultatively anaerobic bacteria (Lowrance et al. 1984, Brettar et al. 2002). Floodplain forests and riparian zones have been shown to act as important nitrate sinks in river systems (Burt et al. 1999, Forshay and Stanley 2005, Wriedt et al. 2007, Aquilina et al. 2012). Physical parameters within floodplain forests, such as high water table levels, elevated water temperatures, and sufficient amounts of organic carbon, increase denitrification rates (Kronvang et al. 1999, Richardson et al. 2004). Rates are also increased when the system is anoxic and nitrate concentrations are high. Given these requirements for denitrification to occur, there are few areas in the UMRS which provide suitable conditions. Water flowing through the main river channel tends to be highly oxygenated, and does not regularly carry high concentrations of organic matter. Despite

floodplain forests covering a relatively low percentage of the river banks in the UMRS, they have the ability to have a disproportionately high importance on nitrate fluxes within the system (Cooper 1990). Denitrification rates have been found to be at least four times higher in floodplains than in river channels (Natho et al. 2013). These rates vary throughout the year, and are often dependent on river discharge. Patches of wet organic soil with low oxygen content are regularly present in floodplain forests, and are most abundant where water tables are high and when flooding is a typical occurrence. The formation of these pools in warm summer months leads to excellent conditions for denitrifying bacteria to be productive. Efforts to model nitrogen removal through this process must take into account the rate at which water is withdrawn from the floodplain (Pinay et al. 2000, Richardson et al. 2004, Natho et al. 2013). Event-based hydrology models that are incorporated into nutrient dynamic predictions for river systems should include geomorphic controls on denitrification. Natho et al. (2013) found that nitrogen is retained in floodplain forests more efficiently when short-term flooding events occur regularly. Small-scale soil inundation events improve N-removal capacity in these systems. Large-scale events, on the other hand, tend to cause floodplains to act as nitrogen sources to the river system. Determining the effect that floodplain forests have on nitrogen dynamics in the Mississippi River goes hand-in-hand with the hydrograph.

Vegetation type is an important factor to consider when assessing N retention and removal in floodplains. Assimilation of N into plant biomass occurs in both forested and grassland (e.g. RCG) floodplains, but the rate of return after plants die and decompose varies. Hefting et al. (2005) found that forested riparian zones in several sites throughout Europe had higher rates of N uptake and retention than herbaceous buffers. This difference is likely due to plant biomass production rates between sites. Floodplain trees have high growth rates in early forest successional stages, quickly developing roots, stems, branches, and leaves. It is during this time of accelerated biomass additions that their rates of nutrient uptake from the soil are greatest (Ericsson 1994). Nutrient removal from the floodplain through plant uptake decreases as vegetation ages. While maturing forests experience a decline in their rates of nutrient sequestration through plant uptake, they are increasing their deposition rates of organic material through leaf litter and fine root turnover. Decomposition of this organic material leads to higher denitrification rates. Active management of grasslands, through harvesting or mowing, can

result in higher production rates, and thus higher N removal rates (Osborne and Kovacic 1993, Hefting et al. 2005). However, this would be a labor intensive management practice, and impractical on a large scale. Osborne and Kovacic (1993) found that forested buffer strips are more efficient at filtering out nitrates in shallow ground water than grassland buffer strips. Specifically, they measured nitrate-N concentrations in shallow lysimeters and found that grassland systems had about three times more nitrate than forested systems. Kreiling et al. (2015) had similar results in a study that assessed a suite of ion concentrations in an Upper Mississippi River floodplain forest. Reed canary grass plots were found to have significantly higher nitrate supply rates compared to some forest plots (Tukey pairwise tests, $p < 0.05$). These findings support the idea that extensive RCG invasions might increase nitrate concentrations in the UMRS. However, more research is needed on the interplay of seasonality and timing of soil inundation events with regards to the effect of RCG on overall nitrogen dynamics.

Pollution reduction

Floodplain forests have the capacity to absorb chemicals without returning those chemicals to the environment through leaf fall and decomposition (Lowrance et al. 1984). The strongest influences of pollution in river systems come from precipitation and runoff within agricultural land in the catchment. Trees can process organic pesticides that originate from non-point sources through biomass assimilation. Pesticides are biologically transformed and assimilated into plant tissues, at which point they are able to be degraded (Lin et al. 2004, Dosskey et al. 2010). Other organic and inorganic compounds that are able to be volatilized (e.g. benzene, trichloroethylene, toluene, selenium, organo-mercury) are also removed from the environment by floodplain forests, and subsequently released into the atmosphere through tree respiration (Hussein et al. 2007, Dosskey et al. 2010).

It should be noted that different plant species in riparian zones and vegetated floodplains have unique physiological processes, and thus vary in their capabilities to take up and degrade herbicides and pesticides. For example, Lin et al. (2004) found significant differences in atrazine tolerances between C3 and C4 grasses, with the C3 grasses used in their study being more sensitive to the common herbicide chemical. In contrast, they found that the C4 grasses used in their study were more sensitive to isoxaflutole, which is the active ingredient in other agricultural

herbicides (e.g. Balance™). These results emphasize the importance of species diversity among floodplain plant communities in removal of anthropogenic pollution. Less information is available on the effectiveness of woody plants in removing atrazine or isoxaflutole. Burken and Schnoor (1997) found that hybrid poplar trees exhibit the ability to degrade atrazine. This is one of the few studies that investigated herbicide removal capabilities for woody perennials, as opposed to grasses or other annual crops. More research is needed on the ability of various floodplain forest tree species in the Upper Mississippi River to improve water quality through phytoremediation.

Ecosystem Services Classification and Questions

As defined in the Millennium Assessment (MA), ecosystem services (ES) are “the benefits people obtain from ecosystems.” (Millenium Ecosystem Assessment 2005) Human influence on our environment has led to a decline in the integrity of ecosystems causing reduction in the quality of services we receive, and consequently, a reduction in quality of human livelihoods (Bullock et al. 2011). Since they can be considered natural capital, interest in ES is growing both environmentally and economically, and there have been attempts to place economic values on them. However, that approach risks the possibility of not accounting for all benefits (e.g. recreation, wildlife habitat, aesthetic appeal) of an ecosystem (Keeler et al. 2012). Placing a value on selected services can be useful in evaluating the tradeoffs between potential actions and provide another way of examining how land use changes affect people worldwide. For example, Costanza et al. (2014) estimated that between 1997 and 2011, there was a loss of \$20.2 trillion/year in ES due to actions like deforestation and loss of wetlands. Although these estimates come with high standard deviations and variation in the biomes analyzed, they believed these estimated losses were conservative.

MA is an assessment framework that has been widely used since 2005 and classifies ES into four categories; provisioning, regulating, supporting, and cultural (Millenium Ecosystem Assessment 2005), but accurately quantifying ES at all the different levels of an ecosystem is difficult. An issue that can occur is double-counting also known as service-overlapping. This is when a single service is considered in two different categories. Each of these has a value and are aggregated;

leading to an overestimation of the service's value (Ojea et al. 2012). Other issues that can arise from ES valuation are changes in value based on supply and demand effects. Does a service increase in value because it becomes rarer? (Turner et al. 2010). This could imply that an ES is more valuable as a system degrades or is lost; resulting in postponing conservation or restoration until the ecosystem is too far degraded for full function to be restored or loss of the system entirely because other actions are more profitable.

Another issue is that there are knowledge gaps of the complexities of ecosystems and how they function and interact, and no single ES valuation system can encompass all the ecosystem's complexities of a system. There is also debate about what scale is most accurate for an ecosystem or landscape when measuring ES. Do different services have different appropriate scales? Even with uncertainties, in today's national and world economy, not understanding the potential value of an ecosystem can result in the degradation of that system in pursuit of another resource, and valuation of ES can be an important tool for management decisions. Even though there is no perfectly accurate way to quantify the value of services gained from ecosystems, the concept of ES is still relevant and cannot be overlooked by the conservation community. Previous sections have described how forest cover or reforestation affects water quality, soil properties, and hydrology, but forests and woodlands provide other ecosystem services as well.

Forests and Water Availability

There is debate about the costs and benefits of reforestation efforts. One argument is that having trees in an area decreases water availability. Trees have deep-reaching roots with the potential to lower the water table, and the canopy intercepts precipitation before it reaches a waterway (Brown 2013). Other studies concluded the opposite and argue that forests help maintain the water cycle for terrestrial ecosystems (Makarieva et al. 2006, Ellison et al. 2012). The main difference separating these two views is scale. Studies that support the idea that trees reduce water availability were performed at small scales. The other side of the argument suggests that at a regional/global scale, forests act like biological water pumps.

Precipitation is often considered an abiotic factor that determines or constrains an ecosystem, but researchers are finding that this idea may be misleading (Makarieva et al. 2006). Forests may not

simply be a product of a wet environment, but be drivers for the water cycle in their environment. Through multiple mechanisms, trees recycle water back into the atmosphere, recharging and intensifying the terrestrial water cycle. Good et al. (2015) found that on a global scale, 74% of continental precipitation returned to the atmosphere through evapotranspiration including 20% that was due to interception of precipitation by vegetation and 48% due to transpiration alone. A study by Ellison et al. (2012) modeled this process in the Mississippi River Basin and found 58% of the precipitation was generated by evapotranspiration. They also concluded that seasonality may play a large roll with summer evapotranspiration rates being the largest contributor to precipitation.

The forest structure itself decreases the out-flowing of atmospheric moisture. The combination of increasing moisture in the air and preventing the loss of that moist air helps recharge the local and regional water cycle and maintains soil moisture locally (Makarieva et al. 2006). This effect is never more important than during a drought. Trees can better maintain their evapotranspiration rates during dry periods due to their roots' ability to access water from a greater depth (Zhang et al. 2001). Furthermore, when a forest is mature enough to develop a closed canopy; it can lead to a temperature gradient from ground to canopy. This change in temperature keeps water vapor under the canopy in aerostatic equilibrium and reduces water loss from the soil. In woodlands where there is open canopy or in grasslands, this equilibrium does not occur, and water is quickly lost from the soil to the atmosphere (Makarieva et al. 2006) and could account for 6% of the evapotranspiration. There are still many unknowns about soil evaporation and the factors, including ecosystem structure, that affect it (Good et al. 2015).

Canopy closure is not the only factor that can affect a forest's interaction with the atmosphere. The canopy structure is also important with a positive correlation between surface roughness and drag (Ellison et al. 2012). The combination of soil moisture availability, prolonged evapotranspiration rates, and reduced loss of atmospheric moisture through increased drag supports a forest's ability to reduce drought affects and recharge the water cycle in the area. Canopy roughness is caused by two factors, age class and species diversity.

Biodiversity effects on Ecosystem Services

Biodiversity is intrinsically valued by some as a product of evolution, but others see it as a mechanism for ecosystem stability. One of the goals of many restoration projects is to increase biodiversity, and there have been a growing number of studies working to understand the relationship between biodiversity and ecosystem function. Reich et al. (2012) found that all species within a system contribute to the ecosystem's function, and this becomes more apparent as time continues. They suggest that this result is due to the complementarity of species within a system through functional diversity and niche partitioning (Reich et al. 2012). Another meta-analysis using data from 103 publications covering 446 ecosystem properties found that biodiversity in plant and microbial communities overall had a positive impact on preventing soil erosion and promoting nutrient cycling. This study also found evidence that populations may become more fluctuating with increased biodiversity, but communities overall become more stable (Balvanera et al. 2006). An analysis of several studies using grassland systems found that 84% of the species studied promoted ecosystem function at least once, and species that promoted a single function for many years were not the same as the species that promoted multiple functions during a single year. Similarly, different species had effects on function under different conditions at varying locations in different years, and even rare species had an impact (Isbell et al. 2011). For environments that are infested with RCG, these results imply a severe problem. RCG forms a dense monoculture preventing diversity, and this leads to a decrease in the function of the ecosystem and availability of ES. RCG may have a positive impact on one or two ES, but monocultures are not stable ecosystems and cannot withstand change while maintaining their function. Under a RCG environment, an ES may be available only during specific conditions and lost during all others. If there are more species in an area, the ES could be available throughout changing conditions and be more dependable.

In forest ecosystems, increased diversity leads to better ecosystem function/services and productivity (Zhang et al. 2012, Gamfeldt et al. 2013). Gamfeldt et al. (2013) concluded that tree diversity in temperate and boreal forests positively affected biomass production and ecosystem services, but noted that not all services are supported universally and trade-offs do occur in these systems. A global meta-analysis found that productivity was 23.7% higher in polyculture

systems than monocultures and generally an increase in productivity with stand age (Zhang et al. 2012). Biodiversity is required for maintaining ecosystem function/services through time as environmental factors fluctuate, and the loss of any species from a system may prove to decrease the effectiveness of an ES (Isbell et al. 2011).

Many studies in the past have focused on the effect of biodiversity on one process. However, there is seldom a species that affects only one process or a process that does not interact with other ecosystem processes. Few studies have examined interactions of multiple processes, and it remains unknown territory in many ways, but one that is critical to understanding biodiversity's effects on ecosystems and their services (Reiss et al. 2009). Although there is evidence linking biodiversity to improved ecosystem function, restoring biodiversity may not inevitably alter or enhance a particular ES. Management decisions that are made to enhance a service must acknowledge that altering or restoring an ecosystem with the sole intent of a particular ES has its own concerns and could lead to further detriment of the system including introduction of invasive species, further loss of biodiversity, and loss of other ES. All ES are complex with multiple factors (Bullock et al. 2011). There needs to be further research into the effects of restoration through time, and how restoration efforts affect biodiversity and ES through time.

Carbon (C) Sequestration

Carbon sequestration by forest ecosystems plays an important role in efforts to mitigate global climate change. Based on a metadata analysis of existing scientific literature, Pan et al. (2011) estimated that the world's forests contain 861 ± 66 Pg C with 55% in tropical forests, 32% in boreal forests, and 14% in temperate forests. Although there are differences among the various forest ecosystems, in general, 44% of the carbon is in the top 1-m of soil, 42% in live biomass (above and below ground), 8% in deadwood, and 5% in litter. A significant difference among forest ecosystems is that boreal forests store 60% their carbon in the soil whereas tropical forests store only 32% in the soil.

Temperate forests in the U.S. have shown a 33% increase in overall land cover from 1990 to 2000 due to forest regrowth in areas once used for agriculture or harvesting (Pan et al. 2011). Old-growth forests act as a net carbon sink, and dominant species often influence the carbon

turnover rates in above-ground stocks. Carbon storage in the soil is also critical, and root traits like depth, chemical outputs, and the diversity of the symbiotic relationships may impact how species affect the carbon storage capabilities of a forest ecosystem (Díaz et al. 2009).

In the Lower Mississippi River Valley, a study was conducted using samples of even-aged planted, natural regeneration, and some uneven-aged natural origin stands; 67 stands total ranging from four to 121 years old. Their carbon storage model resulted in significant predictive power; $R^2 = 0.83$, a $P < 0.0001$. Stands of trees less than 20 years old had a lower carbon sequestration of under 40 Mb C/ha. As the stands aged, carbon storage increased and the storage rate seemed to slowly taper off to a steady level. Stands that were older than 100 years had wide ranging carbon storage between 120 and >240 Mg C/ha (Shoch et al. 2009). The carbon sequestration ability of forests and how species diversity and other diversity factors like relative abundance and functional trait composition affect rates is a key question that needs to be answered (Díaz et al. 2009). One of the key aspects of forests, biodiversity, carbon sequestration, and other ecosystem services that is often overlooked is time. There is such a need for information and research that studies are often completed in a time frame that does not accurately capture a system and its complexities. Ecological relationships take time to develop and gain consistency, but many studies are done in a span of four or less years. Long-term studies are needed to fully understand an ecosystem and the services it can potentially provide.

Time and Legacy Effects

Legacy effects are one aspect of time that cannot be overlooked in restorations attempts. In some cases, they may only last a few years, but other types of land use change or disturbances will have effects for decades or the ecosystem may never reach a pre-disturbance state (Allan 2004). Consequently, some ecosystem services require a longer recovery time depending on the ecoregion. Even if achieving a pre-disturbance state is not feasible, many ecosystem services can be restored to a high level of function. A study was performed that compared restored ecosystems and their ability to produce services to those of degraded and reference (undisturbed) systems. They found that 10 years after restoration efforts, the ratio of ES values in restored to reference forests and wetlands could be approximately 90% (Dodds et al. 2008). Nutrient cycling alone could be restored to 72% of reference systems. A meta-analysis of 89 studies world-wide

and ranging in time scales of less than 5 years to 300 years found similar results. Using the MA classification, supporting and regulating ES were higher in restored systems than in degraded systems but still lower than in the reference systems. Combined ecosystem services of restored systems reached 80% of the reference systems' values (Benayas et al. 2009).

This type of success is only possible when the structure of the ecosystem is restored. Restoration is essentially a way of altering or accelerating succession of a system, but it takes time to solidify the structure of the ecosystem before services can resume. As succession progresses in forests, the structure of the forest changes and the availability of certain ES changes and fluctuates (Dodds et al. 2008). The time it takes for a system to reach a stage where it begins providing more ecosystem services depends on a number of factors.

Generation time of restored organisms, and the environment's ability to establish new species are factors that need to be considered when monitoring for ES after restoration efforts (Dodds et al. 2008). With long life spans, trees take longer to reach an age where they can perform certain services. A forest takes decades to reach a stage of equilibrium, and ES will fluctuate during the different stages of succession. Similarly, the more degraded the environment is, or the more established invasive species are, the more time it will take to reestablish a functional forest that provides ecosystem services. The best option is to prevent ecosystem degradation. Maintaining healthy ecosystems and their services is more cost effective than allowing it to degrade and then spending money to restore it in an attempt to regain those services. In the case of floodplain forests, it is better to prevent the premature loss of mature trees and encourage recruitment of seedlings than to allow deforestation and subsequent colonization by invasives that are difficult and expensive to remove.

Monitoring Scales

Scale is another factor that needs consideration when discussing ecosystem services, restoration, and land management planning. Monitoring is often implemented during restoration projects to determine if efforts are producing the desired results, but the monitoring scale can affect the results. There are often three scales that are considered when monitoring water quality: reach, riparian, and catchment. Reach scale is typically a land buffer of 100 to several hundred meters

from the streambank and usually several kilometers in length. The riparian scale includes the reach, but usually continues for the entire length of the stream. A catchment includes the riparian and the entire watershed that feeds into the stream (Allan 2004).

Nitrogen fixation may be best monitored on a small scale of $< 1\text{km}^2$ while full nutrient reduction and uptake is better studied at the catchment-level. Temperature and dissolved oxygen are examples of small scale variables where local changes will have a large effect. Variables like sediments and nutrients can be transported long distances and can be monitored at multiple levels, but variables like hydrology, total nutrient loading, and infiltration are best done at a catchment scale (Allan 2004). When considering issues like flood protection, river flow, groundwater, erosion, and sedimentation; catchment, landscape, and even biome-level may be appropriate to gain an accurate understanding of effects of land use changes. ES like carbon sequestration and climate regulation are global, and it is difficult to fully understand them at other levels (De Groot et al. 2010).

There are many factors that influence ES and their evaluation; market values, diversity, time, and scale. Biodiversity has been shown to affect the availability and reliability of ES, but the relationship between the two requires more in depth and long-term study. Forests in particular have many more services to offer than lumber and recreation. Carbon sequestration, improvement of water and soil quality, and preservation of the terrestrial water cycle are only a few of the possibilities. More research is needed to determine how forest structure, diversity, age, and size affect these and other ecosystem services.

Major Findings

- Floodplain forest cover in the UMRS is dramatically reduced from historical levels due to basin-wide changes in land use and development. This trend is most pronounced in southerly reaches where up to 70-80% of the Mississippi River floodplain is protected by levees and the dominant land cover is agriculture. Altered hydrology and additional ecological stressors have led to further declines in forest health and diversity in many areas, and induced shifts in forest composition towards more flood tolerant tree species.
- In recognition of this trend, multiple stakeholders including federal and state agencies have initiated a variety of floodplain forest restoration efforts involving a spectrum of activities ranging from simple tree plantings to more intensive site-level topographic diversity enhancements. However, restoration efforts are often met with limited success and low survival rates among tree plantings. This is often attributable to discrete flood events or other disturbances; an incomplete understanding of how the ecological tolerances of tree species have impacted micro-elevational shifts and other vegetation community responses to changes in baseline hydrology, inundation frequency, and alluvial soil complexes throughout the floodplain; competition from invasive species; or in many cases a combination of the above factors.
- Reed canary grass (RCG) is one of the most damaging invasive plant species in the UMRS, especially in the northerly river reaches. RCG has high light requirements, establishes itself quickly in floodplain forest openings and edges, forms dense monocultures, prevents germination and out-competes tree seedlings. This has resulted in a gradual thinning of the forest canopy, loss of bottomland forest, reduced biodiversity, and an increase in the size and extent of RCG meadows in many floodplain areas throughout the upper reaches.
- The potential impacts of climate change on UMRS floodplain forests are not yet well known, but changes in temperature and precipitation patterns, longer growing seasons, higher atmospheric CO₂ levels, and increased flood and disturbance frequencies all have the potential to lead to increased ecological stresses and impact biodiversity in Mississippi River floodplain ecosystems.
- High biodiversity in an ecosystem helps to stabilize the system and maintain its services. Through time, species have complementary effects on each other to support different

ecosystem functions. High biodiversity of plants and their associated microbial communities is positively associated with reducing soil erosion and promoting nutrient cycling. Floodplain forests not only occupy wet habitats, but they are drivers of the terrestrial water cycle. Using their deep-reaching roots, they maintain evapotranspiration rates to recharge precipitation even in drought conditions. In the Mississippi River Basin, 58% of the precipitation may be generated by this evapotranspiration. Variables like canopy closure, age structure, and canopy roughness are all factors that influence a forest's impact on the water cycle.

- Restoration efforts can be highly successful in reestablishing system functions and services, but the goal should be to prevent ecosystem degradation and invasion by non-native species. It is more cost effective to protect and preserve existing ecosystem services than to restore a degraded ecosystem. Although forests and wetlands may take years or decades to become stable ecosystems, their functions can be restored. With sufficient effort and monitoring, function and services could be renewed to approximately 90% of undisturbed systems in as little as 10 years.
- Past research has found a significant difference between how nutrients are removed by grass vs. forested buffer strips. Forested buffer strips have been found to be more efficient at filtering out nitrates when compared to crops or grasses. Higher nitrate uptake rates for lowland forested buffer strips has been measured in groundwater as well as surface runoff in headwater stream systems.

Research Recommendations

- Development of a system-wide, georeferenced, data-driven model clarifying the functional relationships, tolerance ranges, and interactions between vegetation communities (and individual tree species) and hydrological regimes, micro-elevation, and soil properties in the Upper Mississippi River System (UMRS). This effort would in large part build upon localized studies that have already been completed and/or ongoing local-level floodplain forest restoration and adaptive management projects.
- Development of a system-wide GIS-driven effort to map RCG dominated areas in the UMRS and identify areas that are likely to transition from floodplain forest cover to RCG based on underlying forest community characteristics and dynamics (i.e., age and structure profiles), elevation, and/or soil properties.
- Additional research into the role of UMRS floodplain forests and vegetation communities on nutrient and carbon sequestration and fluxes, including the impact of widespread invasive species like RCG on river-floodplain nutrient dynamics, is highly recommended. Again, there have been a few localized studies, but a comprehensive assessment would provide much-needed information on large river nutrient loads and dynamics at basin-level scales relevant to issues such as Gulf Coast hypoxia.
- Development of additional site-level experimental research projects in the UMRS focused on topics related to: improving guidelines for the implementation of specific floodplain forest restoration methods and techniques; clarifying interactions between floodplain forests and invasive species, and the effects of those interactions on vegetation dynamics and biogeophysical processes; and nutrient and carbon fluxes and sequestration, including their links to vegetation structure and water quality at multiple scales.
- Few studies have investigated removal capabilities with regards to chemical herbicide and pesticides for woody perennials. While more research has been done on uptake and removal of these chemicals for grasses and other annual crops, additional investigation is needed on the ability of various floodplain forest tree species in the UMRS to improve water quality through phytoremediation.

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Appendix A

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Appendix B

Reference Management: EndNote

EndNote is a reference manager software program produced by Thomson Reuters. The program allows the user to collect and organize references and to create bibliographies. EndNote can be used with Microsoft Word or other word-processors to create bibliographies through an automatized process. EndNote has the ability to format bibliographies in any citation style (e.g. MLA, APA, Vancouver, Chicago, etc.), and can reformat bibliographies after they have already been created within a document. Over 2,000 bibliographic formats exist in the EndNote program.

Users can create individual “libraries” containing their citations. These libraries group references together, and can be shared among users. When collaborating with other users, library files can be transferred (e.g. email, USB, etc.) directly. Libraries have the file extension *.enl . All *.enl files also have a *.data folder which corresponds to the library. These two folders must be transferred together at all times to avoid file corruption and accessibility issues. Another way to share libraries is through the “Share Library” feature in EndNote X7. This feature allows a user to create a library and share it with other users who have registered accounts with Thomson Reuters. When EndNote is connected to the internet, collaborating users using this feature can add new citations and sync those changes so that the shared library is updated in real-time. References can be accessed through a word processor while writing a document. When the EndNote software is installed on a computer, a feature called Cite While You Write™ will appear in the Tools menu of Microsoft Word. This feature allows users to easily insert references into the body of a manuscript, and the bibliography is automatically generated at the end of the document. Word files can be generated by the Cite While You Write™ feature as HTML, plain text, Rich Text Format, or XML.

References can be added to a library through a number of ways. One manner is to create a new reference in the program and manually enter the information. A second is by exporting references from online databases such as Web of Science, PubMed, Google Scholar, etc. into

EndNote. A third way of adding a reference to a user's library is by searching for and importing articles from library catalogs and free databases within the EndNote program itself. Other methods of adding references exist, and all references added to the library can include a PDF of the article in addition to the general reference information (e.g. author, ISBN, publication date, etc.). EndNote provides an extremely user friendly platform for organizing PDF files on the hard drive and for accessing those files while working in the program. Other files, such as images, word documents, spreadsheets, etc., can be attached to each reference.

The "Works Cited" section above includes only references that have been used in the body of this report. Throughout the development of this literature review, however, many additional articles other than those cited above were found to be relevant to the link between floodplain forests and water quality. For this reason, a "Full Bibliography" section has been included as Appendix A above to provide readers supplementary resources. The EndNote library files for the cited references or for all references are available upon request.