Ecological Implications and Science-Based Strategies for Invasive Carp in Minnesota

Luther Aadland, Amy Childers, Ian Chisholm

Stream Habitat Program, Division of Ecological and Water Resources, Minnesota Department of Natural Resources

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Executive Summary

Statement of the Issue

Bigheaded carps (silver and bighead carp) were introduced to the U.S. over 40 years ago and have since become abundant in the Mississippi, Missouri, and Illinois rivers. As these fish species spread, it is feared that they will become a dominant part of the fish community and adversely affect or displace native species. In response to potential ecologic and economic effects of spreading bigheaded carp populations, state and federal agencies have responded with plans and strategies for addressing bigheaded carp dispersal, including construction of dams and electric barriers. However, fragmentation of river systems by barriers has been shown to be among the most definitive causes of extinction and extirpation of native species.

To address this management issue we examine four underlying questions:

- Why are some introduced species successful while many native species decline?
- What are the effects of carps on native species and their populations?
- How effective are barriers for introduced carp?
- What are the effects of barriers on native fish species in Minnesota?

Based on scientific review and empirical findings within these four fundamental questions, alternative strategies are proposed. These strategies address invasive carp while prioritizing native species and the river ecosystems upon which they depend.

Question 1. Why are some Introduced Species Successful while many Native Species Decline?

Native species have adaptive traits that favor their survival under prevalent historical conditions. Their life history cycles take advantage of river networks that provide corridors for migrations to allow access to a range of habitats. Their adaptation to connected river ecosystems allows them to recolonize after droughts, floods, and other disturbances. Many native species are vulnerable to poor water quality, including low dissolved oxygen, and depend on the ability to migrate in response to these impairments.

Eutrophication, habitat degradation, hydrologic changes, impoundment and fragmentation have altered and degraded stream habitat and conditions resulting in depleted native stream biodiversity. Meanwhile these conditions favor tolerant native and introduced species.

For example, bigheaded carps (silver and bighead) have adaptations that include their ability to a) tolerate impaired water quality (while they have become widely distributed, they often reach highest densities in impoundments) and b) consume and digest blue-green algae (cyanobacteria) that are inedible or toxic to most native fishes, which provides an otherwise unexploited food resource to these fishes. These species have been selectively bred in Asia for over 1,000 years to tolerate conditions in hyper-eutrophic ponds. River and lake systems with elevated nutrient levels, particularly in agricultural watersheds, create eutrophic conditions favorable to these species.

Question 2. What are the Effects of Carps on Native Species and their Populations?

Even at extreme biomass levels, bigheaded carps have <u>not</u> been shown to be causative in the extirpation of native species. Correlated declines in abundance of some primarily planktivorous species on the Illinois River began prior to the arrival of bigheaded carps. Bigheaded carps were associated with changes in zooplankton composition, which appeared to result in a slight decline in condition factor of two planktivorous fish species in the Illinois River.

The effects of introduced carps are conditional, based on the degree of fragmentation and eutrophication. The case evidence suggests that as water quality deteriorates and fragmentation increases the risk of bigheaded carp success also increases. It is unclear whether this high risk of effects on native species is primarily due to the bigheaded carps or due to the degraded water quality and fragmentation.

Question 3. How Effective are Barriers for Introduced Carp?

Electric barriers have not been proven to be effective in limiting upstream dispersal or abundance of common carp or other introduced carps due to power outages, flood flows, alternate dispersal pathways, and other factors. Most dams have not proven to be effective barriers to introduced carp due to inundation by large floods or alternate dispersal pathways. The ability of silver carp to jump 10 feet and burst to over 20 feet per second makes them much more likely to pass barriers than most native species.

Barriers may actually increase the success of carp and other tolerant species (native and non-native) by reducing competition and predation controls by native species. Based on case examples in Minnesota and elsewhere, barrier effectiveness for limiting range expansion and abundance of introduced carps is predicted to be low with high certainty.

Question 4. What are the Effects of Barriers on Native Fish Species in Minnesota?

Barriers have been shown to be among the most definitive causes of loss of native species in Minnesota waters and globally. An assessment of 32 dams on streams throughout Minnesota found that barriers have a substantial negative effect on native biodiversity of fish and mussels. Species richness of native fish was an average of 41% lower upstream of nineteen complete barrier dams. When dams were removed or failed, an average of 68% of the absent fish species returned to the upstream watershed. In addition, three extirpated mussel species recolonized the Pomme de Terre River following removal of the Appleton Dam.

Loss of biodiversity due to barriers was also shown to extend to entire watersheds. This loss in biodiversity can adversely affect (a) water quality through loss of filtration by mussels, (b) the bait industry through loss of shiners and other bait species, (c) recreation by loss of fisheries, and (d) overall watershed health. Based on this assessment, effects of new barriers on native biodiversity are predicted to be high with high certainty.

Documented adverse effects of barriers on native species are far greater than documented negative effects of bigheaded carps on native species. **Therefore, fish barriers should not be considered as a viable alternative on naturally free-flowing rivers.**

Management Implications and Recommendations

1. Alternative strategies for addressing a biomass dominated by bigheaded carps include:
   (a) improvements in water quality, dam removal, and restoration of natural habitat to increase native species abundance and diversity to increase resistance to invasion,
   (b) increased protection of flathead catfish and other predators of bigheaded carps, and
   (c) commercial harvest of bigheaded carps.

   These strategies can be used in combination with other strategies that specifically target carp without adversely affecting native species.
2. Barriers that re-establish either watershed divides in artificially connected watersheds (ditches and canals) or natural barriers would not adversely affect native biodiversity. Re-establishing natural barriers, such as Upper St. Anthony Falls, is an example of reestablishment of a natural barrier.
3. Based on the systematic review of current science within this document for each of the questions outlined, the conditions on the Minnesota River, and the dramatic need for maintaining native species everywhere - we recommend a focus on maintaining and restoring healthy watershed conditions for native species. This will not be easy or quick. But effective prevention and control of biotic invasions requires a long-term, large scale strategy, rather than a tactical approach that focuses on battling individual introduced species. Ultimately, to be successful, an effort will have to be made to include stakeholders, engage them fully in the science and ecology of these systems, and work together towards long-term solutions that emphasize healthy systems and native species.

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Introduction

Statement of DNR's Goal

Connected landscapes, including river networks, are critical to the health and sustainability of native plants and animals. A primary goal of the agency and its management is to protect the health of native species and their populations. In general, we support restoring connectivity. At the same time, we are fully exploring all control and management options, including the construction or removal of barriers, to address the invasive carp issue, on a case by case basis.

Statement of the Issue

Bigheaded carps (silver and bighead carp) were introduced to the U.S. over 40 years ago and have since become abundant in the Mississippi, Missouri, and Illinois rivers. As these fish species spread, it is feared that they will become a dominant part of the fish community and irreparably alter existing ecosystems. To address potential ecologic and economic effects of spreading bigheaded carp populations, state and federal agencies have responded with plans and strategies for addressing bigheaded carps invasions, including construction of dams and electric barriers.

Social aspects of this issue cannot be ignored. Numerous regional and local newspaper articles have been written about introduced carps, generally sounding the alarm and calling for immediate action (e.g., Anderson, Close locks now, before Asian carp infestation grows, Star Tribune 8/7/2013; Asian carp still a problem in Illinois waterways, Chicago Tribune.,7/28/2013; Big or small, Asian carp mean trouble for South Dakota waters, Rapid City Journal, 7/18/2013). Groups have formed around the perceived threat these introduced species pose to Minnesota species, waterways, recreationists, and economies (e.g., Stop Carp Coalition - a group whose 19 members are comprised of well-recognized conservation organizations, including Audubon Minnesota, National Izaak Walton League, National Wildlife Federation, Minnesota Center for Environmental Advocacy, Conservation Minnesota and others). Websites have been created around the introduced carps issue, to inform and advocate for active control and management of introduced carps (e.g., www.asiancarp.us/index.htm; http://stopcarp.org/; www.greatlakesunited.org/en/asiancarp).

At the outset, it is important to recognize that underlying the concern over introduced species, such as the bigheaded carps, is the fear of what they will do to the native species, the broader ecosystems, and the services that the ecosystems provide. As a result, there are two distinct aspects of this issue that cannot be avoided and need to be addressed explicitly in management actions:

(1) how can we restore and protect the native species and

(2) what can we do to slow or stop the introduced species that will not adversely affect the native species?

Previous Planning Documents

As noted above, plans have been drafted by various organizations to address introduced carps movement throughout the U.S. and into Minnesota waters. In 2002, the Aquatic Nuisance Species Task

Force requested the USFWS develop a national management and control plan for introduced carps. This effort produced a document drafted to comprehensively address introduced carp issues, entitled, The Management and Control Plan for Bighead, Black, Grass and Silver Carps in the United States (Conover et al. 2007). It had seven goals and sections: (1) prevent introductions, (2) contain and control expansion, (3) extirpate feral populations, (4) minimize potential adverse effects of invasive carps, (5) provide information to the public to improve effective management, (6) conduct research to improve the science of management, and (7) nationally coordinate implementation efforts. With the exception of goal 4, the emphasis of the Plan is almost entirely focused on the introduced carps, with comparatively little explicit consideration of native species.

A fundamental consideration of any control and management effort should be to understand the impacts of our techniques and strategies on the environment and on non-targeted species (e.g., native fish and mussel species). This introduces a complexity, which calls for immediate response. A summary statement within the Conover et al. (2007) Management and Control Plan was that, “Implementation of the plan should begin immediately to prevent further introduction and stopping the spread of Asian carps into uninvaded waters throughout the United States.” In 2009 Minnesota’s Management Plan for Invasive Species was developed, which was endorsed by over a dozen major state organizations (Minnesota Invasive Species Advisory Council, 2009), and addresses all invasives generally. In this plan a call to manage the pathways of introduction and spread of invasives was advanced and 4 key elements were presented for accomplishment: (1) Prevention, (2) Early Detection, Rapid Response, and Containment, (3) Management of Invasive Species, and (4) Leadership and Coordination. The recommended actions and strategies outlined one instance of direct reference to native species and the role of intact ecosystems as important elements in the Management Plan for Invasives (Element III, Item 8b).

While these documents are plans for action to attempt control of introduced species, thorough examination of the inter-related goals and objectives forwarded in the plans and their potentially negative impacts on native species are essentially not addressed or addressed only lightly. Kolar et al. (2005) deal with the environmental effects of bighead and silver carps, discussing their impacts under five broad categories: habitat alteration, trophic alteration, spatial alteration, gene pool deterioration, and disease transmission. Because the impact of introduced species control and management is central to the health of native species and their populations in Minnesota, there is a need to re-examine the concern over introduced carp explicitly in light of the impacts to native species and their populations.

Report Approach

A fundamental part of the introduced species management process, as outlined in this Report, includes examination and consideration of: (1) the impact of the introduced species on native species and their populations, (2) the actual or probable efficacy of the barrier for preventing spread of the introduced species, and (3) the impacts of barriers on native species. Implications of these elements on recreation are inherent due to the importance of native species to recreational activities while boater collisions with jumping fish are related to silver carp densities. Another piece of the management strategy should include documenting estimates of risks and uncertainties. Together, these elements provide a basis for engaging the issue in an adaptive management framework, where we learn by doing, and avoid undoing our fundamental goal – protecting native species, their populations, and the related recreation and economies.

Report Objective

The objective of this Report is to provide reliable scientific information and synthesis of the specific underlying questions involved with the bigheaded carps issue and subsequent management actions. In particular, we examine the issue of bigheaded carps invasion with respect to the native species and health of existing ecosystems. The purpose of this document is to help guide decision making and prevent avoidable adverse impacts on native fishes and associated impacts on recreationists and our economy.

Question 1. Why are some Introduced Species Successful while many Native Species Decline?

Overview: Natural selection continually favors fitness traits suited to existing environmental conditions and dynamics. When environmental conditions change, a new set of traits favor survival in the new conditions. This can lead to a shift in species composition towards those that are most fit in the altered habitat.

Changes to river systems and their watersheds have greatly altered the quality and availability of aquatic habitat across the state and globally. More specifically, the current health of watersheds in the Minnesota River basin, as it pertains to the spread of bigheaded carps (silver and bighead), provides insight into the potential of their establishment in those waters. Bigheaded carps possess adaptive traits, such as the ability to consume cyanobacteria and exist in low oxygen, polluted waters, which favor their survival in degraded, fragmented, and impounded systems.

The success of introduced species is also facilitated by the loss of competition and predation through the loss of native biodiversity. Extinctions and extirpations (local extinction) of native fish have been caused by numerous factors, including alteration of the landscape, fragmentation by dams, loss of habitat, and pollution. The loss of native biodiversity is a catastrophic problem globally and demands specific attention for sustainable management of aquatic systems.

Impairments to river systems that result in the extirpation of native fishes are the same factors that result in disproportionate abundance of bigheaded carps and other tolerant non-native species.

Ecological Definitions

Many of the concerns regarding invasive species are rooted in discrete definitions of native and non-native. Species assemblages are in a constant state of flux over time. Species invasion is a necessity of survival and these natural invasions constantly reshape biodiversity. This makes the term “nativeness” time-scale dependent. So the question becomes “At what point is a new species acknowledged as part of the ecosystem?” For example, brown trout and common carp, while purposely introduced by humans, have been naturalized in Minnesota for well over a century. In this respect, are they fundamentally different from other species that invaded our waters? As stated by Thompson (2014), “At some scale, all species are invaders and all ecosystems are novel.”

For example, all the streams and rivers in Minnesota were fishless until the end of the last ice age. They were recolonized, or invaded, by fish species primarily from southern waters of the Mississippi Basin once Minnesota streams became inhabitable. Considering the climate is currently warming at an unprecedented rate, we can assume species are or will be migrating in response. Minnesota will likely begin to see more southern species expanding their range, or invading, in a northerly direction. For example, the golden redhorse, native to the Red River in MN was first found in Manitoba, Canada in 1985 (Stewart and Watkinson 2004), so by some definitions this new species is an invasive, non-native species that expanded its range to Canada.

The terms native and alien were first coined by H.C. Watson in the mid-19th century. A species is native (indigenous) if its presence is the result of only natural processes, with no human intervention. Unfortunately it is virtually impossible to separate this given the degree to which humans have globally intervened or influenced the dispersal and success of species. While humans are increasing dispersal through intentional and unintentional introductions, we are impeding natural dispersal through fragmentation. For the purposes of this analysis, species that have been presumed to be present prior to European settlement are described as native to Minnesota even though it ignores that species assemblages are in a constant state of flux.

A species is introduced (non-native, alien, exotic, non-indigenous) if it is living outside its native range and has arrived there by human activity, either deliberate or accidental.

Defining a species as invasive can be misleading because all species were/are invasive at some point. Species naturally and continuously attempt to expand their range and invade new areas. Their success in new areas depends on their suitability to the new environment, ability to compete for resources or fill an niche, and ability to avoid predation.

Colonizing or pioneer species are capable of quickly inhabiting a disturbed ecosystem. Stream fish communities are in a recurrent state of recolonization following drought, seasonal low flows and extreme winter conditions. These events become more inhospitable in agricultural watersheds due to higher nutrient loading (low dissolved oxygen) and loss of wetland storage and channelization (accentuated low flows). In these watersheds, species that can tolerate low dissolved oxygen and have high reproductive potential have a competitive advantage since they aren't dependent on lengthy migrations and can quickly repopulate.

Tolerant species are those that can tolerate degraded water quality conditions, which can include low dissolved oxygen, high nutrient loads, increased turbidity, elevated temperatures, and chemical pollutants. Pollution *intolerant* or *sensitive* species are those that can not survive in one or more of these conditions.

Habitat specialists are those that require specific habitat characteristics for a portion of their life cycle. Often these habitats are relatively rare (i.e. lake sturgeon that spawn in rapids). *Habitat generalists* can exploit a wide variety of habitats and food sources (i.e. fathead minnows)

A distinguishing characteristic of most introduced species perceived as problems is their tendency to become "ecological dominants" or species whose biomass far exceeds that of other species. In some cases, non-native ecological dominants can decrease the diversity of other species through competition or predation. In contrast, keystone species can also be ecological dominants but tend to support and increase the biodiversity of other species. Freshwater mussels are examples of keystone species that can be dominant species while performing critical functions of water filtration, nutrient processing, stabilization of substrates, and increasing diversity of the benthic community.

Below are examples of fish exhibiting these characteristics:

- Bigheaded (silver and bighead) carps: *introduced, tolerant, ecological dominants in nutrient-rich plankton-rich systems*
- Common carp: *introduced, tolerant, generalists, ecological dominants in degraded systems*
- Channel catfish: *native, keystone species (top predator and host to at least 13 mussel species)*
- *Greater redhorse: native, sensitive, specialized*
- Mussels: *native, keystone species, historical ecological dominants (many species are now extirpated or extinct)*
- Black bullhead: *native, tolerant, ecological dominants in degraded systems*

The Extent of Introduced Species

Introduced species have expanded globally. According to Meinesz (2003), roughly 7,000 species have been introduced globally, of which about 15% have caused ecological or economic damage. In most cases, the introduced species viewed as damaging become very abundant and dominant biomass in certain habitats.

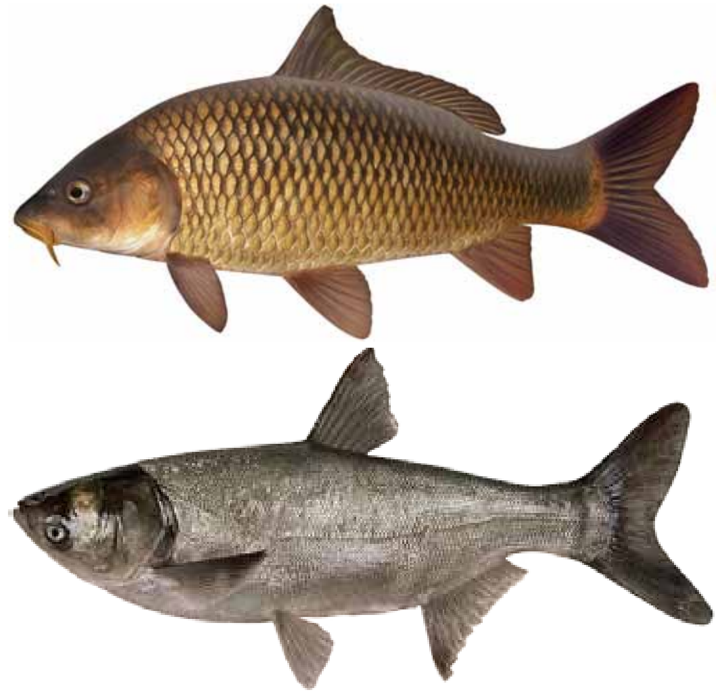
Fishes are among the most introduced group of aquatic animal in the world and also one of the most threatened.

Fishes are among the most introduced group of aquatic animal in the world (i.e. 624 species, Gozlan 2008). Fish species are introduced around the world because of societal demands for fish products for food aquaculture (51%), ornamental fish (21%), sport fishing (12%) and fisheries (7%) (Gozlan 2008). Socioeconomic forces suggest that the increasing trend of non-native fish introductions will continue. There are at least 15 introduced aquatic animal species (8 of which are fish species) and 10 aquatic plant species identified as 'invasive' by the MN DNR as of 2013. There are additional non-native fish species such as brown and rainbow trout that have been introduced for fisheries management in Minnesota waters. Popular game species such as smallmouth bass, native to some Minnesota waters, have also been introduced to waters to which they were not native.

There are nine species of heavy bodied cyprinids, all originating from Asia, that have been widely introduced outside of their range. These species have been poly-cultured in fertilized ponds and selectively bred to tolerate eutrophic conditions in China for at least a thousand years and include:

- common carp *Cyprinus carpio*,
- grass carp *Ctenopharyngodon idella*,
- silver carp *Hypophthalmichthys molitrix*,
- bighead carp *Hypophthalmichthys nobilis*,
- largescale silver carp *Hypophthalmichthys harmandi*,
- black carp *Mylopharyngodon piceus*,
- common goldfish *Carassius auratus*,
- crucian carp *Carassius carassius*, and
- mud carp *Cirrhinus molitorella*.

Of the nine widely introduced carp species, all native to Asia, three species were intentionally introduced to Minnesota while the bighead and silver carps have recently reached Minnesota waters via the Mississippi River.



(top) A common carp, Minnesota's most widely distributed and abundant introduced species. *Credit Pat Tully.* (bottom) A silver carp. *Credit Fish Market Development Association.*

Of these, common carp, goldfish, grass carp and more recently, silver carp and bighead carp have been documented in portions of Minnesota. Goldfish were introduced in Minnesota at least 70 years ago and have been caught in several lakes in the Minneapolis - St. Paul area (Eddy and Surber 1947) but have remained rare. Common carp have been well established in Minnesota for 120 years and dominate many eutrophic lakes. Silver and bighead carp, known collectively as "bigheaded carps" will be the primary focus of this paper because they are dominant species in parts of the Upper Mississippi basin and have the potential to have impacts in Minnesota. However, the broader ecological context of native and introduced species warrants discussion.

Ecological Context: Why are some introduced species abundant while many native species decline?

Increasing abundance of tolerant introduced species has trended concurrently with habitat alteration and the decline of many sensitive native species. This suggests that these trends are related. As posed by Thompson (2014) "Are exotic species really the main cause of decline, or are they just filling in the gaps left by pollution, climate change, and habitat degradation?" Accurately diagnosing the underlying causes of these trends is critical to reaching effective solutions and implementing effective management strategies.

Accurately diagnosing the underlying causes of the decline of many native species is critical to reaching effective solutions and implementing effective management strategies.

Factors that lead to the disproportionate abundance of a fish species include:

1) adaptation to the available physical habitat and food resources for the suite of their life history stages.
2) ability to tolerate associated physical, thermal, and chemical disturbances associated with current environmental conditions. When these conditions change the competitive success of species reflects the degree of adaptation to the new environment.
3) little or no competition for food or adaptations to exploit an open niche,
4) ability to develop predator avoidance and competition strategies. Additionally, the simple lack of predators or decreases in their abundance through habitat changes, and
5) high fecundity and sexual maturity at young age.

When species with these adaptations are introduced to an altered environment with degradation of habitat and the resulting loss of the loss of native biodiversity (potential competitors and predators) - an environment ripe for successful invasion is created.

Ecological dominance of introduced species has often coincided with habitat degradation, water quality declines, fragmentation, and associated declines of native fauna.

Adaptive Traits

The success of a species is a function of its traits and adaptations to the ecosystem (Darwin 1859). Introduced species that are successful typically are colonizers with high reproductive potential and are often tolerant of harsh environmental conditions. When ecosystems change, it is intuitive that the relative success of a species may also change and introduced species may have competitive advantages in an altered system. Ecological dominance of introduced species has often coincided with habitat degradation, water quality declines, fragmentation, and associated declines of native fauna (Stachowicz et al. 2002; MacDougall and Turkington 2005; Carey and Wahl 2010). Tolerant species that have generalized habitat requirements are the most likely to have a competitive advantage in degraded, fragmented systems.

Bigheaded carps have the following adaptations to be successful in degraded, fragmented streams and rivers.

(top) Common carp in the Minnesota River. (bottom) Silver carp in the Platte River near confluence with Missouri River. *Credit DNR SHP.*

In unaltered, free flowing rivers native species assemblages thrive because...

- they are survivors of centuries of natural selection.
- habitat is diverse providing a diversity of niches.
- Introduced species may become established but face ongoing competition and predation resistance.
- sensitive species can recolonize following winter, drought, and other natural disturbance events.

In degraded rivers that are fragmented and impounded, species that are generalized, tolerant, and reservoir-adapted (native or introduced) can become disproportionately abundant because...

- the altered habitat is favorable to these tolerant species.
- most native species have declined since they are not adapted to altered and homogenous habitat reducing competition and predation controls.
- tolerant species can survive the degraded condition making them less dependent on recolonization.
- sensitive native species can not recolonize due to barriers.

The ability to digest cyanobacteria (which is toxic to most natvie fish) may enable them to attain high biomass by utilizing an otherwise unexploited resource.

What They Eat

Bighead and silver carp are tolerant planktivores, which means they filter plankton (primarily phytoplankton, but also zooplankton) out of the water column. It has been shown that their consumption of zooplankton can affect zooplankton composition and densities , producing community shifts from larger bodied cladocerans and copepods to small bodied rotifers in the Illinois River (Cooke et al. 2009; Garvey et al. 2012). In contrast, Zhang et al (2013) found an increase in zooplankton size associated with bigheaded carps consumption of cyanobacteria. These differences in effect of bigheaded carps on plankton composition are likely due to complex interactions.

Bigheaded carps also have the unique ability to digest cyanobacteria (blue-green algae) and may even prefer it (Chiang 1971). Cyanobacteria thrive in nutrient enriched, low velocity waters, including lakes and impounded rivers. Few native fish can digest cyanobacteria and some species of *Microcystis*, a freshwater cyanobacteria, can be toxic enough to be a significant mortality factor in fish populations (Ernst 2008; Lewin et al. 2003). This may enable them to attain high biomass by utilizing an otherwise unexploited resource. Several studies have shown reductions in cyanobacteria blooms in lakes and ponds due to consumption by silver and bighead carp (Opuszynski and Shireman 1993; Xie and Liu 2001; Leventer and Telsch 1990). Bighead carp tend to consume more zooplankton than silver carp, but gut contents of both species are often dominated by algae, including blue-green algae (Opuszynski and Shireman 1993; Xie and Liu 2001; Williamson and Garvey 2005; Cooke et al. 2009).

Common carp are omnivores able to consume a wide range of plant and animal materials, including algae, grasses, terrestrial and aquatic insects (especially tolerant species), and small fish (Becker 1983).

Tolerant

Silver and common carp are designated as tolerant species by the US EPA (Barbour et al. 1999). Bigheaded and common carp have been cultured in fertilized ponds for over a thousand years (Rabanal 1988). As a result, these domesticated species are, in part, a product of selective breeding for hypereutrophic waters.

Bigheaded carps are tolerant of low oxygen and poor water quality, which is demonstrated by the fact that these fish can thrive in sewage lagoons (FAO 1984; Sin and Chiu 1987). Juvenile silver carp have been found to grow a vascularized extension of their lower jaw in response to low dissolved oxygen enabling them to use oxygen at the surface interface in anoxic waters (Amberg et al. 2012). The ability to exist in anoxic waters can also provide juveniles with a refuge from most predators (Duane Chapman, USGS, personal communications 2013).

Similarly, common carp are tolerant of low dissolved oxygen and impaired water quality as demonstrated by the fact they too can thrive in sewage lagoons (Minnesota PCA; Sin and Chiu 1987; Azim and Wahab 2003). Their physostomous swim bladders enable them to gulp atmospheric oxygen when dissolved oxygen is low (Mackay 1963). Butler and Wahl (2010) found that individual common carp spent most of their time in impounded reaches of the Fox River in Illinois and that they were the dominate species in catches from the impoundments where low dissolved oxygen and other water impairments were frequently documented.

Reproductive Advantages

It was previously thought that bigheaded carps needed many miles of free flowing large rivers to keep the eggs suspended; however, recent evidence suggests that they successfully hatch in impounded reaches. Silver carp eggs have successfully developed in river lengths of only 15 miles (Murphy and Jackson 2013) and in reservoirs with small watersheds (Tang 1960). The Illinois River, where silver carp densities are believed to be the highest in the world (Sass et al. 2010), is entirely impounded by dams. Otolith analyses by Garvey et al. (2012) determined 72% of sampled adult silver carp were spawned within the Illinois River. These recent findings show that these fish are not limited to spawning in large free flowing rivers and that they can in fact successfully spawn in impoundments. During floods many dams of the Illinois and Mississippi rivers are operated in open gate condition, which may facilitate the suspension and development of bigheaded carp eggs.

After hatching, fry drift into and as they develop they seek out shallow, low-velocity near-shore areas and backwaters (Deters et. al 2013). Some of these backwaters become anoxic with high water temperatures. These areas provide refuge from predators where the fry can grow rapidly. The fry and juvenile bigheaded carps have an adaptive advantage in these backwaters since most predators can't survive in the low dissolved oxygen environment. When dams impound rivers, such as the Illinois, these backwaters areas are a stable refuge because they are maintained at a relatively constant water level. Furthermore, flow regulation can stabilize water levels in the channel and connected backwaters, thereby favoring bigheaded carps, at the same time, eliminating natural flow regimes to which native fishes have evolved. Conversely, the backwaters on free-flowing rivers are more dynamic as they can dry up and disconnect as river flows decrease.

Adaptations to Impoundment

When rivers are dammed their ecology is altered in many ways. In free flowing rivers, floating organisms are swept away continually so the population of plankton remains low, whereas the benthic invertebrate population may be high (Baxter 1977). In contrast, due to higher retention times, impounded rivers favor plankton production while riverine benthic invertebrates decline and are often displaced by reservoir tolerant chironomids. Algal concentrations of the Upper Mississippi River increased up to 40 times that of the 1920s following construction of the locks and dams (Baker and Baker 1981). Base on observations by commercial fishermen, impoundment of the Upper Mississippi resulted in an increase in the abundance of common carp and planktivorous buffalo (Ictiobus spp.) and a decline in benthic shovelnose sturgeon (UMRCC 1946). The increases in plankton production, and specifically cyanobacteria, as already discussed, logically favor planktivorous bigheaded carps in impounded rivers.

Life history characteristics of silver, bighead, and common carp are summarized in Table 1.

Table 1. Life history characteristics of Bigheaded and Common Carp

Characteristics	Silver carp	Bighead carp	Common carp
			
Life expectancy	15-20 years	Max age 25 yrs	9-15 years Max 47 yrs
Average size reported in Garvey et al. 2012	3.5 lbs	12.3 lbs	
Record size (International Game Fish Association)	70 lbs	90 lbs Missouri record: 106 lbs (MO DOC) Iowa record: 93 lbs (IA DNR)	75 lbs MN record: 55 lbs (MN DNR)
Jump height	10 feet	*assumed to be comparable to silver carp*	3 feet
Burst speed	>25 feet per second based on observed jump height	*assumed to be comparable to silver carp*	up to 14 feet per second
Where they eat	Water column	Water column	Primarily benthic
What the adults eat	Tolerant planktivore, including blue-green algae, feed continuously because lack true stomach	Tolerant planktivore, including blue-green algae, feed continuously because lack true stomach	Tolerant omnivore
Preferred habitat	Low velocity water except for spawning	Low velocity water except for spawning	Low velocity water during all life stages
Sexual maturity (likely older in northern latitudes)	Females: 3-4 yrs Males: 2 yrs	Females: 3 yrs Males: 2 yrs	Females: 3 yrs Males: 2 yrs
Spawning	Pelagic, large rivers during high spring flow. Eggs require current to keep off bottom. Floodplains provide nursery areas for larvae & juvenile forms	Pelagic, large rivers during high spring flows. Eggs require current to keep off bottom. Floodplains provide nursery areas for larvae & juvenile forms	Spawn on vegetation or debris in lakes, bays, floodplains, backwaters or wetlands
Fecundity	50,000 to 5 million eggs (depends on size & age)	478,000 - 3 million eggs (depends on size & age)	56,400 – 2.2 million eggs (depends on size)
Extent in Minnesota	In Mississippi R. reproducing populations as far north as Dubuque IA with a few occurrences in Minnesota	In Mississippi R. reproducing populations as far north as Dubuque IA with a few occurrences in Minnesota	Were introduced throughout MN, widely established populations
Adaptive traits: Food	Can eat cyanobacteria	Can eat cyanobacteria	Onmivore, including tolerant invertebrates
Adaptive traits: Reproductive Advantages	Juveniles can survive in anoxic backwaters.	Juveniles can survive in anoxic backwaters.	
Adaptive traits: Tolerant of low DO	Juveniles develop vascularized lower jaw extension enabling use of atmospheric O_2	Juveniles develop vascularized lower jaw extension enabling use of atmospheric O_2	Can gulp atmospheric O_2
Issues & Concerns	Jump out of water when disturbed; prefer eutrophic waters; alter plankton composition; compromise commercial fishing.	Prefer eutrophic waters; alter plankton composition and food web interactions; compromise commercial fishing.	Prefer eutrophic waters; affect water quality by uprooting vegetation and stirring lake bed sediments.

Lack of Competition and Predation due to Loss of Native Biodiversity

Loss of Aquatic Biodiversity

Loss of biodiversity has been considered the single most significant environmental issue facing humanity. In an interdisciplinary paper authored by 29 international experts, loss of biodiversity was considered to be the most severely exceeded planetary boundary, followed by climate change and alterations to the nitrogen cycle (Figure 1) (Rockström et al. 2009). Current extinction rates are estimated to be 100 to 1,000 times faster than those indicated by the fossil record, which provides a background rate.

Loss of biodiversity has been considered the single most significant environmental issue facing humanity.

The extinction rate for North American freshwater species is estimated to be 5 times the rate of terrestrial species, 877 times that of background rates, which is comparable to that for tropical rainforests (Ricciardi and Rasmussen 1999). Freshwater mussels are among the most endangered group of organisms on the planet with 71.7% of North American species considered imperiled (Williams et al. 1993). Fish are one of the most threatened groups of animals, with the total number of threatened fish species reaching 1,201 in the year 2007 (IUCN 2008) Of North American freshwater fishes, 39% are considered imperiled (Jelks et al. 2008). Sturgeons are the most imperiled group of species overall with 85% of the 27 species at risk of extinction, most of which are critically endangered (IUCN 2010). These rates are projected to continue to increase due to the large number of imperiled species and increasing negative effects of human activities on the Earth's biosphere (Burkhead 2012). This is characteristic of many other ecological issues where the needs of societal development do not necessarily converge with conservation interests (Gozlan & Newton 2009 and Gozlan et al. 2010).

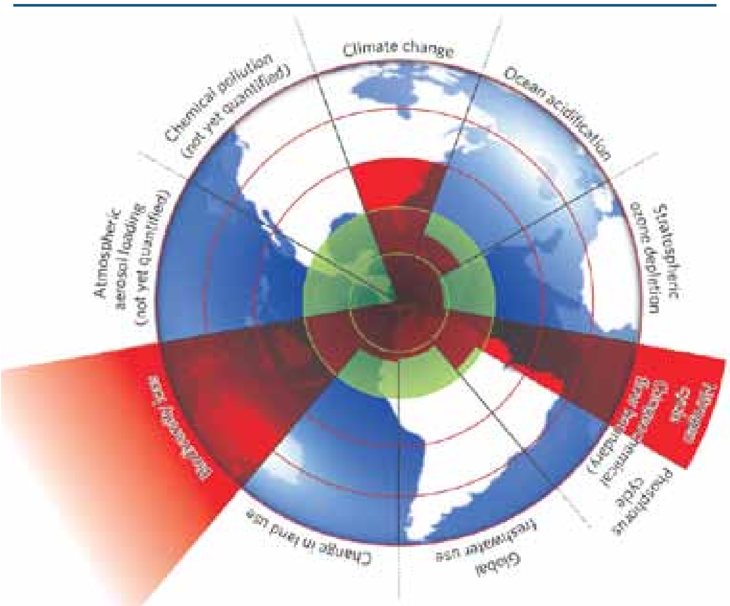


Figure 1. Planetary Boundaries: the nine wedges represent an estimate of the current position of each boundary. The inner green shading represents the proposed safe operating space. Biodiversity loss, nitrogen cycle, and climate change have been transgressed (*from Rockström et al. 2009*).

The extinction rate for North American freshwater species is estimated to be 5 times the rate of terrestrial species, 877 times that of background rates which is comparable to that for tropical rainforests.

While extinction is a widely used measure of the loss of biodiversity, extirpation (local extinction) of populations or meta-populations is more prevalent, but often flies under the radar in terms of awareness. The Endangered Species Act protects species that are "in danger of extinction throughout all or a significant portion of its range" (USFWS 2013). As a result, species can and do become extirpated within watersheds without protection from the Endangered Species Act because they maintain viable populations elsewhere. For instance, lake sturgeon were extirpated from the Red River of the North and Western Lake Superior basins. Their current population is estimated to be at less than 1% of their historic abundance. However lake sturgeon are not considered endangered because there are relatively healthy populations in other segments of their North American range. Therefore, as a measure of lost biodiversity, extirpation may be a more comprehensive indicator than extinction.

Causes of Extinction, Extirpation and Loss of Native Biodiversity

Causes of extinction are difficult to definitively identify due to concurrent changes over the same timeline (Duncan and Lockwood 2001). Of the 27 fish species that have gone extinct in North America in the past century, Miller et al. (1989) found multiple causes cited for 82% of the cases. Loss of physical habitat was cited for 73%, introduced species for 68%, chemical habitat alteration including pollution for 38%, hybridization for 38%, and overharvest adversely affecting 15%. Due to the interaction of variables within aquatic ecosystems, there is frequently a mix of primary and secondary factors (causes and symptoms) within published articles. Prevention or reversal of further loss of biodiversity will depend on the correct identification of the underlying causes.

Alteration of the Landscape

Land-use changes are widely cited causes of habitat loss and resulting loss of biodiversity. Cultivation, agricultural drainage, and urbanization can cause hydrologic changes (Dunn and Leopold 1978), erosion, pollution, sedimentation (Schottler et al. 2013), and habitat loss, all of which result in changes in ecological processes, species composition, and potential loss of biodiversity (Lake et al. 2000). While land-use change is the underlying cause in this sequence, hydrologic change, erosion, sedimentation, habitat loss, and competition or predation by other species may all be identified as causes. Within Minnesota, paleolimnological studies of Lake Pepin and the St. Croix River have shown that cultivation, agricultural drainage, and deforestation caused substantial increases in the rate of sedimentation and eutrophication (Schottler et al. 2013; Edlund et al. 2009). Cultivated lands generally have significantly greater surface runoff and sediment yields than forested or grassland landscapes (Dunn and Leopold 1978). Changes in hydrology and geomorphology translate to changes in habitat and stream processes that can have substantial effects on biodiversity.

Stream Channelization and Ditching

Stream channelization and ditching homogenizes and eliminates critical habitat for native species while providing habitat for species that are habitat generalists, often introduced species. Stream channelization creates a uniform run, while largely eliminating pools and riffles, and filling the interstitial spaces among the substrates with silt resulting in reductions in the abundance of riffle species (Berkman and Rabeni 1987). Loss of habitat diversity associated with channelization favors generalized species, especially herbivores and omnivores, while reducing the abundance of piscivores (Schlosser 1982). Channelization also alters flood frequency and magnitude by increasing velocity and slope, while reducing storage (Moore and Larson 1979; Demissie and Khan 1993). Channel straightening steepens slope which alters sediment transport and can result in upstream channel incision and downstream aggradation which further degrades natural stream habitat necessary for sustaining native diversity (USACE 1994; Urban and Rhoads 2003).

Overharvest

Overharvest has been cited as a cause for species declines, extirpations, and for 15% of the extinctions of some food fishes in North America (Miller et al. 1989). Large-bodied fishes are generally considered more vulnerable to extinction than small-bodied species and are specifically more vulnerable to harvest driven extinction (Olden et al. 2007; Sodhi et al. 2009). Overharvest is considered a particularly significant threat to the sturgeons due to their reproductive vulnerability (easily spotted congregations in spawning areas, intermittent spawning, and old age of maturation - sexual maturity reached at 25 years old for females and 15 years old for males) (Smith 1968; IUCN 2004). While overharvest was likely a contributing factor to the collapse of lake sturgeon in the Red River of the North and Lake Superior basins, concurrent loss of spawning habitat due to dam construction preceded the collapses (Auer 1996; Aadland et al. 2005).

Water Pollution

Water pollution was cited as a contributing cause in 38% of North American fish extinctions (Miller et al. 1989). Pollution mortality of aquatic species can either be short-term, due to toxic chemical spills, or pervasive such as circumstances associated with ongoing release of human or animal waste.

Prior to the Clean Water Act of 1972 and subsequent improvements in wastewater treatment systems, the Mississippi River from Minneapolis - St. Paul to Hastings was a "dead zone" due to untreated human waste. This river reach was almost completely devoid of fish or mussels and acted as a chemical barrier to upstream fish migration (Eddy

Eutrophic agricultural ditch in southern Minnesota. *Credit DNR SHP.*

et al. 1963). Other rivers had frequent fish kills due to untreated paper mill effluent (the St. Louis River near Duluth and the Rainy River near International Falls) or untreated whey from a cheese plant (Otter Tail near Fergus Falls).

Pollution is unlikely to be the sole cause of extinction in a connected river network but may be a contributing factor for species with restricted ranges (Allan and Flecker 1993). The Upper Mississippi River between the Twin Cities and Hastings is evidence of the ability of species to recolonize following extirpation by pollution as both fish and mussels have reestablished diverse assemblages since improvements in water quality were made. Similar recolonizations have occurred in the Rainy, St. Louis, and Otter Tail rivers. However, in rivers fragmented by dams, recolonization cannot occur unless viable populations of a species exist upstream of the current barrier.

Introduction of Non-native Species

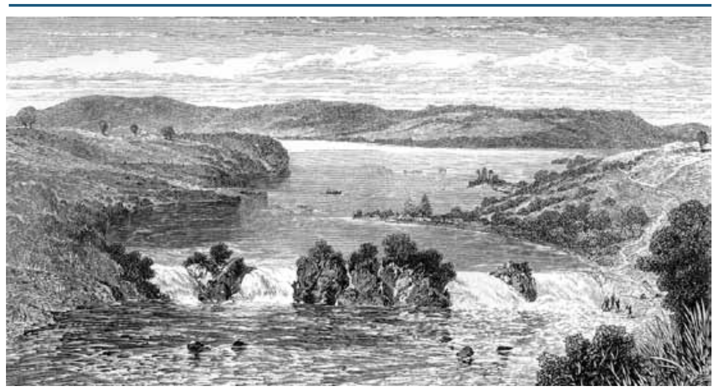
There is debate regarding the primary role of introduced species in causing extinctions in cited continental ecosystems due to concurrent changes in habitat loss, fragmentation, land use, water quality, and other factors (Gurevitch and Padilla 2004). Miller et al. (1989) identified invasive species as a contributing factor in 68% of North American fish extinctions.

There are numerous examples of concurrent habitat change and invasion. In the upper Mississippi River watershed, the role of common carp in habitat destruction has been debated (Dymond 1955; Ellis 1973; Becker 1983). Surber (1923) suggested that habitat destruction and turbidity in Lake Shetek blamed on common carp was more likely due to an outlet dam that raised water levels and caused extensive shoreline erosion.

The introduction of Nile perch (*Lates niloticus*) in Lake Victoria, Africa and associated extinctions of endemic species is one of the most widely cited examples of invasive species caused apparent extinctions. Lake Victoria is a geologically young, isolated basin that has dried out several times over its history, most recently about 14,000 years ago (Danley et al. 2012). Cichlid speciation occurred in the lake over the past 10,000 to 15,000 years resulting in over 500 species, many of which are endemic. An estimated 200 out of 300+ species of endemic haplochromines are believed extinct following Nile perch introductions (Witte et al. 1992). Nile perch are top predators that grow to over 400 pounds. This introduction provided a major fishery, leading to human population growth and associated land use changes, over-exploitation, deforestation, and eutrophication. The fishery for native species was declining by 1928 and had collapsed prior to the introduction of Nile perch in the 1950s. Eutrophication of the lake, deep water anoxia, overfishing, and other changes make it difficult to separate respective roles in the decline of the native species (Gurevitch and Padilla 2004). In 1954 the Nalubaale Dam was built at Owens Falls 4 km downstream of Lake Victoria which inundated Ripon Falls, the natural outlet of the lake. While outflow is, by treaty, intended to mimic natural outflow, significant changes in lake levels have been attributed to dam operations (Kull 2006).

Ross (1991) suggests that the best documented cases of extinctions due to introduced species are associated with top predators (e.g. trout, McDowall 1990; Nile perch, Baskin 1992; peacock bass, Pelicice and Agostinho 2009). Hybridization between non-native and closely related native species was cited as a cause for 38% of North American fish extinctions (Miller et al. 1989).

Distinguishing causation is particularly problematic in impounded river systems. For instance, the Lower Colorado River is impounded by



(top) A drawing of Ripon Falls, the natural outlet of Lake Victoria, by explorer John Hanning Speke, *courtesy of Princeton University Library*. (bottom) Owen Falls Dam/Nalubaale Power Station. *Credit USDA Foreign Agricultural* Service.

stocked in the reservoirs that were no longer suitable habitat for native species. Presently the Lower Colorado River system is home to nearly 80 non-native fish species and of the remaining native species 5 native species have been extirpated, 7 native species remain but are federally endangered, and 3 native species are being reintroduced (Mueller et al. 2005).

Similarly, the White River in Colorado was dominated by native species prior to construction of a main stem dam, but after the dam was constructed it was comprised of 90% non-native species in the reservoir and 80% non-native species below the dam (Martinez et al 1994).

Understanding the mechanisms by which introduced species succeed or fail is critical to their management. Habitat alteration and degradation creating conditions that are favorable for some tolerant introduced species, but are unfavorable for many native species, are likely to set the stage for invasion (Allan and Flecker 1993).

Fragmentation is considered a major threat to biodiversity for a wide range of taxa, both terrestrial and aquatic.

Fragmentation by Dams

Fragmentation is considered a major threat to biodiversity for a wide range of taxa, both terrestrial (Gascon et al. 1999) and aquatic (Haag 2009, Rinne et al. 2005; Allan & Flecker 1993). A total of 87,359 dams greater than 6 feet tall (National Inventory of Dams USACE 2013) and millions of smaller dams and barrier road crossings have been built across the United States. Minnesota rivers are fragmented by over 330 dams greater than 20 feet tall, over 900 dams several feet tall, and countless road crossings (Figure 2).

Dam construction has been considered the primary cause of extinction of freshwater mussels in North America (Bogan 1993; Haag 2009). For freshwater fishes, Harrison and Stiassny (1999) listed habitat loss (including that related to dams) as the leading cause, followed by non-indigenous species, exploitation, and pollution. Taylor et al. (1996) report that 48% of crayfish species are listed as endangered, threatened, or special concern with the primary threats being habitat loss due to dams and channelization, in addition to the introduction of non-indigenous crayfish.

The difficulty of identifying a single cause of extinction is demonstrated when considering the known changes associated with dam construction. These changes include inundation of riffles, rapids and other critical habitat; alteration of flow, temperature, chemical, sediment, and nutrient regimes; downstream channel erosion and incision; upstream sedimentation and aggradation; blockage of fish migrations; water quality and dissolved oxygen changes; and creation of habitat conducive to introduced species. The success and dispersal of introduced species have been linked to impoundment by dams by a number of authors (Havel et al. 2005; Johnson et al. 2008). As a result, assessments of extinction causes in impounded river systems often list physical and chemical habitat loss, blockage of fish migration, altered temperature regimes, water extraction, flow regulation, and introduced species as contributing causes yet each of these, virtually all of the causes cited by Miller et al. (1989), can be caused or initiated by dam construction and impoundment.

The Island Effect

The vulnerability to loss of biodiversity in river systems fragmented by dams can be likened to islands. Islands are particularly vulnerable to the loss of biodiversity with 72% of all documented extinctions since 1500 AD for birds, mammals, reptiles, amphibians, and mollusks due to genetically isolated endemic populations with traits such as flightlessness in birds that make them

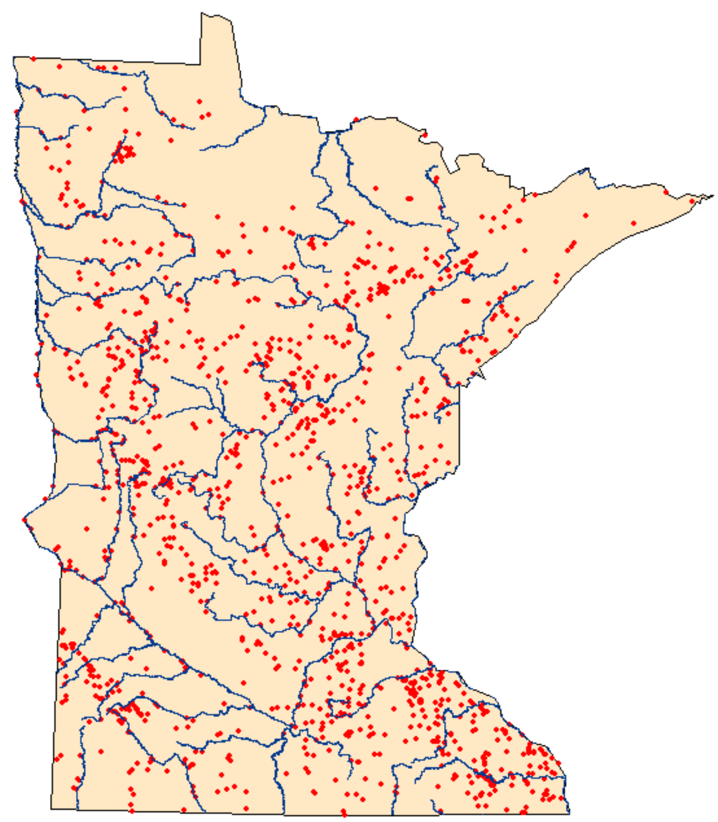


Figure 2. USACE National Inventory of Dams which only includes dams over 6 feet tall (n = 1,300+ dams).

susceptible to introduced predators and over-exploitation (IUCN 2004; Sodhi et al. 2009). Since islands are, by definition, isolated and fragmented from mainlands, there are not direct pathways for recolonization of species unable to fly or otherwise traverse oceans once a population has been lost. Though colonization by new species is considered a threat to the species already present, the biodiversity of islands is still a result of colonization by immigrating species and the endemic species that inhabit them were, initially, invaders. Giant tortoises and iguanas, for example, are believed to have originally immigrated to the Galapagos Islands from South America, where related species exist, by riding floating rafts of vegetation for a distance of roughly 600 miles (Thompson 2014). Species richness for plants and mammals on islands increases with island size, proximity to mainlands, and habitat diversity (Kohn and Walsh 1994; Heaney 1984), while extinction rates increase as island size decreases (Heaney 1984). Newly formed volcanic islands colonize quickly with colonization rates increasing with island size and proximity to the mainland. Extinction rates are initially low, but increase as species richness on the island increases. The diversity of freshwater fish tends to be very low on islands. Fish species richness in Hawaii included no strictly freshwater species but has increased by 800% due to introductions (Thompson 2014).

Isolated lakes show many similarities to islands in terms of species richness. Some isolated mountain lakes were historically fishless with very low diversity of aquatic species. Crater Lake, Oregon, the deepest lake in the United States, was fishless until the late 1800s when six species were introduced. Two of these, rainbow trout, *Oncorhynchus mykiss*, and Kokanee salmon, *Oncorhynchus nerka,* maintain populations but growth rates are low because primary productivity and spawning habitat is limited due to the lack of inflowing streams. Crater Lake has several native amphibians, one of which, the Mazama newt, *Taricha granulosa mazamae*, is an endemic subspecies. The signal crayfish, *Pacifastacus leniusculus*, introduced in 1915 as fish forage, is considered detrimental to the Crater Lake newts by predation and competition for insects.

Crater Lake, Oregon - a historically fishless lake. *Credit DNR SHP.*

Like Islands, isolated lake systems exhibit increased species richness of fish with increased size and connectivity to streams and other lakes (Tonn and Magnuson 1982). Glaciation of Minnesota, as recent as 10,000 years ago, formed most of our lakes and is likely too recent for significant genetic drift and the development of unique endemic species. In contrast, two thirds of the 1,700 plants and animals in Lake Baikal, Russia are endemic. This ancient rift lake is estimated to be 25 to 30 million years old and is volumetrically the largest freshwater lake in the world. The existence of closed basin lakes in Minnesota with fish assemblages predominantly comprised of widely distributed species (in addition to introduced game species) suggests periodic connections and dispersal. These connections are important to maintaining diverse populations in lakes (Hugueny 1989) as they are in river systems (Horowitz 1978).

Connectivity and Biodiversity

Unlike islands, most of Midwestern waters have been connected by stream networks. The fish and mussel communities depend on these connected networks for recolonization after droughts and severe winters, for spawning, and for accessing seasonal and changing habitat needs over their life history (see Drivers of Migration, pg 52). This is supported by the fact that rivers flowing into the sea are less diverse than similar-size rivers that are tributaries of larger river systems. This may be due to their downstream connection to refugia in the larger river systems that allow recolonization and reduce extinction rates (Oberdorff et al. 1997; Hugueny 1989).

The recent glaciation of much of Minnesota that carved most of our lakes would have initially left fishless ecosystems that were subsequently colonized by migrating fish via rivers and streams. The inadequate time period for speciation since glacial retreat (<15,000 years) and a general lack of isolated water bodies left Minnesota with no endemic fish species. Initially waters would have been cold water systems that slowly warmed as the glaciers receded. Species composition of most of our waters has likely been in flux over the past

15,000 years and the term "native" must be viewed in a dynamic context. The interconnected network of rivers and streams has provided fish communities with the resilience to access streams where suitable habitat exists when other streams may be experiencing droughts, floods or other conditions that make them unsuitable. This connectivity also allows fish populations to disperse and adjust to changing climate (Oberdorff et al. 1997).

In determining the causes of global loss of freshwater biodiversity, aquatic habitat loss and degradation due to land use changes (cultivation, deforestation, and urbanization); stream channelization; and dam construction are found to be the prominent causes behind the success of introduced species (that have traits facilitating survival in these altered systems) and the extirpation of native species. Across North and South America, fragmentation of river systems has been one of the most widely cited causes of extinction and extirpation of native species (Rinne et al. 2005) as it has been in the Midwest (Aadland et al. 2005).

Fragmentation of river systems has been one of the most widely cited causes of extinction and extirpation of native species.

Unintended Consequences

Efforts to control introduced species may actually facilitate their success and expansion if they also directly eliminate native fauna or alter the habitat upon which native species depend. These changes can eliminate competition and predation influences that would otherwise resist invasion.

Examples of this unintended effect exist across multiple ecosystems.

1) A dangerous pathogen, *Clostridium difficile*, kills around 30,000 people per year in the U.S. and typically infects patients after antibiotic treatments associated with minor surgery. The antibiotic treatment, which kills both the natural and disease causing bacteria, leaves the patient without natural gut fauna making them much more vulnerable to infection by *C. difficile*. An effective treatment for this very difficult to treat antibiotic resistant infection has been the use of fecal transplants to restore the microbial fauna in the gut (Mayo Clinic 2013).

2) The use of broad spectrum herbicides for weed control leaves the ground bare of vegetation so it is re-colonized by weeds that are adapted to quickly colonize disturbed areas. In addition, wide scale use of glyphosate (i.e. Roundup) has led to a long list of herbicide resistant "superweeds". At least 195 weed species have evolved resistance to 19 herbicides (Waltz 2010). Diverse prairies have been shown to resist colonization by introduced weeds (Naeem et al. 2013).

3) Rotenone and other toxicants have been used to reclaim carp dominated eutrophic lakes since at least the 1950s. By temporarily eliminating all fish, this allows zooplankton to increase and graze down phytoplankton. This increases water clarity and favors the growth of submergent macrophytes. While this approach can be effective in creating a clear water condition initially, carp quickly recolonize and often reach high densities within a few years following treatment. Since the entire fish community is eliminated by the toxin, there is little initial competition or predation influence on the recolonizing carp. High native fish diversity has been shown to mitigate the impact of introduced common carp (Carey and Wahl 2010). Silbernagel (2011) found that native fish in stable lakes controlled recruitment of common carp by preying on the eggs and larvae. It follows therefore, that the elimination of entire fish communities makes them vulnerable to colonization by introduced species.

Efforts to control introduced species may actually facilitate their success and expansion if they also directly eliminate native fauna or alter the habitat upon which native species depend.

Montevideo Dam on the Chippewa River days before it was modified for fish passage in 2012. *Credit DNR SHP.*

Implications for the Minnesota River

The Minnesota River is thought to be vulnerable to colonization by silver and bighead carp due to its unimpeded connection to the Mississippi River and its eutrophic waters due to its agriculturally dominated landscape. A key question is; if they do colonize the Minnesota River, what status are they likely to attain? Possible scenarios include a range of abundance, seasonality of abundance, and life history dynamics that may or may not include reproduction within the Minnesota River Basin.

The bigheaded carps are widely distributed in the Upper Mississippi River basin, but the highest densities are often associated with rivers that have several characteristics:

1) Nutrient-rich and turbid; often associated with cyanobacteria blooms increasing available food supply.
2) Impounded, which increases the amount of low velocity habitat that all life stages except spawning prefer.
3) Regulated flow regime; that reduces seasonal variability in the river and backwaters.
4) Impounded backwaters (often anoxic) with water levels maintained at normal pool elevations.
5) Altered channels for navigation including channelization that creates homogenous habitat unsuitable for native species and constructed wing-dams that create low velocity refugia.
6) Depleted native fish fauna; reducing competition and predation controls.
7) Large river size; typical habitat of these species.

The Minnesota River shares only some of these characteristics with systems where bigheaded carps have done well. The Minnesota River, like the Illinois River, is one of the most nutrient enriched rivers in the U.S. due to agricultural runoff (Figure 3). It is also connected to the Upper Mississippi River where bigheaded carps are established. The Minnesota River Basin has backwater wetlands, eutrophic lakes connected by small streams, and is low in gradient.

The Upper Minnesota is impounded by dams at Granite Falls, Lac qui Parle, Marsh Lake, Highway 75 and Bigstone Lake facilitating phytoplankton and cyanobacteria blooms (Figure 4). Conversely, the lower Minnesota is free of dams over its lower 240 miles. Seasonal fluctuations lead to confined pools in late summer and fall that may limit available habitat for bigheaded carps and make

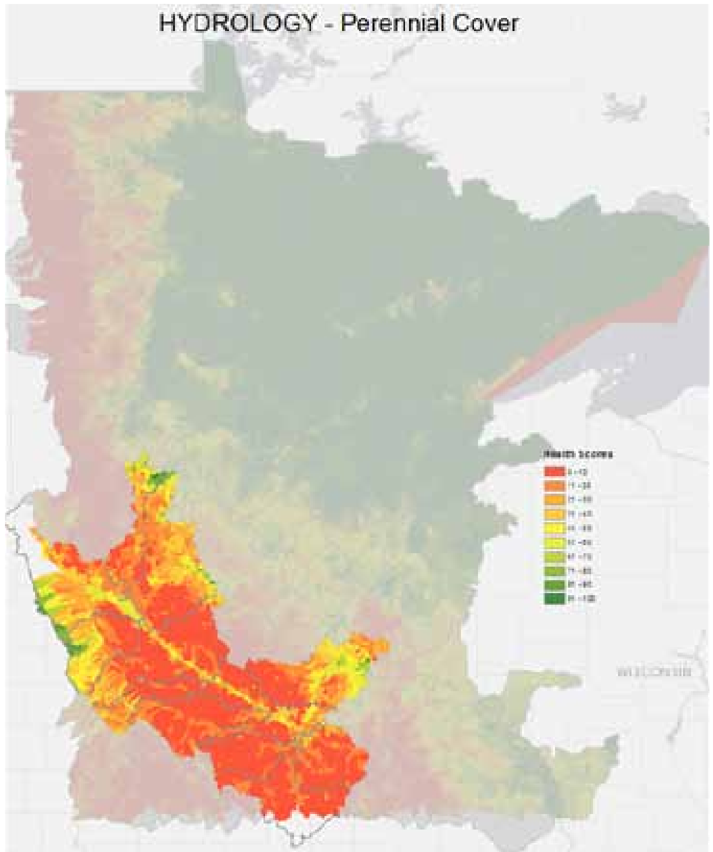


Figure 3. Map of loss of perennial cover for lands in the Minnesota River Basin. Perennial cover is permanent vegetation that covers the landscape year-round (green). Permanent vegetation is removed to be converted to cropland or developed for human use (red). See the Appendix for more watershed health conditions. *Credit MN DNR Watershed Health Assessment Framework (WHAF).*

Figure 4. Dams throughout the Minnesota River basin (●). Dams on the Minnesota River mainstem and Lock & Dam 2 downstream of the confluence with the Mississippi River (△).

them vulnerable to predation. Just downstream of its confluence with the Mississippi, it is impounded by Lock and Dam 2 on the Mississippi River. This reservoir creates connected lakes and backwaters that may be conducive to juvenile bigheaded carps.

High organic turbidity typical of the Minnesota River can limit phytoplankton productivity by limiting light penetration (Dokulil 1994). This in turn may be a limiting factor for planktivorous bigheaded carps.

The Minnesota is relatively small compared to most rivers where bigheaded carps spawn. For instance, silver and bighead carp spawn in the Missouri River, but significant spawning was not observed in 6 tributaries to the Missouri, including the Osage River which averages over twice the flow of the Minnesota River. The Lower Minnesota has a diverse fish assemblage with a healthy population of flathead catfish, longnose and shortnose gar, mooneye and other potential predators on life stages of the bigheaded carps. The northerly latitude of the Minnesota River would favor a later spawn of bigheaded carps and a shorter growing season for juveniles prior to winter. Flows in the Minnesota are typically low by mid-summer which disconnects or dewaters many backwaters and would subject juvenile carp to predation in the main river channel.

It is noteworthy that bigheaded carps have not yet been documented from the Minnesota River. There are several possible explanations:

(1) Bigheaded carps have not reached the Minnesota River, but may eventually do so,

(2) They are not present because of lack of habitat or other factors, or

(3) They are present but have not been observed.

The probability of observing bigheaded carps is relatively low because there is not a commercial fishery on this river.

The possibility of natural reproduction is unknown; however grass carp, with similar spawning requirements as bigheaded carps, have occasionally been found in the Minnesota River basin, but there is no evidence of reproduction to date.

The lack of observations of bigheaded carps in the Minnesota River does not appear to be due to the distance from downstream reproducing populations. From the confluence of the Missouri and Mississippi Rivers, silver carp have dispersed a distance of over 1,220 miles passing more than 40 low-head dams up the James River to the

(top) Lac qui Parle reservoir in the upper Minnesota River, note agricultural landscape and green turbid water. (bottom) A reach typical of the lower Minnesota River - meandering with meander cutoffs that fluctuate seasonally. *Google Earth images.*

The confluence of the Minnesota River (flowing to NE) and the Mississippi River (flowing from west) showing much higher turbidity in the Minnesota River. *Google Earth image.*

Jamestown dam. In comparison, the distance up the Mississippi River from the confluence of the Missouri River to the mouth of the Minnesota River is 648 miles. This suggests that the distance and latitude are not limitations. Whether there are other factors that limit bigheaded carps in Minnesota or that it is just a matter of time before they become established is, at present, uncertain.

Changes to Minnesota Streams

Aquatic ecosystems in Minnesota, and the Midwest in general, have been highly impacted by human activities. In these impacted areas, water quality is generally poor and aquatic habitat is degraded due to various issues, including fertilizer runoff, increased bank erosion from channelization, tiling, ditch construction, and dam construction. These altered conditions can favor introduced species by creating novel conditions that are a) favorable for tolerant species and b) unfavorable for native species because they have evolved to the pre-development environment over thousands of years.

Below is a brief overview of human activities that have had major impacts on Minnesota waters. These and other historical facts are summarized in Figure 5, pg 19-20.

Hydrology, Geomorphology, & Connectivity

Landscape Changes

- 1830-1900 Prairie was converted to farmland. Now less than 2% remains untouched.
- 1850 Swamp Land Act that encouraged drainage of wetlands for cultivation.
- 1821 First sawmill built at St. Anthony Falls, 1840 MN lumbering boom began, peaked in 1899. Rivers, steamboat, then railroad were used to transport cut timber. Less than 4% of MN forests have been untouched.
- 1862 First railroads reach MN to move timber and grain accelerating timber harvest and land conversion.
- Homestead Act of 1862 made public land available to be cultivated.
- 1950s Use of drain tile became widespread.

Dams

- 1859-1910 Most major rivers were blocked. (Figure 6)
- 1910s-1960s Mississippi River was channelized and fragmented by numerous locks & dams. There are 43 dams between Lake Itasca & St. Louis (29 are locks & dams downstream of Twin Cities).

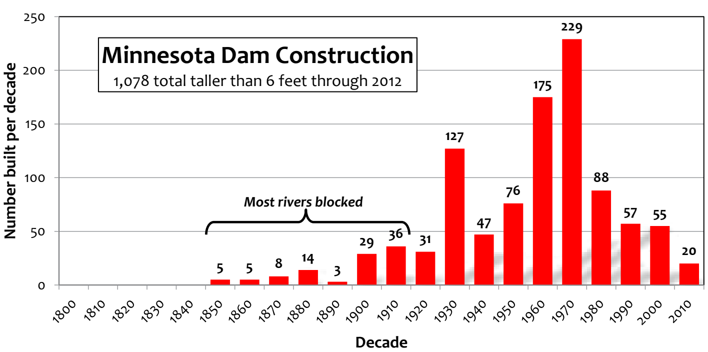


Figure 6. The number of dams constructed every decade in Minnesota.

- 1930s peak in dam construction in response to the drought from 1931 to 1935 . As of 2013 a total of 1,078 dams taller than 6 feet tall have been built across MN.
- 1994-present Several dams removed for ecological, economic, safety, and recreational reasons. As of 2013 over 20 dams have been removed and another 30+ have been modified for fish passage.

Channelization & Ditching

- 1858 first Drainage Act in MN.
- 1900-1915 Proliferation of drainage activity in MN, especially near large trade centers and the railroads.
- 1948 Army Corps of Engineers authorized by congress to channelize several rivers in NW MN, including the Otter Tail, Wild Rice, Sandhill, Marsh, Mustinka, Sheyenne, Rush, Bois de Sioux, Maple and Clearwater rivers.
- Now almost one third of MN streams are now ditched or channelized.

Biology

Fish

- 1880 -1890 Carp are introduced.
- 1880s Stocking of walleye and a number of introduced species began in response to declining native stock due to habitat loss and overfishing.
- Prior to 1890 No restriction on fish harvest.
- 1890 Local law enforcement agencies are given fish & game authority. One game warden for entire state.
- 1891 depletion of fish by seining is noted by the Game and Fish Commission and efforts began to prevent overharvesting of fish.
- 1924 63 game wardens patrol the state.
- 1927 Efforts begin to block carp movement.
- 1947 Last lake sturgeon recorded in Red River of the North basin.
- 1960s State begins to use rotenone to reclaim carp lakes.

Mussels

- 1880s Commercial harvest of mussels began in earnest.
- By 1900 clammers could not find live mussels in Mississippi above Lake Pepin, 43 species gone.
- Currently 21 of 43 mussel species no longer exist in the Minnesota River watershed.

Water Quality

- 1840s The sediment flux to Lake Pepin from the Minnesota River began to increase and reached 5 times background rates in 1940s and over 8 times by 1960s.
- 1885-1980s Many rivers polluted with raw human sewage. Example: 1800s -1970s Raw sewage from the Twin Cities polluted the Mississippi River & created a low oxygen dead zone.
- 1930s fish kills were common.
- 1950s Use of fertilizer & pesticide became widespread.

Changes in the Minnesota River Basin

The Minnesota River is a prominent feature of the southern Minnesota landscape. Early explorers told of a river that was swift and clear, and supported an abundance of freshwater mussels (more accurately termed unionids). George W. Featherstonhaugh was one such explorer, and he recorded observations from his 1835 journey up the Minnesota (Featherstonhaugh 1970 [1847]), or St. Peter's River as it was called at the time. With regard to mussels he writes:

> *"A great profusion of unios were lying in the sandy bottom, buried to their umbones; the species called [Unio] fasciatus [currently recognized as the mucket,* Actinonaias ligamentina*; Figure 5.f1], with singularly beautiful nacres tinged with a brilliant carnation, being the most prevalent… some specimens of which outstripped in elegance any I had yet seen. I made a good collection of these shells…." He goes on to write that "… the water [was] beautifully transparent, and the unios stuck in countless numbers in the pure white sand, so that I could, by baring my arm, select them as we went along."*

In comparison with Featherstonhaugh's historical account, conditions in the Minnesota River Basin today have changed drastically (Figure 7). In Minnesota, watershed health scores have been derived and are presented online, by the Department of Natural Resources, Watershed Health Assessment Framework (WHAF, www.dnr.state.mn.us/whaf/index.html). This website provides detailed explanations of the health indices, calculation protocols, metadata, and other fundamental background for the results reported below, which are briefly summarized in Appendix A.

Hydrology and Geomorphology

The Minnesota River Basin is now predominantly agricultural land use, and much of the Basin's perennial cover has been converted to crops. (Appendix A, Figure 1). Altered hydrology in these areas, including the Minnesota River Basin, is a common condition, reflecting the change in runoff resulting from conversion to annual vegetation, tile and other drainage, and altered water courses (Appendix A, Figure 2). Impervious cover further alters runoff and basin hydrology and is generally low in the Basin, however there are catchments throughout the Minnesota River Basin that have high amounts of impervious cover (Appendix A, Figure 3). Water use, as a percent of available water is generally low, with notable exceptions in the Chippewa and Minnesota River – Shakopee major

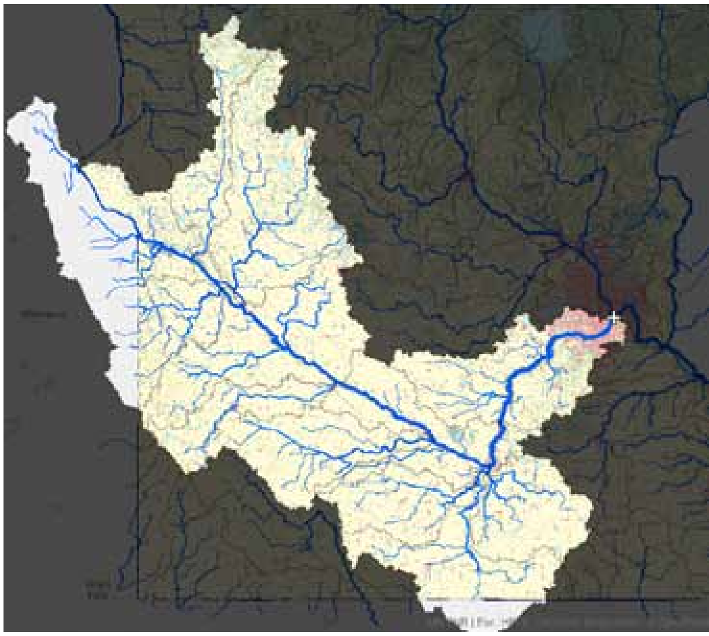


Figure 7. A map of 2006 National Land Cover Database with the Minnesota River Basin highlighted. Cultivated land (light yellow) comprised 77.5 % of the watershed in 2011. *Credit MN DNR Watershed Health Assessment Framework.*

watersheds (Appendix A, Figure 4). Catchments in the Basin are not particularly susceptible to soil erosion, but there are areas that are worse than others, for example in the steeper landscapes that are associated with the Basin's Coteau des Prairies region (Appendix A, Figure 5).

Connectivity

As a consequence of the conversion of native prairie, hardwoods, wetlands, and other lands in this basin to crops, there is only small patches of native habitat, so terrestrial connectivity is uniformly poor/low, especially when scored at the watershed scale (Appendix A, Figure 6).

Riparian connectivity, on the other hand, is more spatially variable, especially when evaluated at the catchment level. Riparian connectivity is generally higher along the mainstem of the Minnesota River, in the headwaters of the Basin, and near the confluence of the Minnesota River with the Mississippi River (Appendix A, Figure 7). The downstream portion of the Minnesota Basin is largely residential and mixed land use.

Aquatic connectivity, which is a measure of the number of bridges, dams, and culverts on the stream network, is a mixture of good and poor throughout the Basin (Appendix A, Figure 8). It should be noted here that the mainstem Minnesota River is free-flowing for approximately 240 miles, up to its first dam at Granite Falls, MN.

1800s

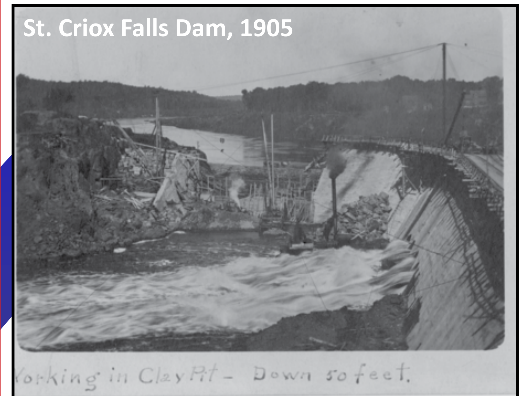
1800s-1808 • Early accounts of fish diversity & abundance & habitat in northwest MN by Alexander Henry

1835 • George Featherstonhaugh took notes about easily identified food fishes.

1880s Stocking of walleye, and a number of introduced species began in response to declining native stock due to habitat loss and overfishing.

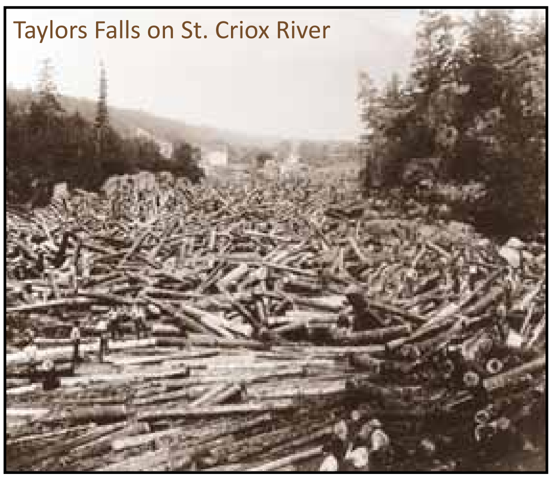
1880s • Commercial harvest of mussels began in earnest, by 1900 clammers could not find live mussels in Mississippi above Lake Pepin, all 43 species gone for a century.

St. Criox Falls Dam, 1905



1859-1910 • Many major rivers blocked by dams

Taylors Falls on St. Criox River



1820s-1900s • Timber harvest, peaked in 1899. Rivers, steamboat, then railroad were used to transport cut timber.

Less than 4% of MN forests have been untouched.

1830-1900 • Prairie was converted to farmland.

1891• Depletion of fish by seining is noted by the Game and Fish Commission and efforts began to enforce overharvesting of fish.

1650s -1830s • Fur trade drew explorers and traders to MN (beaver almost extinct by 1837).

Over 7,500 years ago Native Americans inhabited the state.

Over 15,000 years ago MN was covered by glaciers so lakes & rivers were devoid of fish & mussels.

1840s • The sediment flux to Lake Pepin from the Minnesota River began to increase and reached 5x background rates in 1940s and over 8x by 1960s.

1862 • First railroads reach MN to move timber and grain accelerating timber harvest and land conversion.

Homestead Act of 1862 made public land available to be cultivated.

1850 • Swamp Land Act that encouraged drainage of wetlands for cultivation.

1858 • MN first Drainage Act

1887 • First Comprehensive Drainage Law

1897 • State Drainage Commssion

Landscape Changes
Documentation of Fish Diversity
Dams
Ditching & Channelization
Introduced Species
Mussels and fish
Water Quality

1800s

Figure 5. Timeline of human activites that affected watershed and stream health.

Question 1. Why are some species successful while native species decline?

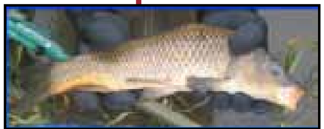
1900s — **2000s**

1895 • Earliest fish surveys of Red River Basin by Woolman

1920s • Biological exploration, including fish identification and river condition of MN basins by Thaddeus Surber

Documentation of aquatic biodiveristy before rivers were altered and degraded is very limited.

1880s • Common carp are introduced to MN waters



1947 • Last lake sturgeon recorded in Red River of the North basin

1970s • Fish surveys by MN DNR begin

1960s • State begins to use rotenone to reclaim carp lakes

1900 • Common carp considered a nuisance

1927 • Efforts begin to block carp movement

1996 • A bighead carp caught in St. Croix River.

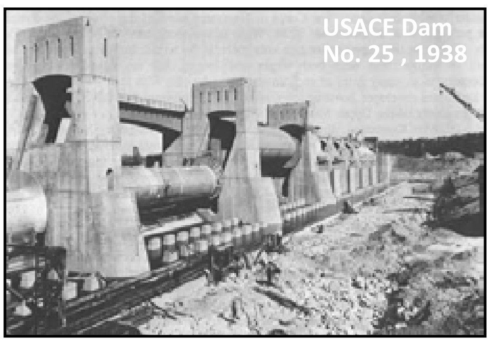
1997 & 2003 • A bighead carp caught in Lake Pepin.

Sewage mats on the Mississippi, 1933

1885-1980s • Many rivers polluted with raw human sewage. Ex. Mississippi River dead zone.

1948 • Army Corps of Engineers authorized by congress to channelize several rivers in NW MN including the Otter Tail, Wild Rice, Sandhill, Marsh, Mustinka, Sheyenne, Rush, Bois de Sioux, Maple and Clearwater rivers.

1910s-1960s • Mississippi River was channelized and numerous locks & dams were built.



USACE Dam No. 25 , 1938

1994-1995 • Several dams removed for ecological, economic, safety, and recreational reasons.

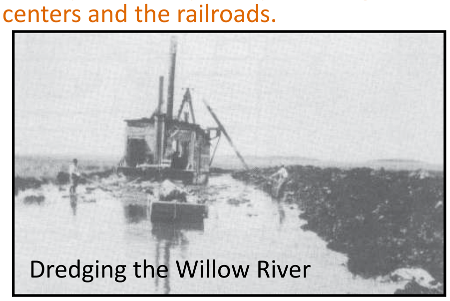
Less than 2% of native priaire remains untouched.

Beltrami County, 1915

1950s • Use of drain tile became widespread.

Almost one third of MN streams are now ditched or channelized.

1900-1915 • Proliferation of drainage activity in MN, especially near large trade centers and the railroads.



Dredging the Willow River

1985 • Farm Bill established Conservation Reserve Program (CRP) & denied benefits on converted wetland.s

1950s • Use of fertilizer & pesticide became widespread.

1991 • Wetland Conservation Act, established a "no net loss" of wetlands.

1972 • Clean Water Act is enacted

1967 • MN PCA established by State Legislature

1900s — **2000s**

Water Quality

Water quality within the Basin is in a general sense, degraded. Impairments for waters, as currently determined by the Minnesota Pollution Control Agency, are presented in Appendix A, Figure 9, and extend throughout the Basin, along the mainstem of the Minnesota River and in many of the tributaries. Much of the waters listed as impaired are affected by sediment and fecal coliform. As for the latter impairment, feedlots are scattered throughout the Basin (Appendix A, Figure 10), and are the likely (and obvious) source for fecal coliform impairments. However, the problem areas are extensive but perhaps not as intensive a problem as this previous graphic might indicate, and the health score results for the number of animal units in the catchments provides a clearer picture of the extent of 'hot spots' for point source pollution in the Basin (Appendix A, Figure 11). Non-point source pollution is summarized by watershed in Appendix A, Figure 12; the Index shown combines two metrics, the rate of chemical application to cropland and the amount of impervious surface in the riparian zone. The results show degraded conditions throughout the Basin, when considered at the watershed scale. Scores of 50 or less for the entire Basin can only reflect the generally intensive agricultural land use, with high fertilizer and chemical application rates.

Biology

Despite the degraded conditions throughout the Minnesota River Basin, the number of fish species is fairly high (97 species), just behind the St. Croix River Basin (106 species), and the 128 species for the Mississippi River below St. Anthony Falls (Table 2).

Table 2. Fish Biodiversity within Minnesota Waters

Basin	# of native fish species
Missouri River Basin	42
Mississippi River Basin above St. Anthony Falls	63 (historic)
Rainy River Basin	73
Red River Basin	83
Lake Superior Basin	83
Minnesota River Basin	**97**
St. Croix River Basin	106
Mississippi River Basin below St. Anthony Falls	128

Table 2. Number of native fish species in each of Minnesota's major river basins.

The IBI-based fish index (Appendix A, Figure 13) does reflect the degraded conditions reported for virtually all other parameters however. This index is based on the fish Index of Biotic Integrity (IBI) published by the Minnesota Pollution Control Agency; IBI site scores were transformed to a 0-100 scale and catchment scores represent an average of fish IBI scores in a given catchment. Scores in the Basin were predominately 50 or below, with a fairly large number of catchments scoring less than 30. These scores reflect poor conditions for fish; reasons for this are complex, but the primary causes are likely pollution and land-use disturbances that result in degraded channel stability, and consequently, poor stream habitat. Loss of riparian corridor, draining of wetlands, and tiling of farmland increases the 'flashiness' of runoff, and results in riverbank erosion and an increase insediment entering and being transported by the river. As would be expected under these conditions, mussels have also declined in the Basin. Fifty-three percent of the original mussel species present (43 species) in the Basin remain today, constituting a loss of 20 species. Of the species occurring in the river, present and past, over half are listed by the State of Minnesota as Endangered, Threatened, or of Special Concern, and three are listed by the Federal government as Endangered. These observations reflect a national (Bogan 1993, Neves 1993) and global (Lydeard et al. 2004) trend of declining freshwater molluscan diversity. The health scores for mussels based the MN DNR statewide survey, not surprisingly, reveals a similar trend - low site quality scores for nearly all of the Basin, with average mussel site quality being the highest and generally only found in the upper extents of the Basin (Appendix A, Figure 14).

The History of Common Carp Introduction in Minnesota

The historical chronology of the events leading up to the introduction of common carp and subsequent management provide important context for introduced species in general. The following is primarily from Hoffbeck 2001 and Eddy and Underhill 1974.

1870s The watersheds and streams began to be altered. Much of the prairie is converted to cropland following the Homestead Act of 1862 and the railroad expansion in the early 1870s. Dam building accelerated and dams blocked many of the major rivers and tributaries.

1874 The Minnesota Fish Commission is established and wants northern pike "outlawed" in most waters.

1880 The first carp are introduced - 15 carp are stocked in lakes near Buffalo, Minnesota.

1882 69 carp are stocked in Lake Como and in Stevens County.

1884 9,000 carp are stocked in 90 different places in Minnesota.

1885 3,150 carp are stocked statewide, state begins stocking walleyes.

1890 Attitude towards carp reverses - last batch of 154 carp are stocked, people begin complaining about carp.

1909 State issues permits for seining rough fish, carp are "deadly enemies."

1927 State builds carp screens between lakes and later, builds control dams.

1942 MN Department of Conservation hires rough fish removal crews.

1960s State begins using rotenone to reclaim carp lakes.

Despite more than a century of efforts to eradicate common carp, they still persist across most of the state and are one of our most widely distributed and abundant species.

Common carp. *Credit Konrad Schmidt.*

Examples of Thriving Silver Carp Populations

The success of silver carp in the lower Illinois River provides insight into habitat conditions likely to favor their success elsewhere.

The Lower Illinois River

This river has the highest densities of silver carp in the world with bigheaded carps comprising up to 63% of the total fish biomass (Sass et al. 2010; Garvey et al. 2012). Their success in this river reach provides insight into habitat conditions likely to favor their success elsewhere.

The Illinois River has been highly altered and polluted. The Chicago Sanitary and Ship Canal, when first opened in 1900, allowed untreated sewage to be diverted down the Illinois River and by 1911 was described by Forbes and Richardson as completely anoxic and sludge-like. Water treatment facilities have improved water quality, but the Illinois River is still impaired for total suspended solids, total phosphorus, nitrates, and fecal coliforms (EPA 2012). Arsenic, aluminum, chromium, lead, and zinc are present at elevated levels in the sediments (Bhowmilk and Demissie 1986) and ammonium toxicity is likely the cause of the disappearance of benthic macroinvertebrates in the 1950s (Sparks 1984).

The river was channelized and lined with levees in the 1930s, which homogenized the river habitat and disconnected the river from its floodplain. The lower 80 miles of the Illinois River (Alton Pool) is impounded by Mel Price Dam, which is the downstream most dam on the Mississippi River. The remainder of the Illinois is impounded by a series of 8 navigation dams (Figure 8). Impoundment affected floodplain and backwater wetlands by stabilizing water levels and favoring sediment deposition. Unlike backwaters on free-flowing rivers that fluctuate with river flows, backwaters created by impoundment maintain more constant water levels associated with the normal pool elevations of the reservoir. The changes were summarized by G.R. Wade, a commercial fisherman (Meredosia Fish Company) as, "*Building of new dams has prevented the fish from coming up from the southern waters of the Mississippi and fish cannot get up-stream to spawn for all low ground has been leveed for farming. Lastly, much pollution is still found in our streams and is killing what fish we have left.*"(Howard Edlen 1976).

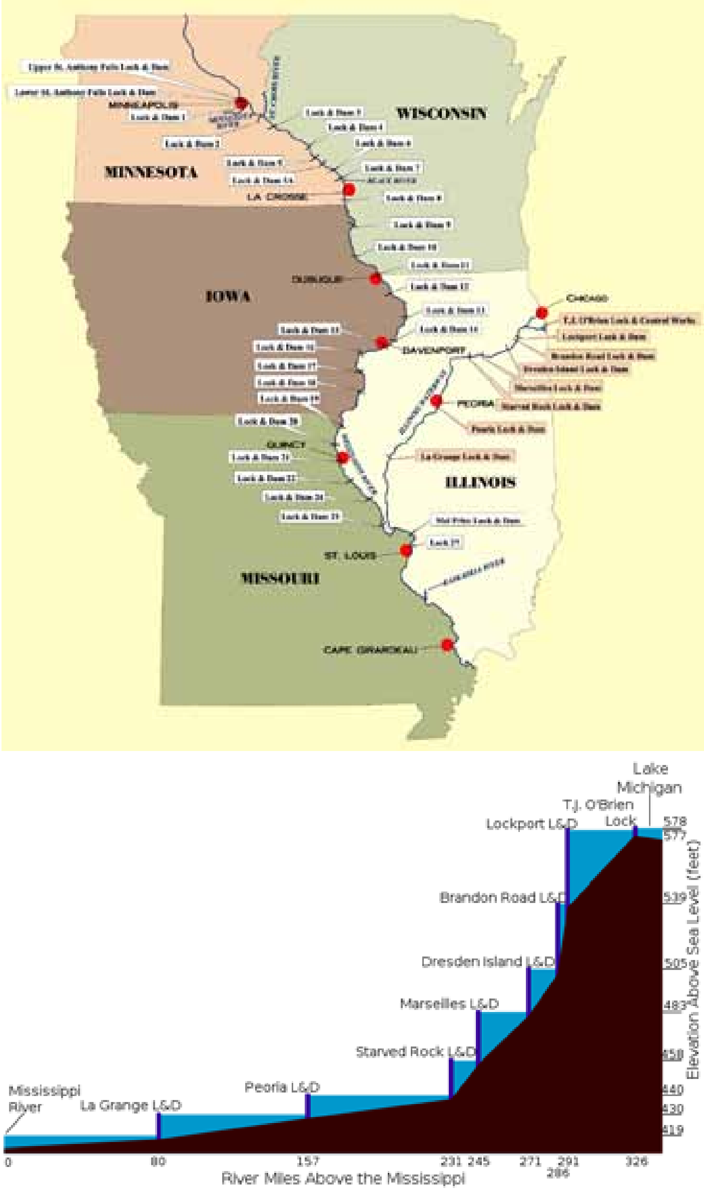


Figure 8. (top) Lock and dam system of the Upper Mississippi and Illinois rivers. *Credit USACE.* (bottom) A schematic of the Illinois Waterway. *Drawn by D. M. Short based on USACE data.*

In comparison, the free-flowing Wabash River, that flows west then south through Indiana to the Ohio River, which flows into the lower Mississippi River (so has the same source of bigheaded carps), while also eutrophic, had one third the bigheaded carp density of the Illinois (Stuck 2015).

The James River

The James River, which is impounded by over 230 lowhead dams as it flows south through North and South Dakota, is hypereutrophic and subject to high water temperatures, low dissolved oxygen, and low flows (Figure 9). The hydrology downstream of the 74-foot high Jamestown Dam is regulated due to dam operations (Berry et al. 1993). In the 1950s, 83 miles of the upper James River, 3 lakes and numerous sloughs in the watershed were

Figure 9. The James River. La Moure, ND (●) where first silver carp was caught in 2011. The Jamestown Dam (▲).

treated with rotenone as an unsuccessful attempt to eliminate common carp (Cadieux 1954). Thirteen species historically found in the James River have not been recently collected, including paddlefish, shovelnose sturgeon, blue sucker, and a number of minnow species (Berry et al. 1993).

In 2011, a juvenile silver carp (18 inches long) was caught near La Moure, ND on the James River then later that year a 17.7 inch, 2.4 pound silver carp was caught below Jamestown dam by ND Game and Fish staff. In 2013 five additional silver carp were caught ranging in size from 23 to 26 inches and 4.8 to 7.8 pounds then on June 18, 2014 three fish were caught that were 26 to 28 inches long and 8.3 to 10.6 pounds (Gene Van Eckhoudt, SE District Fisheries Manager, ND Game and Fish, personal communication 2014). Based on initial otolith examinations, these fish appear to have been spawned in 2009. While it is uncertain whether this was a juvenile migration or the result of a silver carp spawning migration, no adults were caught or observed. Furthermore, the small size and sluggish flows of this impounded river would be well outside of the criteria believed to be required for successful silver carp reproduction.

The James River has very low velocities throughout the year, even during high spring flows (only bridge crossings have localized higher velocities). During record flooding of over 11,000 cfs in 2009, measured mean velocities for the gage at the North Dakota and South Dakota border were less than 0.8 feet per second due to the very low gradient (0.5 feet/ mile) and impoundment by dams. The mean annual flow at the North Dakota – South

Dakota border is 384 cfs and the river frequently stops flowing in the fall. The river actually has occasional reverse flows during south winds due to impoundment and its very flat gradient.

Juvenile silver carp are known to migrate long distances and the impounded and hypereutrophic conditions in the James are typical of their preferred habitat. This year class was subsequently sampled in 2013 and 2014, where they were up to 27 inches long.

The Lower Missouri River

Historically, the Missouri River was a braided river with floods that had two peaks - the first with the initial snow melt in March and April and the second with melting of the Rocky Mountain snow pack in June followed by declining flows in July through fall and winter. The Missouri has been substantially altered by construction of more than 1,200 dams including some of the largest in North America. These dams stabilize flow and shift the timing of flows by reducing spring floods and increasing summer and fall flows. This effect is greatest in South Dakota below Fort Randall Dam and diminishes downstream as less regulated tributaries add flow through the state of Missouri (Pegg et al. 2003). Construction of Fort Peck (1937), Garrison (1953), Oahe (1958), Big Bend (1963), Fort Randall (1952), and Gavins Point (1955) dams resulted in the collective interception of 89,000 acre-feet of sediment per year (USACE 1998) (Figure 10). Downstream effects of the dams extend to the mouth of the Mississippi River where a 70% reduction in sediment supply to the Mississippi delta was observed upon completion of the Missouri River dams in the 1950s (Williams and Wolman 1984). Over 730 miles of the Missouri River is channelized from Sioux City, Iowa to its mouth.

The fish community of the Missouri River has been altered by fragmentation, inundation of critical habitat, flow alteration by dams, habitat loss due to channelization, introduction of non-native species (particularly in reservoirs), and watershed changes. The majority (53%) of species whose populations are increasing were introduced in the reservoirs or river while 96% of the species whose populations are declining are native species (Galat et al. 2005). This paper also documented major declines in sauger populations from 1963 to the early 1990s in the river downstream of Gavin's Point Dam. Pflieger and Grace (1987) found increases in the abundance of planktivorous fishes such as gizzard shad between 1940 and 1983, while fishes dependent on turbid water or specialized river habitat decreased. These changes in abundance occurred prior to the establishment of the bigheaded carps and are an example of the pitfalls of attributing similar changes to competition by bigheaded carps that may be a result of habitat changes or other factors.

The Lower Missouri has established silver and bighead carp populations though density estimates or comparisons have not been published. The Missouri River is similar in size and character to Asian rivers where bigheaded carps are native. Channelization of the lower Missouri creates higher velocities and more continuous turbulence believed to favor spawning and egg suspension while thousands of wing-dams create slack water habitat preferred by juvenile bigheaded carps. Conversely, channelization and levee construction disconnects backwaters and floodplain habitats considered important to juvenile survival. Impoundment and reductions in the suspended sediment load of the lower Missouri decreased turbidity, increased light penetration and the production of phytoplankton (Whitley and Campbell 1974). The increases in phytoplankton would favor planktivorous species such as the bigheaded carps. Finally, the decline of native species, due to loss of riverine habitat, may reduce competition and predation pressures on bigheaded carps.

Figure 10. (top) The Missouri River and tributaries with the major dams along the mainstem marked (▲), Fort Randall Dam (▲).

Wing dams that are prevalent in the lower channelized reach in the Missour River. *Google Earth image.*

Summary:

Loss of native biodiversity has been considered the single most significant environmental issue facing humanity. Freshwater extinction rates are 5 times higher than terrestrial rates. Current extinction rates are estimated to be 100 to 1,000 times faster than those indicated by the fossil record, which provides a background rate. Causes of losses of native biodiversity are numerous but include fragmentation and habitat loss due to dam construction, land use change, stream channelization and ditching, as well as water pollution, overharvest, and introduction of non-native species. Reductions in native biodiversity result in a reduction of competition and predation pressures on introduced species and provide them with an advantage in altered systems.

Species naturally and continuously attempt to expand their range and invade new areas. Their success in new areas depends on their suitability to the new environment, ability to compete for resources or fill a niche, and ability to avoid predation. All fish species in Minnesota are the result of post-glacial invasion.

Species directly introduced by humans have expanded globally, with fishes being the most widely introduced group of aquatic animals. Their success is attributed to human activity, and more specifically to the degradation of habitat, primarily through fragmentation of river networks.

Bigheaded carps have adaptive traits that give them an advantage in impaired waters, including the unique ability to digest cyanobacteria, preferences for impounded or low velocity waters, and tolerance of low oxygen and eutrophic water quality.

Special attention has been focused on the Minnesota River due to its eutrophic state and southerly proximity to the Mississippi River and existing bigheaded carp populations. The Minnesota River Watershed has been altered due to landuse change, primarily associated with conversion to annual row crops. A result of this change is an increase in non-point source pollution, particularly the transport of sediment and associated nutrients. In addition, altered hydrology from ditching and tiling, loss of hydrologic storage, and channel instability with associated loss of riverine habitat has occurred virtually basin wide. These degraded health conditions may predispose the Minnesota River to establishment of bigheaded carps.

Conversely, the Minnesota River mainstem remains free flowing for its lower 240 miles and supports 96 species of native fish. This expanse of free-flowing river and high level of fish diversity provide competition and predation resistance against the establishment of bigheaded carps. Unregulated low flows associated with the river may limit pool habitat for bigheaded carps in late summer, fall, and winter increasing their vulnerability to predation by flathead catfish and other predators. In addition, high inorganic turbidity may limit phytoplankton production thereby limiting planktonic food resources. To date, no bigheaded carp have been collected in the Minnesota River.

Question 2. What are the Effects of Introduced Carps on Native Species and their Populations?

Overview: Introduced carps have become abundant in degraded, fragmented habitats where native communities have concurrently declined. While shifts in plankton composition have been reported, causal effects of bigheaded carps on the species richness or biomass of native fishes are inconclusive, even at extreme densities. This chapter critically examines the current literature on carp impacts to native species and provides an assessment of their likely consequences to aquatic biota, qualified by environmental conditions.

Introduced fish species are increasingly perceived as a significant threat to freshwater biodiversity. This perceived threat is connected to and combined synergistically with habitat loss and fragmentation, hydrological alteration, climate change, over-exploitation, and pollution (Postel and Carpenter 1997, Dudgeon et al. 2006). While there has been a sense of urgency in addressing the dispersal of bigheaded carps, the actual effects of these species on the native aquatic communities is poorly known. "*Ecologists can make some powerful and wide-ranging predictions about invasions. . . On the other hand, ecologists cannot accurately predict the results of a single invasion or introduction event.*" (Ehrlich 1989). Some generalizations are supported by several types of evidence; for instance, all other things being equal, a given species is more likely to succeed in invading a species-poor community than a species-rich community (Lodge 1993). Yet because all patterns are characterized by exceptions and variance, predicting the outcome of any particular introduction cannot be done with much confidence.

While there has been a sense of urgency in addressing the dispersal of bigheaded carps, the actual effects of these species on the native aquatic communities is poorly known.

Efforts to understand the magnitude and array of potential impacts of introduced fishes on freshwater diversity are ongoing. Kolar et al. (2005) recognized the challenge of documenting and quantifying ecological changes due to introduced species, such as bigheaded (silver and bighead) carps. In particular, they cite the lack of knowledge about ecology of fishes or plankton communities in large river ecosystems, in relation to co-varying factors such as changing hydrology, water temperatures, flow rates, abundances of other biota, and human activities, all of which further confound efforts to document the effects of introduced carps. Not surprisingly, studies and documentation relating to the impact of bigheaded carps in the United States are fairly scarce.

Introduced species have little impact on the native community until they become established. Establishment results from propagule pressure (the quality, quantity and frequency of invading organisms, Groom 2006) among other things (e.g., global warming, Ficke et al. 2007). Propagule pressure is recognized as one of the major factors leading to establishment of introduced species, including freshwater fishes, in a new environment. (Ruesink 2005). Establishment rates (% of introduced species that become naturally reproducing) for freshwater fishes have been reported to range between 38% and 77% (Ross 1991). Ruesink (2005) reported an overall establishment rate of 64% for intentional introductions of freshwater fishes (1,424 globally). Fifty percent of 1,205 fish introductions recorded for aquaculture have established in the wild (Casal 2006). As these figures indicate, consideration of the requirements of bigheaded carps for reproduction and growth is fundamental to addressing the risk of propagation and spread. Those requirements are addressed below.

Effects of Bighead and Silver Carp on Native Species

A useful framework for evaluating effects of introduced species on native species is presented by Cucherousset and Olden (2011). These authors recently reviewed existing published papers and used multiple levels of biological organization to provide a current state of knowledge. We adopt their approach (Figure 10) to report general impacts, and provide specifics related to bigheaded carps where available.

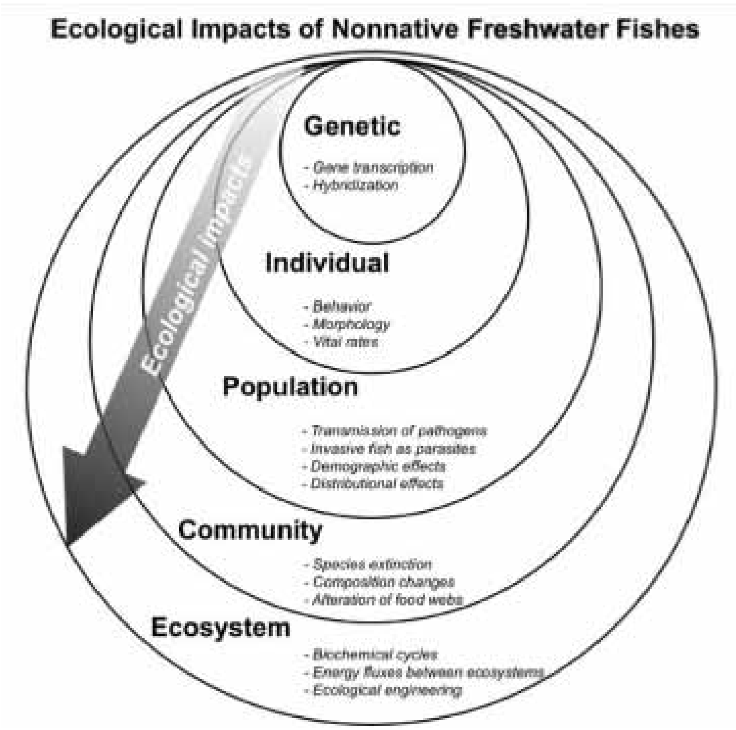


Figure 10. Schematic representation of the ecological impacts of introduced freshwater fishes at the five selected levels of biological organization. The black arrow indicates that the impacts of introduced fish species are often not restricted to one level but cascade across multiple hierarchical levels (from Cucherousset and Olden (2011).

Genetic Level

The impact of bigheaded carps on native fishes through hybridization and subsequent loss of gene pool integrity is not expected because there are no close relatives of these fishes (*Hypophthalmichthys*) in North America (Kolar et al. 2005). Hybridization between bighead and silver carps is common; in the Illinois and Wabash rivers hybrid silver carp comprised 72.9% and 24.5% of sampled fish, respectively (Stuck 2012).

Individual Level

Growing evidence suggests that introduced fishes that are numerically dominant are more likely to modify the behavior of native species (Cucherousset and Olden 2011). Most studies demonstrating altered behavior regards predator prey interactions. Since bigheaded carp are not predators and feed at the lowest trophic level, their effects on the behavior of other species are less likely.

Bigheaded carps are filter feeders, feeding on plankton and organic particles down to 20 µm for bighead, and 10 µm for silver carp (Jennings 1988, Smith 1989). Rogowski et al. (2009) conducted a stable isotope study and showed that bighead carp fed higher up the plankton food chain than silver carp, which tend to eat more phytoplankton. Zhou et al. (2009) performed a stable isotope study in a hypereutrophic bay in Lake Taihu, China, where the fish were held in a large pen, and showed that the two species occupied the same trophic level and consumed very similar amounts of phytoplankton and zooplankton. In a study conducted to examine the influence of fish densities on diet in these two fishes, also using large pens, Ke et al. (2008) found that under low densities, both species consumed more zooplankton, but silver carp always included more phytoplankton than bighead carp. When fish densities increased, the diet breadth of bighead carp increased and they consumed more phytoplankton.

Studies have investigated the possible competition among native and non-native planktivores. Irons et al. (2007) looked at the correlation between bigheaded carp abundance (measured by commercial catch) and the abundance and condition of bigmouth buffalo and gizzard shad in the Illinois River. Here bigheaded carp densities comprise up to 63% of the total fish biomass and silver carp densities are believed to be the highest in the world (Garvey et al. 2012 and

(top) A bigmouth buffalo. (bottom) A gizzard shad. *Credit DNR SHP.*

Sass et al. 2010). These two native species, also filter feeding planktivores, were studied as indicators of a competitive effect and reduced fitness caused by high densities of bigheaded carps. Declines in gizzard shad and bigmouth buffalo condition (5-7%) were significantly correlated with increased commercial harvest of bigheaded carps and poorly correlated with temperature, chlorophyll *a* and discharge (variables also thought to influence body condition).

A similar study was reported subsequently in the same river system by Garvey et al. (2012). Garvey et al. (2012) tracked trends of catches for a number of native species in addition to trends in the abundance of silver and bighead carp, which became established around 2000. Over the period of 1994 to 2010 declines were observed in native bigmouth buffalo, white bass, freshwater drum, sauger, black crappie, and another aquatic introduced species - common carp. However, they concluded that none of these declines could be directly attributed to bigheaded carps since the population declines began prior to their arrival.

For competition between individuals to occur there must be a) a limiting resource, b) the organisms in question must share a common need for that resource, and c) there must be a negative effect on growth or some other measure of fitness (Crowder 1990). Bigheaded carps can affect plankton size and species composition in certain conditions, which could create a limiting resource for the native community. Garvey et al. 2012, cited above, observed a shift in Mississippi River zooplankton composition from cladocerans and copepods to rotifers. Total zooplankton abundance was positively correlated with bigheaded carp densities as the plankton community shifted to smaller bodied zooplankton. In contrast, a study of 96 lakes in China found that the size distribution of cladocerans and copepods shifted to larger bodied species in lakes with bigheaded carps (Zhang et al. 2013). They attributed this shift towards larger zooplankton to the consumption of cyanobacteria by bigheaded carps; cyanobacteria are avoided and can have deleterious effects on grazing by large-bodied zooplankton and can limit the growth of their main food source - green algae.

Fishes that are planktivorous throughout their lives are of special concern for negative interactions with bigheaded carps. Paddlefish, which are also large filter feeders inhabiting the Mississippi River basin, were thought to represent one important species that could be negatively affected by the presence of bigheaded carps, especially in light of recent population declines (Tucker 1996, Pflieger 1997; Schrank et al. 2003). Schrank et al. (2003) documented negative impacts to relative growth of paddlefish held in experimental ponds with bighead carp, which suggested that bighead carp have the potential to negatively affect the growth of paddlefish when food resources are limited. In contrast, in studies of actual river systems, Sampson et al. (2009) looked for diet overlap among bighead and silver carp and paddlefish,

Paddlefish - threatened in MN & WI, special concern in ND, extirpated in Canada. *Credit Dave Helms.*

gizzard shad and bigmouth buffalo in backwater lakes on the Illinois and Mississippi rivers. Very little overlap was found with paddlefish. Paddlefish consume primarily zooplankton along with various aquatic insect larvae and occasionally feed near the bottom (Wagner 1908, Coker 1930). Freedman et al. (2012) used stable isotope analysis to examine food webs of high and low bigheaded carp density sections of the Illinois River. Their results suggested that high trophic overlap exists between carps and bluegill, emerald shiner, and gizzard shad, which are facultative planktivores (prefer plankton but will eat other foods such as benthic invertebrates). Less overlap and apparent competition was found between bigheaded carps and filter feeding planktivores -bigmouth buffalo and paddlefish. It is important to recognize that many of these river studies were done on impounded systems with elevated plankton densities.

Paddlefish are known to have been much more abundant prior to the impoundment and eutrophication of the Mississippi River. Consequently, it does not necessarily follow that increased planktivory by bigheaded carps causes decline of native planktivores; other factors (fragmentation, habitat loss, commercial harvest etc.) may be more important in defining abundance of native planktivores.

Currently, it is not known (a) if plankton resources in Minnesota's large rivers are limiting for planktivorous fishes, (b) if the bigheaded carps could cause resources to become limited, or (c) at what threshold of abundance plankton becomes limiting. To further complicate things, bigheaded carps may affect trophic dynamics in unpredictable ways – some of which may favor certain native species while negatively affecting others (Kolar et al. 2005). For instance, spawning bigheaded carps can produce large quantities of eggs and fry that may provide food resources for native planktivorous fish.

Population Level

Bigheaded carps are now well established in the middle reaches of the Mississippi River, making extirpation unlikely (Williamson and Garvey 2005). Bighead and silver carps exhibit strong schooling behavior and can travel great distances in the mid-Mississippi River, particularly during higher flows (DeGrandchamp et al. 2008). There is similarity in habitat selection between bighead and silver carps, suggesting that they co-exist by partitioning resources other than space (DeGrandchamp et al. 2008).

Successful recruitment is a key element of establishing a competitive advantage. Bigheaded carps are very fecund and can reproduce at a young age. Bigheaded carps became established in the Illinois River in 2000, after which their population grew rapidly (Garvey et. al. 2012). Hoff et al. (2011) developed a Ricker stock-recruitment model using bighead carp population data collected in the LaGrange Reach of the Illinois River and in Pool 26 of the Mississippi River. The functional relationship of the model explained 83% of the observed recruitment variation. In the LaGrange Reach of Illinois River and Pool 26 of Mississippi River, stock size accounted for 72% of the variation in recruitment and river discharge an additional 11% of the variation. Assuming conditions are similar to those where the data was collected, an increase in stock size would show rapid increases in recruitment and establishment of bighead carp. Hoff et al. (2011) results suggested that control of populations of bighead carp in the LaGrange Reach and Pool 26 should focus on reducing stock size abundance through harvest or other means of suppressing adult abundance. Further, the authors concluded that, based on their modelling, low discharge variability resulted in higher recruitment. Thus stable hydrology (by impoundment) and flow regulation favor bigheaded carp reproduction.

Silver, bighead and grass carp are pelagic spawners laying eggs in turbulent currents where they become semi-buoyant and require adequate current to keep them suspended as they absorb water. Bighead and silver carp larvae in the drift

Spawning silver carp near Havanna, IL in the Illinois River. *Credit Prairie Rivers Network.*

that are younger than the gas bladder stage do not appear to strongly avoid capture by nets or siphon tubes; they probably have poor predator avoidance strategies (Duane Chapman, USGS, personal communications 2013, cited within Cudmore et al. 2012). Older larvae exhibit horizontal swimming behavior (swimming towards river margins and backwater habitat) and have more ability to avoid pelagic ichthyoplanktivores. Studies of predation, by fishes or birds on bighead and silver carp in North American waters, for any of their life stages, have not been conducted (Kipp et al. 2011). The reproductive success of all three species has been associated with large rivers. Understanding of the length of flowing water required for egg development has been changing. Recent data suggest that 15 miles may be adequate for silver carp, and some reproductive success has been documented in reservoirs with small watersheds (Murphy and Jackson 2013, Tang 1960).

Although high fecundity and migratory behaviors make bigheaded carps resilient to high adult mortality (Garvey et al. 2012), predation on other life stages may be an important mechanism for population control in aquatic environments with high diversity of native fishes. Eggs floating in the current, fry swimming vertically in the water column, then horizontally to backwaters, and juveniles moving in schools, all can be vulnerable to a wide range of species and predators in Minnesota's river basins. Even adults, though large bodied, may be susceptible to predators such as the flathead catfish, until they reach extreme size classes.

Community Level

Successful establishment of introduced fishes varies widely between geographic regions (38-77%) (Ross 1991). However, aquatic communities that are depauperate in fish species or altered by man are most vulnerable to such invasion (Ross 1991). A number of studies showing significant community-environment association have stated that abiotic factors (e.g., physical/chemical) and recolonization dynamics appear to be more important in structuring stream fish assemblages than biotic interactions (Schlosser 1982; Poff and Allan 1995; Taylor 1997). Conversely, experimental and field studies at small scales show the importance of local competition among stream fishes, with large scale studies emphasizing abiotic controls (Jackson et al. 2001, for a review). This suggests that in the larger size rivers, that bigheaded carps are thought to prefer, fish communities may likely be driven by abiotic controls (i.e., physical and chemical suitability). For instance, clear, clean water in streams with naturally variable flows supports one fish assemblage, while an impounded, eutrophic stream favors another. The fish assemblage of the impounded and eutrophic stream tends to favor tolerant species, which can include introduced species.

What are mechanisms defining competition between bigheaded carps and native fishes? Their consumption of plankton, and resulting alteration of food webs (trophic alteration), is considered a primary category of negative effects for introduced bighead and silver carps. In terms of habitat and water quality, silver and bighead carp are tolerant species that can thrive in sewage lagoons (FAO 1984, Sin and Chiu 1987). Juvenile silver carp have been found to grow a vascularized extension of their lower jaw in response to low dissolved oxygen enabling them to respirate at the water interface where more oxygen is available in otherwise anoxic waters (Amberg et al. 2012). The ability to exist in anoxic waters can also provide juveniles with refuge from most predators (Chapman, personal communications 2013), and a competitive advantage over native fish in degraded systems. Finally, bigheaded carps can extract energy from cyanobacteria; a plentiful resource in impounded rivers that is not well utilized by native fishes or zooplankton.

Ecosystem Level

Large-bodied species, such as bigheaded carps, can induce new biological interactions between native species of prey, competitors, and predators (Cucherousset et al. 2012). These new interactions arise from direct (e.g., competition and predation) and indirect (e.g., trophic cascade) effects that

Winterkill (mostly common carp) on eutrophic Lake Shetek, MN caused by low dissolved oxygen. *Credit DNR Fisheries.*

can destabilize native communities and food webs (Baxter et al. 2004; Lockwood et al. 2007). However, differentiating between the effects of introduced fish and other environmental changes induced by human activities on aquatic ecosystems is difficult (Dudgeon 2006; Cucherousset and Olden 2011).

Like other carps, common carp are not top predators but have been associated or correlated with changes to other freshwater species by uprooting vegetation, re-suspending sediments and nutrients, increasing turbidity, inhibiting growth of submerged macrophytes and shifting shallow lakes from a clear-water to a turbid state (Weber and Brown 2011). However, there has been long running debate about the relative extent to which these effects are attributable to common carp or to concurrent land use, increased nutrient loading, habitat alteration, and impoundment (Becker 1983). Surber (1923) concluded that declines in the aquatic vegetation and water quality of Lake Shetek, Minnesota, that were widely attributed to common carp, were largely due to a dam at the lake's outlet that raised water levels and caused massive shoreline erosion and sedimentation.

Extreme densities of common carp are often symptomatic of a decline of overall environmental health, but may also be a roadblock to improvement (Penne and Pierce 2008; Wahl 2001; Schrage and Downing 2004). Robel (1961) and Crivelli (1983) found very high carp densities at or above 675 kg/ha (600 pounds per acre) negatively influenced aquatic plant growth.

Tolerant native fishes can also reach high densities in eutrophic systems and can have similar effects on vegetation and trophic states of lakes, further complicating attribution of effects directly to common carp. In enclosure studies, native black bullheads, as well as common carp, have been shown to uproot aquatic plants, and while neither species increased turbidity directly, their behavior resulted in increased suspension by wave action due to the absence of vegetation (Berry et al. 1990). Similarly, planktivorous species such as bigmouth buffalo and fathead minnows can graze down large zooplankton and reduce grazing pressure on phytoplankton, which increases phytoplankton densities and turbidity and inhibits growth of aquatic macrophytes (Hanson and Butler 1990).

Although high densities of common carp and tolerant native planktivorous and benthic species can have significant effects on submergent vegetation, these effects are strongly facilitated by eutrophication, habitat alteration, and impoundment. These human impacts all have direct effects on water quality in addition to promoting the extreme abundance of carp. Common carp are widespread in Minnesota and the Midwest, but do not seem to cause significant problems in relatively pristine clear-water watersheds (Becker 1983).

Examples do exist from previous experience with introduced (common) carp that can help elucidate the mechanisms at play. Lake Christina, a historically exceptional waterfowl lake in West Central Minnesota (Figure 11), declined in waterfowl production in the 1950s due to increased turbidity and loss of submergent vegetation. The lake was treated with rotenone, a fish toxicant, in 1965, 1987, and 2003 to eliminate fish that were assumed to be the cause of the turbid condition. These treatments resulted in short-term shifts to clear-water states followed by reversion to a turbid state. Fish surveys prior to rotenone treatment in 2003 found a low abundance of piscivorous fish and common carp standing stock was estimated at only 1 kg/ha. Post-treatment abundance of carp actually increased due to a higher reproductive success (Figure 12) Meanwhile black bullheads comprised the highest biomass (16 kg/ha) and fathead minnows the greatest abundance (Graeb et al. 2004).

A later study by Hobbs et al. (2012) involved a paleolimnology analysis of lake sediment cores to determine that the lake was in a stable clear-water state from approximately 1750 to 1946 followed by a turbid state starting around 1950. They concluded that a dam built in 1936 at the outlet of connected Pelican Lake (that roughly doubled the depth of Lake Christina) coupled with a wetter climate beginning in the 1940s and eutrophication triggered the shift to a turbid state by increasing densities of native planktivorous fish. Turbidity, due to phytoplankton increased, because the planktivorous fish reduced the abundance of large zooplankton, which reduced the grazing pressure on phytoplankton.

Archeological surveys by Mulholland et al. (2011) of campsites around the Lake Christina showed that a diverse fish assemblage was historically present prior to 1880. The historic fish assemblage included several species -freshwater drum, channel catfish, and smallmouth buffalo - that have since been extirpated from the lake and the entire Pomme de Terre watershed upstream of Morris Dam. Exclusion of piscivorous fish from Lake Christina by an electric barrier built in 1989, the outlet dam, a carp barrier

Figure 11. Lake Christina, Pelican Lake, Pomme de Terre River and location of dams (red squares). Morris Dam (not shown) is downstream to the south. Blue arrows indicate direction of flow.

dam, and other downstream dams may favor early planktivorous life stages that would increase grazing pressure on larger zooplankton.

It has been shown in wetlands that turbidity is reduced when piscivorous fish, such as walleye, reduce fathead minnow abundance which results in lower predation on and increased abundance of large zooplankton that graze on phytoplankton (Herwig et al. 2004). It follows then that a barrier-induced reduction in piscivores may cause the opposite effect – increased turbidity due to higher abundance of phytoplankton (Figure 13).

Leprieur et al. (2008) tested three hypotheses, as to whether fish invasions in the world's river systems were caused by 'human activity', 'biotic resistance', or 'biotic acceptance' and reported a global map of fish invasions (i.e., the number of non-native fish per river basin). Their work showed that the human activity indicators (such as GDP, population density, and percentage of urban area) of the world's river basins were positively related to the number of established introduced fish species. In addition, they accounted for most of the global variation in introduced species richness – giving support for the 'human activity' hypothesis. They highlighted that the level of economic activity of a given river basin strongly determines its vulnerability to invasion. They "show that the biogeography of fish invasions matches the geography of human impact at the global scale, which means that natural processes are blurred by human activities in driving fish invasions in the world's river systems." Evidence of the negative

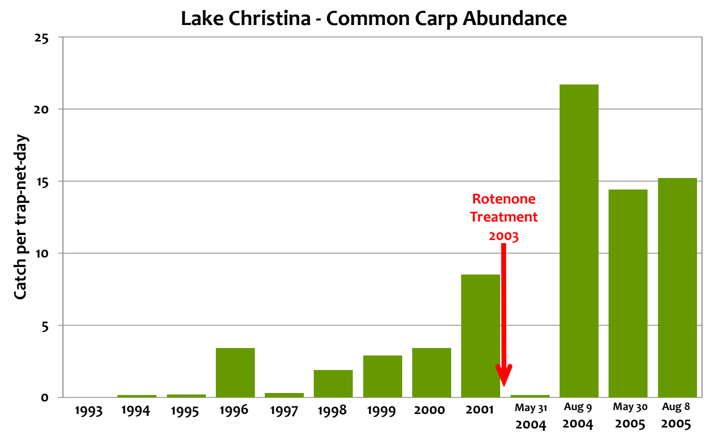


Figure 12. The effect of rotenone treatment on carp abundance and reproduction in Lake Christina.

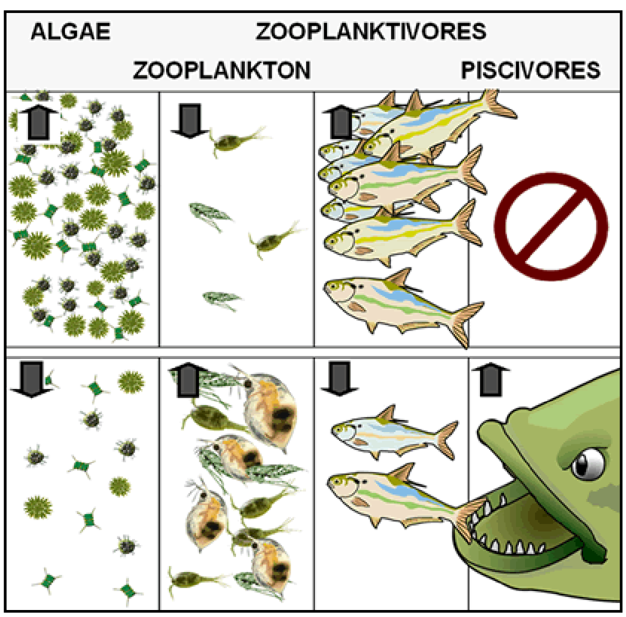


Figure 13. The cascading effects of piscivores on lower trophic levels. *Credit Tony Thorpe, Lakes of Missouri Volunteer Program.*

synergy from fragmentation by reservoirs, land use, and hydrologic changes and invasive species and fish community shifts has been found. Gido et al. (2010) found that fragmentation from dams, increased sediment supply derived from row-crop agriculture, and reduced discharge from groundwater withdrawal was related to fish community shifts in three Great Plains basins. While such work may be too coarse for determining the non-native potential and eventual impacts of introduced carps in Minnesota, it strongly suggests that the causal mechanism of fish invasions is altered ecosystems.

Additional insights may be forthcoming from novel research. Stable isotope analysis is an emerging approach which may be a powerful tool in determining the ecological effects and underlying mechanisms of introduced fish species on recipient ecosystems (Cucherousset et al. 2012). Other research, looking at the spatial occupancy of species, the movement of individuals, and their seasonal use of particular habitats using telemetry may be especially important to understanding introduced carps in the context of native assemblages.

Identifying Causality in Degraded Ecosystems

Scientifically documenting and quantifying community and ecological changes due to introduced species is full of challenges. The lack of knowledge about ecology of fishes or plankton communities in large river ecosystems, in addition to co-varying factors such as changing hydrology, water temperatures, flow rates, abundances of other biota, and human activities, all confound efforts to document the direct effects of introduced bigheaded carps on native fishes.

The lack of knowledge about ecology of fishes or plankton communities in large river ecosystems, in addition to co-varying factors, confound efforts to document the direct effects of introduced bigheaded carps on native fishes.

At this time, very little has been reported which definitively shows that bigheaded carps have had a negative, direct impact on native fishes in North America. Relationships between the extirpations and extinctions of native fishes and the introductions of bigheaded carps have been suggested elsewhere (Kumar 2000); however, fragmentation (dam and levee construction), severe water pollution, direct removal of native species, water removal and other factors were associated with these cases.

An algal bloom in Donghu Lake in August 2009, note dead fish in foreground. *Credit China.org.cn.*

For example, introduction of bigheaded carps in Donghu Lake, China have been associated with the "virtual disappearance of all the 60 species of fish native to the lake" (Kumar 2000), but when put in context of concurrent alterations and pollution, the causal associations become questionable.

- First, Donghu Lake was historically a backwater lake connected to the Yangzte River separated only recently by dikes. Bigheaded carp are native to the Yangzte River and therefore would have been native to Donghu Lake.
- The lake was (starting in the 1950s) and still is heavily polluted with untreated sewage, heavy metals, organic pollutants, and non-point runoff (Wang et al. 2002).
- Grass, silver, and bighead carp were heavily stocked, beginning in 1973, and the lake was declared a fish culture pond. No natural reproduction of these species occurs in the lake.
- The lake was divided by damming the coves for production of millions of fingerlings.
- Predatory fish were intensively removed by seining in areas where they spawned.

Attributing extirpations of native species on bigheaded carps that were native to the watershed is incongruous given the pollution, fragmentation and other issues in Donghu Lake. Like Donghu Lake, introductions of bigheaded carp elsewhere have generally been done for purposes of fish culture and production, with very high densities maintained by annual stocking. Consequently, effects of ongoing heavy stocking of a species are likely to be greater than those associated with natural reproduction.

Many of the examples cited by Kolar et al. 2007 and Kumar 2000 are from reported effects on native species in reservoirs. Since fragmentation by dams and associated habitat alteration is widely recognized as prominent in the extirpation of native species, the causative role of introduced bigheaded carps is obscure.

High biomass proportions comprised of bigheaded carps does not necessarily indicate a commensurate decline in the biomass of native species. Arthur et al. (2010) made paired comparisons at a total of 46 wetlands where bighead carp and tilapia were stocked, to similar wetlands where no introduced species were stocked in the Mekong region in Southeast Asia. Total biomass in the stocked wetlands averaged 180% higher than that in the wetlands where these species were absent, but no significant differences in native biomass, species richness, diversity indices, species composition, or feeding guild composition was observed. The authors concluded that the limited effects may be due to low niche overlap and the ability of the introduced species to utilize blue-green algae (cyanobacteria).

Most research on inter-specific interactions between bigheaded carps and other species has examined predation of bigheaded carps on plankton and potential competitive effects through diet overlap between bigheaded carps and other fish species (Kipp et al. 2011). Kolar et al. (2005, 2007) have been cited as a source for bighead and silver carps likely competition with the young of native fishes and with all stages of native planktivores. However, actual field research into diet overlap and species abundances over time has shown a more complex picture: gizzard shad and bigmouth buffalo showed a decrease in condition (Irons et al. 2007), but abundance declines in these and 4 other fishes could not be attributed directly to bigheaded carps, as their declines started before the bigheaded carps showed up (Garvey et al. 2012). Diet overlap was examined between bigheaded carps and paddlefish as well; Sampson et al. (2009) and Freedman et al. (2012) were able to document little overlap between these species.

Establishing impact of one species on others definitively is an inherently difficult research task, and some of the challenge may reflect the distinction between invasiveness and impact. Ricciardi and Cohen (2007) tested the relationship between the invasiveness (capability to spread via rapid colonization) of a species and its impact on native biodiversity and found no evidence, in general, for this association. In this context, bigheaded carps are clearly invasive in their ability to colonize, particularly degraded and impounded habitat, but definitive causal impacts to native species are unsubstantiated in free-flowing systems.

Potential Range in Minnesota

A major factor in the invasive success of bigheaded carps appears to be environmental context or abiotic factors. Cudmore and Mandrak (2011) assessed the biological risk of Asian carps (grass, bighead, silver, and black) to Canada using an expert workshop approach. They concluded, in part, that the risk of impact was high in some parts of Canada, including the southern Great Lakes basin for all four species of Asian carps considered. Maps were provided within the article, which showed that environmental suitability for all four Asian carps was 100% for all of Minnesota, and for most of the Mississippi River basin. Cudmore and Mandrak (2011) also concluded that the consequences of establishment of grass and silver carps in Canada, based on the ecology of native systems, are high with high certainty. This study used broad-scale factors like air temperature and river length greater than 50 km to predict likelihood and consequence of bigheaded carps establishment. The degree of eutrophy, fragmentation, and impoundment were not assessed by this study.

Abiotic factors appear to be affecting the invasive success of bigheaded carps in Minnesota. Silver carp were found in the wild within the Mississippi River beginning in 1974-75. Since they still have not established in Minnesota waters, this suggests that some mechanism may be at play that is dampening the spread of bigheaded carps in the Mississippi River basin, or has kept their spread to selective conditions over the past 40 years. For example, silver carp are present in the James River in North and South Dakota, a tributary of the Missouri River, which then flows more than 650 miles downstream to connect to the Mississippi at St. Louis, MO. The James River is almost entirely impounded, eutrophic, and often hypoxic in its lower stretches.

Bigheaded carp still have not established in Minnesota waters, suggesting that some mechanism may be at play that is dampening their spread in the Mississippi River basin.

The science that has been conducted does not support a straight-forward view of the over-whelming perception of the impact of the bigheaded carps on native ecosystems; rather, it seems conditional – based on the presence of conditions favorable to their adaptations and on the researcher's assessment of the prevalence of those conditions. Research on their potential establishment and impact in the Great Lakes provides an example of different conclusions. Cooke and Hill (2010) conducted a bioenergetics model to assess the theoretical potential of bighead and silver carp to colonize habitats in the Laurentian Great Lakes, based on plankton biomass and surface water temperature data. Their modelling results indicated that the low concentrations of plankton in many open-water regions of the Laurentian Great Lakes cannot support growth of silver and bighead carp. However, their results also suggested that in some habitats (such as Green Bay, the western basin of Lake Erie, and some other embayments and wetlands) plankton resources are sufficient to support positive growth of bighead and silver carp, even when taking into account higher swimming costs.

Recent spatially comprehensive studies show that low plankton biomass is prevalent in both near-shore and offshore regions of Lake Michigan (Vanderploeg et al. 2007). Cooke and Hill (2010) state: "If Asian carp were to enter the 'plankton desert' of Lake Michigan via the Chicago Sanitary and Ship Canal, it seems unlikely (but not impossible) that they would be able to derive enough energy from the plankton to support the energetic costs of travelling to Green Bay or another 'plankton oasis'." Cudmore et al. (2012) stated that there was enough food for bighead and silver carp survival in the Great Lakes, especially in Lake Erie and productive embayments in the other lakes. They determined that there were many tributaries suitable for bigheaded carps spawning and should they become established, that their spread would not be limited. Among the ecological consequences, Cudmore et al. (2012) included competition for planktonic food leading to reduced growth rates, recruitment, and abundance of planktivores. Further, they stated that this would lead to reduced stocks of piscivores and abundance of fishes with pelagic early life stages. Cudmore et al. (2012) conclude that the overall risk of bigheaded carps to the Great Lakes is highest for lakes Michigan, Huron, and Erie, followed by Lake Ontario, then Lake Superior.

Competitve Advantages and Ecosystem Health

While introduced species are unlikely to become prominent in systems where environmental conditions are unfavorable for their survival, a major focus of the literature on invasive resistance centers on biotic resistance associated with predation and competition (Li and Moyle 1981; Case 1991; Lodge 1993). Competition is presumed to be a means by which introduced species can displace native species (Byers 2000; Juliano 1998). But these findings raise a basic question:

> *Does an introduced species in a novel environment have a competitive advantage over resident species, which have been evolving to that environment for thousands of years?*

The concept of competitive advantages inherently favoring native species is widespread and commonly acknowledged in the ecological literature (Case 1991; Massot et al. 1994). Byers (2002) argues that the answer stems from rapid anthropogenic alteration of selection regimes – eutrophication and the selective removal of top predators. Changes in the environment reverse outcomes of competitive interactions among species. Anthropogenic disturbances, like dam construction, may so alter environments that it is the native species that find themselves in a novel environment. Extreme disturbances may erase a native species' prior advantage of local environmental adaptation that it has accrued over time (Byers 2002). Successful invasions in aquatic systems are most likely to occur when native assemblages of organisms have been temporarily disrupted or depleted (Moyle and Light 1996).

Anthropogenic disturbances, like dam construction, may so alter environments that it is the native species that find themselves in a novel environment.

When the impacts of bigheaded carps are placed within the context of the ecosystem, its current condition and degree of alteration, a recurring pattern emerges - these fish do well in fragmented, nutrient-rich river systems. Impoundments and eutrophic water quality are the notable environmental conditions where these fish have established themselves in abundance. Of course, for management, knowing the threshold, the precise degree of alteration, dis-connectivity, and degradation, that allows bigheaded carps to populate and overwhelm the native community, are important parameters. Competition with native fishes over plankton is reported for bighead and silver carps (Kolar et al. 2005, 2007), but studies in the Illinois and Mississippi rivers report few definitive impacts to native species .

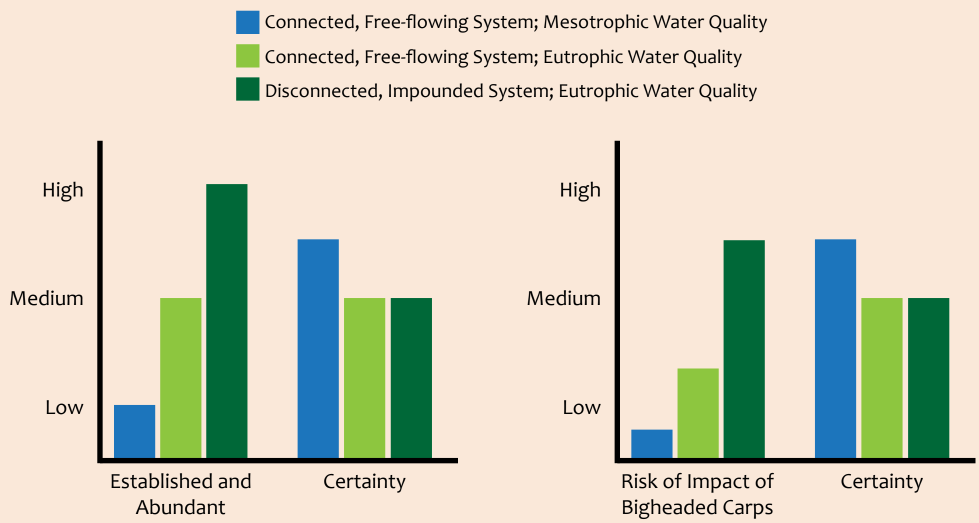
When the impacts of bigheaded carps are placed within the context of the ecosystem, its current condition and degree of alteration, a recurring pattern emerges: these fish do well in fragmented, nutrient-rich river systems.

Summary:

Scientifically documenting and quantifying community and ecological changes due specifically to introduced species is full of challenges. Most research on interspecific interactions between bigheaded carps and other planktivorous species has evaluated competitive effects through diet overlap. *In situ* studies of diet overlap and species abundances over time show a complex picture highly dependent on the environmental condition. Consumption of cyanobacteria, including toxic *Microcystis,* by bigheaded carps is a unique adaptation. Since native species are not able to digest cyanobacteria, bigheaded carps do not compete with them for this resource.

Introduced carp do not appear to be restricted by the hydrologic and temperature regime in Minnesota, as they are present in the James River, North Dakota, a tributary to the Missouri River. However, they may not be able to sustain the abundances reached in more southern waters due to winter mortality and other climatic factors.

Based on our review of the available literature, the risk potential of introduced carp impacts on native fishes depends on the environmental conditions present. The case evidence suggests that as water quality deteriorates and fragmentation increases the risk of bigheaded carp success also increases. It is unclear whether this high risk of effects on native species is primarily due to the bigheaded carps or due to the degraded water quality and fragmentation.



A qualitative characterization of risk and certainty of bigheaded carps becoming abundant in Minnesota waters.

A qualitative characterization of risk and certainty for bigheaded carp effects on native fishes in Minnesota.

Question 3. How Effective are Barriers for Introduced Carps?

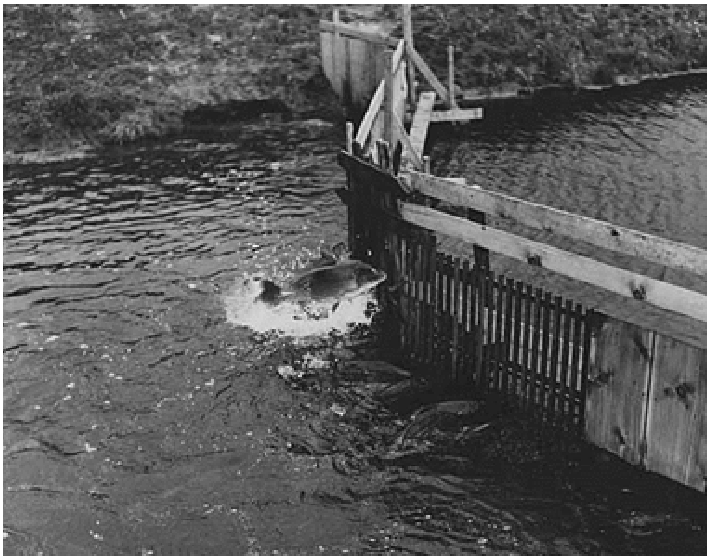
Overview: A key consideration for the use of barriers is their effectiveness. Examination of fish hydrodynamics for bigheaded carps in particular shows these fish to be highly adept swimmers and jumpers (silver carp) making their blockage more challenging. Examples from existing barriers in Minnesota designed to block common carp and in other states to block other introduced carps provide evidence for likely outcomes for their use in Minnesota.

This chapter outlines the swimming capabilities of carp, examines case examples of barriers and their effectiveness, and provides additional information on issues related to the efficiency of barriers.

Over the long-term, barriers have generally been shown to be ineffective in controlling the spread of introduced carps, particularly in eutrophic watersheds due to alternate pathways, flood related inefficiencies, power outages, structural failure, and other problems.

Barriers have been used to exclude and control introduced common carp in Minnesota since at least 1927 (Hoffbeck 2001). These have included a variety of designs including screens, high velocity culverts, dams, and electric barriers. Various physical barriers have been used throughout the United States to control the spread of introduced fish, including dams, electric fields, bubble and sound fences, screens, and high velocity culverts.

The effectiveness of barriers will depend on fish hydrodynamics as well as physical factors that define the reliability of the barrier.



A fish screen on Six Mile Creek near Lake Minnetonka in 1965. *Credit Minnesota Historical Society.*

Fish Hydrodynamics & Implications for Carp

The effectiveness of physical barriers for a given species is a function of (a) that species' hydrodynamics including swimming speed capabilities, body size and physiology, swimming behavior, and jumping capabilities and (b) stream hydrology and other factors related to the continuity and effectiveness of the barrier.

Swimming speed has typically been measured in laboratory flumes where prolonged and burst speeds can be quantified. There are several methodological issues that must be considered with these measurements and how they are interpreted. **First**, the flumes are usually different, both physically and hydraulically, than a natural stream or an "*in situ*" barrier. Flumes are often made of smooth materials to yield a laminar (uniform) flow pattern. Rivers, on the other hand, have complex, often turbulent flow patterns due to bed and boundary roughness, debris, and vegetation. River velocities and velocity distributions change during floods, ice cover, ice and debris flow, and backwater effects. Fish have adapted to these complexities and can use the lowest velocity areas of the cross-section to effectively navigate through high velocity areas, usually near the stream bed or banks. Natural flumes and physical models have been used for some studies of fish passage and are likely to provide more realistic results. **Second**, fish in laboratory studies are often stressed and may not be motivated to burst at their physiological potential especially when compared to those exhibiting their full potential during spawning migrations.

Brown trout jumping attempts over a falls. *Credit Gary Cooley.*

In situ observations of fish hydrodynamics have their own complications but provide insights that are difficult to match in a laboratory. Field data for over 150,000 individuals and over 100 species have been collected in Minnesota in addition to measurements of velocity gradients through which the fish are observed passing (Aadland and Kuitunen 2006). Additionally, underwater videos have provided frame by frame measurements of displacement by bursting fish. In many cases, these *in situ* measurements demonstrate passage though stream velocities that exceed those suggested by laboratory studies.

For species that jump, back-calculation of exit velocities necessary to reach a specific height (center of mass) at an optimal trajectory can be used as a minimum estimate of burst speed. Most fish do not jump at an optimal trajectory and a sub-optimal trajectory will result in a lower jump height, so many jumps will result in a conservative estimate of burst speed. Conversely, some thrust can be maintained until the caudal fin leaves the water and must be considered in these estimates.

Key Differences between Common and Bigheaded Carps

Common carp have been used as a surrogate to assess barrier effectiveness for silver and bighead carp (Stainbrook et al. 2005) but there are some notable differences between these species.

Burst speed

Common carp have been observed burst speeds from 6.6 feet per second (10 inch fish, Tudorache et al. 2007) to 14 feet per second (unknown length, Bell 1991) and have been reported jumping up to 3 feet (personal observations). A jump of 3 feet would require a burst of 14 feet per second at an optimal trajectory based on computations of momentum and gravitational acceleration.

Silver carp, on the other hand, have been observed jumping to as much as 10 feet (Duane Chapman, USGS, personal communications 2013), which would require an exit velocity of over 20 feet per second at an optimal trajectory. This relationship is described by:

$$\text{Maximum jump height} = V_o^2 / 2g$$

where:

V_o = takeoff velocity in ft/sec and

g = gravitational acceleration = 32.15 ft/sec^2.

Few native species can attain these burst speeds.

Location in Water Column

Silver and bighead carp are planktivorous species that move high in the water column while common carp are primarily benthic. Most electric barriers have electrodes embedded in the stream bed, which concentrates the field near the bed and yields a weaker field near the surface (Holliman 2010) as observed *in situ* by Verrill and Berry (1995).

Vulnerability to Electric Field

Small bodied fish tend to be less vulnerable to electric fields because their shorter body length is subject to less voltage differential from head to tail than longer fish in a given voltage gradient. Juvenile carp have been shown to be less vulnerable to a given field strength than adults (Reynolds 1983; Holliman 2010). In addition, silver and bighead carp have also been shown to disperse most actively during floods (DeGrandchamp et al. 2008) when barrier dams can be inundated and electric barriers may be less effective (Verrill and Berry 1995). Hence, juvenile carp are less vulnerable to the electric barriers because of their size and bigheaded carps are less vulnerable than common carp due to their tendency to migrate during spring high flows. While common carp also move during floods, data from the Breckenridge Fishway in West Central Minnesota showed common carp migrations peaking in late June and early July when flows are usually lower. Both bigheaded and common carp are likely to be more successful in passing lowhead dams and electric barriers during high flows.

The greater burst speed, jumping ability, and other attributes of silver carp suggest that barriers effective for common carp may not be effective for silver carp.

The greater burst speed, jumping ability, and other attributes of silver carp (tendency to migrate during spring high flows) suggest that barriers effective for common carp may not be effective silver carp.

Types of Barriers

Electric Barriers

Electric barriers are widely used for blocking introduced fish. However, electric barrier design must account for the species swimming and jumping abilities (e.g., jumping ability of silver carp), such that an adequate length of stream must be electrified to prevent passage.

Safety concerns to water recreationists exist with an electric barrier, and there are questions of long-term cost and efficacy in larger river systems, particularly those with high sediment, debris, and ice flows. An electric barrier has not been deployed on a larger river, such as the Minnesota River where a barrier has been proposed. Other practical considerations such as: width of the floodway (valley), back-up contingencies during power outages, the need for redundant systems during maintenance and to increase effectiveness, and very high energy and operation costs can make the final design complicated and expensive.

The following examples present the issues, concerns, and effectiveness of various electric barriers.

Illinois River An electrical barrier designed to repel fish was constructed in the Illinois River, which is connected to the Great Lakes via the Chicago Sanitary and Ship Canal. "*It is experimental and may not be 100% effective but remains the only defense against the upstream movement of bighead and silver carp from the Illinois River into the Great Lakes.*" (www.fws.gov/midwest/News/documents/AsianCarp.pdf). Of immediate concern is the perceived threat that two of these species (bighead carp and silver carp) pose to the Great Lakes. Due to the proximity of large populations of bighead and silver carp in the middle and lower segments of the Illinois River, the upper Illinois River (Waterway) and the Chicago Area Waterway System (CAWS) have been under intensive monitoring. Since 2009, this effort has been overseen by the Asian Carp Regional Coordinating Committee (ACRCC). In addition, the U.S. Army Corps of Engineers (USACE) has constructed a series of electric barriers on the Chicago Sanitary and Ship Canal outside Chicago, Illinois, in hopes of preventing the further spread of aquatic introduced species, such as bigheaded carps, between the Mississippi River Basin and the Great Lakes. The first barrier began providing electricity to the water in 2002 and to date, only one bighead carp has been found above the barrier in the waterway (in Lake Calumet, June 2010).

Minnesota All of the current 6 electric barriers in Minnesota were built as a means of controlling common carp in shallow waterfowl lakes. These lakes were historically important waterfowl lakes with prevalent submergent vegetation that shifted to turbid, blue-green algae dominated systems attributed to land-use changes, elevated water levels due to dams, and colonization by common carp. Three of these lakes have been monitored and are presented here.

South Heron Lake is a shallow 2,845 acre water body in Southern Minnesota that was historically an important waterfowl lake that became too turbid to support submergent vegetation. The lake's 462 mi^2 watershed is comprised of 93% agricultural lands, almost entirely corn and soybeans, and the lake is impaired for turbidity, phosphorus, and bacteria (Heron Lake Watershed District 2011). An electric barrier was built in 1991 on Lake Outlet Creek as a means of controlling carp and other fish species in the lake. Then in 1997-1998, the lake was poisoned with rotenone. Verrill and Berry (1995) did not catch any of 1,600 carp tagged downstream of the barrier in the lake but noted that 3 carp per hour were observed passing the electric barrier near the surface during high water. DNR Section of Fisheries surveys of the lake indicated 7.78, 10, 36.3 and 40 carp per gill-net-day in 1997, 1998, 1999, and 2011 respectively. The normal range for this lake type is 0.5 to 9.1 carp per gill-net-day (Section of Fisheries data). It is unknown whether the carp abundance in the lake was due to recolonization from the upstream watershed, carp surviving the rotenone treatment, or upstream migrants through the electric barrier.

Lake Christina is another historically important waterfowl lake in West Central Minnesota that declined following watershed changes and construction of an outlet dam that raised water levels about 2 feet (Figure 11, pg 34). The lake was treated with rotenone in 1967, 1987, and 2003 in an effort to eliminate carp and planktivorous fish that would in turn increase zooplankton densities, water clarity, and submergent macrophytes.

An electric fish barrier was built at the lake outlet in 1987 (Figure 14). Pre-treatment carp abundance in 2003 was actually low (<1 kg/ha) and below levels known to affect aquatic macrophyte production. Predator abundance was also low despite a 3 year stocking program and the fish community was dominated by small individuals that were likely planktivorous (Graeb et al. 2004). The low predator abundance and small size structure may be due to frequent winterkills and the electric barrier which would block migrations in and out of the lake. Case in point, Tonn and Magnuson (1983) found that northern pike maintained populations in winterkill lakes that had connections to oxygen refugia in streams.

Figure 14. (top) The electric barrier on Lake Christina. The screen is dammed up with lake debris in the spring. (bottom) A carp dam downstream of Pelican Lake on Pelican Creek. *Credit DNR SHP.*

Rotenone treatments did result in improvements in water clarity and growth of submergent vegetation for periods following treatments. Nevertheless, based on the 2005 Fisheries catch of 14.5 carp per trap-net-day (mostly age-1), neither the rotenone treatments nor the electric barrier were effective in controlling common carp for more than a short window of time. The normal range for this lake type was 0.7-5.1 carp per trap-net day.

Historic fish communities in Lake Christina included channel catfish, freshwater drum and several other species that were absent upstream of the Appleton Dam on the Pomme de Terre until its removal in 1998 and are currently present upstream to the next barrier, Morris Dam. This indicates that the lake had quality waterfowl habitat for many years in the presence of a diverse fish community.

Lake Maria is a shallow headwater lake in Central Minnesota that, like Christina and South Heron lakes, had lost its submergent vegetation due to heavy growth of blue-green algae. Like Christina, its water levels had been raised and stabilized by an outlet dam. In 2007 it was equipped with an electric barrier as an effort to eradicate and control carp, and reestablish aquatic macrophytes. Also, the downstream ditch was treated with rotenone. Unlike South Heron Lake, the small watershed had no upstream lakes and the lake could be entirely pumped dry to assure elimination of all fish. The lake was refilled in 2008 and the project was successful in reestablishing vegetation and waterfowl habitat evident in 2012 aerial photos. However, carp and fathead minnows did reappear in the lake by 2010 presumably due to short duration power outage before the backup generator came on-line (Nicole Hansel-Welch, Shallow Lakes Coordinator, personal communications).

Power Outages Similar problems with electric barrier failures have been observed in other states. As a means of safeguarding against power outages, redundant barriers have been built on the Chicago Sanitary and Shipping canal connecting the Upper Mississippi and Great Lakes watersheds. To be effective against power outages, redundant barriers would need to have separate and independent power sources. Clarkson (2003) documented power outages due to lightening, manufacturing flaws, and human error that represented less than 0.001% of operation time on Central Arizona Project barriers, but concluded that significant passage of targeted non-indigenous species likely occurred and documented upstream sampling of grass carp following one of the failures. He also observed red shiner, an introduced species, passing one of the barriers while it was operating at design standards. Based on the reproductive capabilities of these species, he further concludes that barriers less than 100% effective are "ineffective."

Barriers less than 100% effective are "ineffective." (Clarkson 2003).

A pre-spawn walleye migrating through an area of dense bubbles in a rapids. *Credit DNR SHP.*

Bubble Curtains & Acoustic Barriers

Considering most potential barrier sites are on large expanses of rivers and the low efficiency rates of these technologies, we do not see these as technically viable options. While bubble curtains and acoustic fences may present social benefits, these options may be tenuous and should be weighed cautiously given the potential impacts to native species and the likelihood of eventual passage.

Migrating fish can actually be attracted by the sounds made by breaking bubbles. This is an adaptive advantage to navigating through turbulent rapids and riffles during migrations. Robert Newbury was able to attract migrating walleyes to an air compressor as a demonstration of this phenomenon (personal communication).

An acoustic barrier is currently being tested at Lock and Dam 8. This is intended as a selective barrier for bigheaded carp based on their higher sensitivity to certain frequencies. Effects on other species like catfish, that may also be sensitive to these frequencies, are not yet known.

Carp Screens and Velocity Barriers

Carp screen design can be effective for larger fish but may be ineffective for juveniles that can fit through gaps in screens (Figure 15). Furthermore, screens plug with debris and subsequently may not block carp that may be able to swim around due to culvert failure or bypass flows.

The effectiveness of velocity barriers (culverts, gates/openings, spillways) depends on adequate length and velocity. Tainter gates on the Mississippi

Figure 15. (top & middle) A fairly ubiquitous barrier on smaller streams and lake outlets are carp screens. These physical barriers often create high velocities which further impede fish passage. (bottom) Northern pike blocked by the barrier. *Credit DNR SHP.*

River dams, thought to be effective fish barriers, were shown to be routinely passed by silver carp when gates were largely closed, with a gap less than 1.2 m (Tripp et al. 2013). Mel Price Dam, one of the dams in the Tripp et al. study, has up to 24 feet of head, which would generate very high velocities through Tainter gates. Exact head loss and velocity during passage was not quantified in this study, but Tainter gates on Lock and Dam 7 have velocities of over 20 feet per second when head-loss is 6.5 feet (Corsi and Schuler 1995). Specific velocity computations would be a requirement for any management approach using velocities to prevent carp passage. Common carp velocity barriers are typically designed for significantly lower velocities, that would not be effective for containing bigheaded carps.

Dams

Numerous dams have been built as fish barriers over the years, most targeting common carp, but few have had assessments of effects on common carp or native species. Many of these have well established carp populations in the upstream watershed that are not dependent on immigration for population maintenance. Conversely, native species, including predators on carp such as northern pike, have declined in many of these watersheds because of their need to migrate.

The following Minnesota examples present the effectiveness of various dams as common carp barriers.

High Island Creek Dam High Island Creek is a tributary to the Minnesota River where a dam was built in 1958 specifically as a barrier to control common carp in the 241 mi^2 watershed (Figure 16). Surveys of the watershed have shown that while carp remain abundant upstream of the dam, 30 of the watershed's 47 native fish species do not

Figure 16. High Island Creek carp dam. *Credit DNR EWR.*

exist upstream of the dam. The effect of barriers on native species becomes a critical question if the barrier's intent is to protect these native species.

Drywood Creek Dam A dam was built on Drywood Creek, a tributary of the Pomme De Terre River, as a barrier to "undesirable fish species" in 1971 (Figure 17). An earlier dam had been present on the creek since the 1930s. Carp, black bullhead, fathead minnows and other tolerant species remained abundant in surveys upstream of Artichoke Lake. However, following the dam's failure in 2001 and removal of the Appleton Dam on the Pomme de Terre River in 1998, freshwater drum, channel catfish, shorthead redhorse, spottail shiner and quillback were present (in addition to 8 other species), all of which had been absent prior to the failure of the dam.

Thief River Falls Dam The 15-foot tall Thief River Falls Dam on the Red Lake River in Northwest Minnesota was the upstream limit of common carp until the early 1980s. Due to unknown causes, carp appeared upstream of the dam by 1988 and are now established.

Figure 17. (top) Drywood Creek, Drywood Creek Dam, and the Pomme de Terre River. Appleton Dam (not shown) is downstream to the south. (bottom) Remains of dam on Drywood Creek, 2013 aerial showing dam failure. *Credit DNR Fisheries.*

Knife River Dam A 14-foot dam was built on the Knife River in East Central Minnesota impounding Knife Lake in 1983. The lake was treated with rotenone in 1989 and subsequently restocked with most of the native fish assemblage with the exception of freshwater drum. Black bullhead, which were one of the target species, were not extirpated by the rotenone treatment and remain in the lake; however, common carp were effectively removed and have not yet reappeared in Knife Lake (Roger Hugill, Area Fisheries Manager, personal communications).

St. Anthony Falls Dam The 49-foot tall St. Anthony Falls Dam was originally built on a natural barrier in the 1872. The dam was a complete barrier until 1963 when it was rebuilt and equipped with a lock. Carp were not found above the dam until 1929 when they became established immediately upstream and subsequently extended their range above Coon Rapids, St. Cloud, Sartell, and Blanchard dams by 1962 (Eddy et al. 1963) (Figure 18). Presumably the carp were transferred with human help, either accidentally or intentionally. Following lock construction in 1963 and the resulting limited fish passage, 10 fish species and 11 mussel species native only downstream of the falls have now become established upstream of the lock and dam.

Summary These examples demonstrate that dams have generally not been effective at blocking the upstream movement of common carp, but have been effective at blocking the movement of native fish species. These four examples are included in an analysis discussed in *Question 4. What are the Effects of Barriers on Native Fish Species?*

Figure 18. Dams on the Mississippi River. Upper Mississippi Watershed highlighted in tan, Lower Mississippi Watershed in blue.

Issues Related to the Efficiency of Dams as Barriers

Flood Flows

The method used to block species must be compatible with local conditions, in that it has to work successfully under the entire range of flows, sediment, debris (including ice), and the temperatures (air and water) encountered.

The Minnesota River is a difficult river in that regard: its valley was formed by the runoff waters of the River Warren and is very wide. In addition, land use (primarily agricultural) and associated drainage in the basin has resulted in a very flashy hydrology, with flood flows that periodically inundate existing structures (dams, bridges, etc.) and beyond.

(top) Granite Falls Dam on the Minnesota River at low flow. (middle) Grantie Falls Dam at high flow. Note the remaining head across the dam face under high flows. (bottom) Flow around Granite Falls Dam in April 2011. Peak flow encountered was 23,700 cfs. In the flood of 1997, the peak flow was 33,900 cfs. *Credit DNR SHP and DNR Fisheries.*

Dam Failures

From 1985 to 1994, there were more than 400 dam failures in the United States, or about 40 per year (NRCS 2000). Minnesota has had dam and embankment failures regularly: Flandrau Dam failed 3 times before being removed in 1995, Breckenridge Dam failed 5 times before being modified in 2007, and Drywood Creek Dam (Figure 17) failed in 2001.

In the context of fish barriers, failure includes embankment failure that provides a short window of time when the dam is passable.

Selective Passage of the Upper Mississippi River Navigation System

The lock and dam system of the Upper Mississippi River (UMR) provides limited passage through the lock chambers. The Tainter gates are passable for most species during flood flows when the gates are open (gates completely out of the water, open to the flow of river so no or minimal head differential) and are passable only for fast swimming species when the gates are partially open (higher velocities

(top) The Minnesota River flooding around an 87-year old pedestrian bridge in Granite Falls. *Courtesy of Granite Falls Tribune.* (bottom) A bicyclist on Highway 22 south of St. Peter observed the floodwater from the nearby Minnesota River flowing across the roadway. *Credit DNR SHP.*

because flows are forced though an opening(s) with differential head loss) (Figure 19). Passage through the gates varies by dam depending on the hydraulic head of the structure and the frequency with which the dam operates in open gate condition.

Tripp et al. (2013) studied passage of tagged silver carp, bighead carp, paddlefish, shovelnose sturgeon, sauger, American eel, and white bass through Locks and Dams 20 through 27. Very little passage was observed through the locks. They found that silver carp moved through Tainter gates during "closed gate" (gate openings of around 0.6 – 1.2 meters) almost as frequently as "open gate" settings. Meanwhile native species predominantly moved during open gate (gates out of the water) conditions with lake sturgeon and paddlefish rarely passing through "closed" gates.

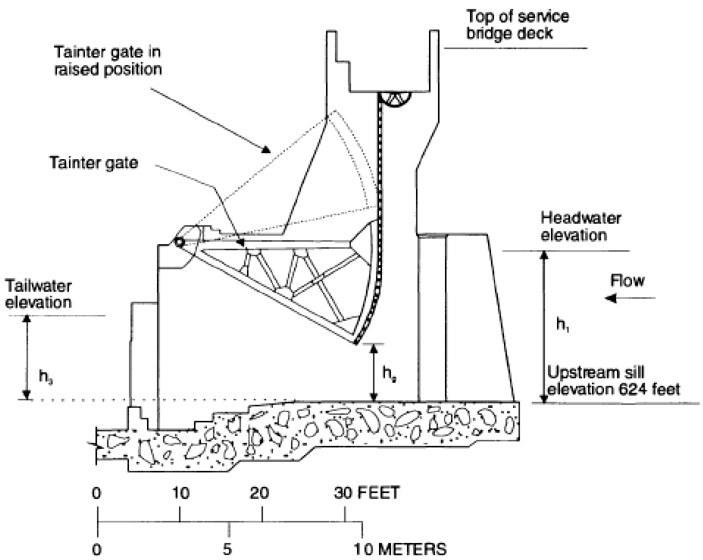


Figure 19. (top) A Lock and Dam on the Mississippi River with the Tainter gates partially open looking upstream. *Credit DNR SHP.* (bottom) Sectional view of a Tainter gate. *Courtesy of USACE.*

Fish passing through Lock and Dam 26 (24 feet of head) during closed gate conditions would be subjected to very high velocities. Exact head loss and velocity during passage was not quantified in the Tripp et al. study but, for perspective, Tainter gates on Lock and Dam 7 have velocities of over 20 feet per second with 6.5 feet of head (Corsi and Schuler 1995). Because of even higher velocities, no fish passed through the 36-foot high Lock and Dam 19. Silver and bighead carp did disperse through Dam 19, presumably through the locks. The greater swimming speeds of silver carp give them a competitive advantage over native species. In effect, the Mississippi River navigational dams provide selective passage for silver carp.

Barrier Site Location

Site selection can be critical, in terms of long-term costs, viability and success of a physical barrier.

On the Minnesota River, one of the narrowest portions of the channel and its valley is near Mankato (Figure 20). At the red line indicated on the LiDAR aerial, the river valley is >2,800 feet across, and the main channel is approximately 300 foot wide. Near Jordan, MN, the channel

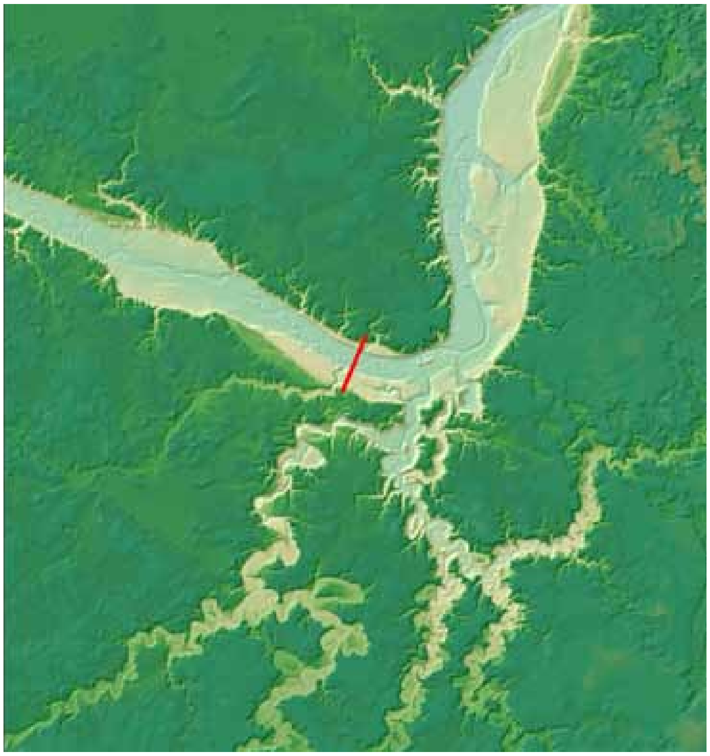


Figure 20. Hillshade LiDAR of Minnesota River at Mankato. Red Line represents a transect across the valley and river channel.

width is approximately 270 feet and the valley width is approximately 4,300 feet wide. Further upstream, past Mankato, the river valley widens to approximately 4,800 feet, while the channel is only 140 foot across. This configuration (wide channel within a very large valley) will make it difficult to ensure protection to upstream portions of the Minnesota River, especially during flood events.

Other Known Pathways of Aquatic Introduced Species

There are numerous potential pathways of introduction and spread of aquatic non-native species through human actions. Some have been intentional (i.e. stocking of carp and game fish, release from home aquaria) and many have been unintentional (i.e. escape of silver and grass carp from sewage lagoons or release of carp minnows mistaken for bait species). Considering silver carp are the most widely cultured fish in the world, and they quickly escaped captivity in North America, the potential for accidental or intentional release is high.

A national Asian Carp Working Group (Higbee and Glassner-Shwayder 2004; Kolar et al. 2007) identified 22 potential pathways related to the movement of introduced carps including:

- the transport and release of baitfishes caught in the wild;
- stocking Asian carps in private or public waters for biological control;
- the production, live transport, and live sales of Asian carps in seafood markets;
- live transport and intentional spread of Asian carps by commercial fishers;
- movement of Asian carps in ballast waters and live wells;
- intentional releases of Asian carps by consumers, hobbyists, and animal rights activists.

The USGS has identified thirteen introduction pathways for nonindigenous aquatic species, of which 68.5% are fish species, for the state of Minnesota (Figure 21).

These alternate pathways are illustrated by grass carp in Zumbro Lake reservoir and from a pond near Owatonna. Since these sites are not accessible from the Mississippi River, it is clear grass carp were introduced by people. Another example is St. Anthony Falls where common carp were present upstream of this natural barrier in 1929, before the lock that provides limited passage, was constructed in 1963.

These facts and examples demonstrate that even a complete barrier to fish passage does not assure that an introduced species will not become established upstream of the physical barrier.

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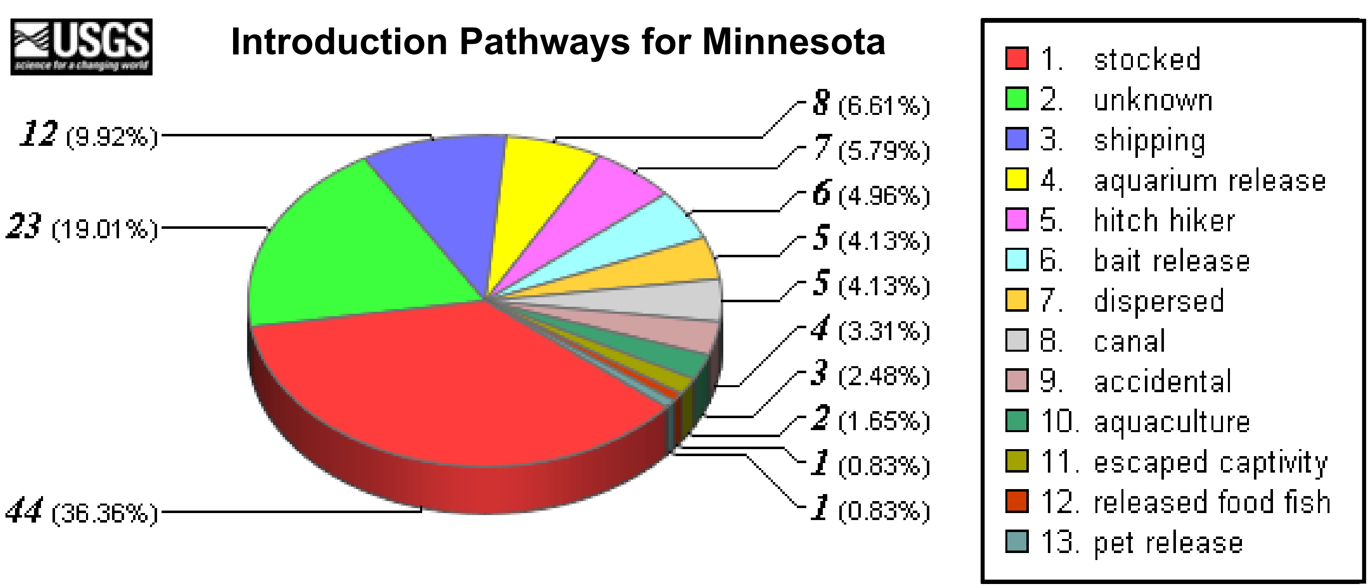


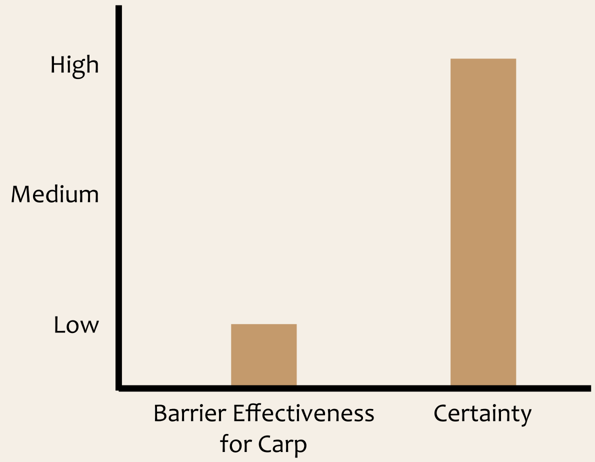
Figure 21. Introduction pathways for Nonindigenous Aquatic Species in Minnesota (68.5% are fishes). Data from USGS Nonindigenous Aquatic Species Database, 2014, http://nas.er.usgs.gov.

Summary:

Bigheaded carps are more adept swimmers than most native species; silver carp can jump as high as 10 feet and burst to around 25 feet per second. Because of these capabilities the bigheaded carps are the least likely to be impeded by physical barriers. Common carp are less proficient swimmers than bigheaded carps, but barriers targeting them can be used to help evaluate their effectiveness and likely ecological effects. Barriers that are not effective for common carp are unlikely to be effective for bigheaded carps.

Electric barriers, for which there were data in Minnesota, were ineffective in controlling the spread of common carp due to power outages and other unknown factors. Other states, for example Arizona, have experienced similar outcomes in attempting to control grass carp. Other issues related to the efficiency of barriers in controlling the spread of introduced species include flood flows, structural failure, human error, barrier site location and scale (smaller channels likely more effectively blocked), and alternate pathways for dispersal. Over the long-term, barriers have generally been shown to be ineffective in controlling the spread of introduced carps, particularly in eutrophic watersheds.

Another issue related to the efficiency of barriers is the minimum number of fish required to establish a population. Two things are important here: (1) their passage past the barrier and (2) the likelihood of population growth to problem levels. In the absence of barriers, fish passage is unimpeded, however habitat refugia and niche space for bigheaded carps are not created. Constructing an incomplete barrier (<100% effective) creates upstream refugia from slower swimming native predatory species unable to pass the barrier, and therefore may actually aid the reproductive success of bigheaded carps. Impoundments also create habitat refugia and favorable conditions that promote the establishment of bigheaded carps (for example lower velocity and plankton productivity).



A qualitative characterization of risk and certainty for barrier effectiveness for carp on rivers in Minnesota.

Question 4. What are the Effects of Barriers on Native Fish Species in Minnesota?

Overview: When evaluating the use of barriers as a method for controlling or limiting range expansion of introduced carp, it is critical to understand and consider the effect of barriers on the native fish community. It is also important to understand the relative vulnerability of different fish species to fragmentation. Rivers are arteries of biodiversity that allow native species to migrate for reproduction, seasonal habitat needs, optimal foraging through all life history stages, and for recolonization following drought, severe winter, and other disturbances. Some species like sturgeon migrate hundreds of miles to access spawning habitat while American eels migrate thousands of miles from the Sargasso Sea to Minnesota waters. Since native mussels depend on host fish species for reproduction and distribution, effects of barriers extend to these vital members of the aquatic community as well.

Based on published literature and empirical findings presented here, barriers are among the most definitive causes of the extirpation and extinction of native species. For 32 barrier dams across Minnesota, an average of 37% of the fish species found in the watershed did not occur upstream of the dam. Dam removal resulted in an average return of 66% of absent species to the upstream watershed. Common carp and other tolerant species were among the least likely to be absent upstream of barrier dams.

This section begins by examining the various drivers of migration to emphasize the many reasons native fish need connected river reaches. The second subsection presents an analysis of fish survey data assessing the effects of dams on native fish populations. Due to its length, the full assessment on barrier effects is included as an attachment. Thirty-two dams throughout MN were analyzed for their effects on the distribution of 134 native fish species and 16 introduced species. In addition, a more detailed analysis assessed the effects of a dam on upstream fish distributions in the Cottonwood River watershed.

Rapids on the Little Fork River are typical of sturgeon spawning habitat. *Credit DNR SHP.*

Migration in Rivers

Similar to birds that migrate seasonally and return to breeding grounds to optimize habitat, reproduction, foraging, and survival - riverine fish depend on seasonal movements through the river network. Native fish have evolved to optimize their use of available habitat in these dynamic systems. Migration is an essential element of native fishes adaptations to the environment in Minnesota and is essential to their survival. The bulk of native fishes in Minnesota depend on rivers and streams for at least part of their life history. Understanding mechanisms of migration is fundamental to grasping the implications of fragmentation on native fish.

Similar to birds that migrate seasonally and return to breeding grounds to optimize habitat, reproduction, foraging, and survival - riverine fish depend on seasonal movements through the river network.

Drivers of Migration

A number of factors are likely drivers of fish migration. These include life history requirements, response to catastrophic events, range expansion and other factors. These drivers of migration shed light on the relative length and importance of these migrations. Table 3 lists maximum documented migration distances for the longer migrating species such as paddlefish (1,240 miles), lake sturgeon (800 miles) and channel catfish (452 miles).

Spawning Requirements

Successful spawning and development of eggs and fry are dependent on very specific habitat characteristics for many species. Eggs have high oxygen requirements and must be well aerated to develop. Silt or detritus covered stream and lake beds often have relatively low oxygen due to the high oxygen demand of the sediments. Spawning in riffles or rapids, on vegetation, on logs, brush, and other objects, or in nests where the parents fan the eggs and keep them free of silt and detritus are common strategies (Figure 22).

Lengthy spawning migrations have long been known for anadromous salmon but more recently for entirely freshwater species. Spawning habitat is often different from that used by adult individuals the rest of the year when not in spawning mode. This is particularly true of large bodied fish that may spawn in water too shallow for adults as summer flows decrease. Most MN fish species spawn in the spring when flows are high due to snow melt and rain events. This enables the larger species the use of small steeper gradient streams where riffles are prevalent but are not available later in the year due to inadequate flows. Minnesota's largest fish species, the lake sturgeon, will ascend small streams to spawn then migrate back downstream to deep pools in large rivers and lakes the rest of the year.

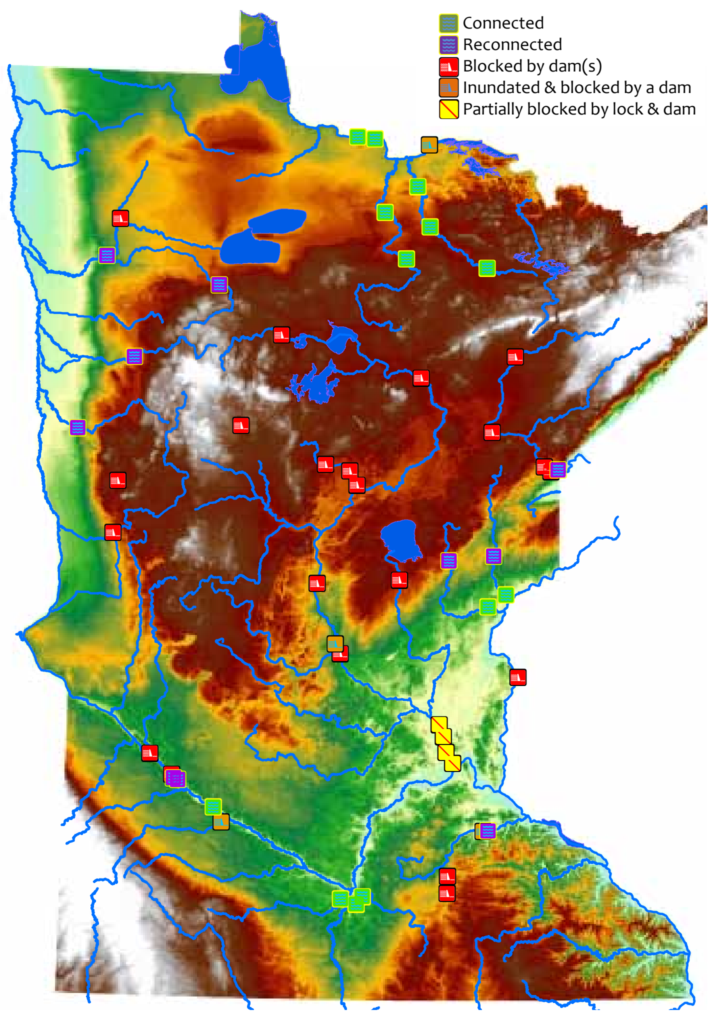


Figure 22. Shaded relief map showing areas with large changes in elevation over short distance (steeper slope) that provide rocky riffle habitat for spawning. Symbols highlight rapids or falls which are spawning hotspots for lake sturgeon and other rheophilic species that spawn in rocky areas.

Winter Survival

While conditions in the Southern United States are relatively moderate throughout the entire year, the winter months in Minnesota can be severe. Ice cover, super-cooled water, frazil and anchor ice, and the lowest flows of the year, can make habitat unsuitable in many small and mid-sized stream reaches.

Seasonal quantitative sampling in the Otter Tail River, a hydrologically stable river with a mean flow of 411 cfs, indicated early winter migrations out of smaller reaches by most of the fish community. Breckenridge Dam, which is 7.7 miles upstream of the confluence with the Red River, had a nature-like fishway, installed in 1996, that has been used to monitor upstream migration (dam was removal in 2007 after several dam failures). Orwell Dam is the first complete barrier on the Otter Tail River 31 miles upstream of the confluence with the Red River. Seasonal monitoring of the fish community just downstream of Orwell dam validates the extent of immigration and emigration (Figure 23, Aadland 2010). While the location to which these fish ultimately migrated is unknown, monitoring of the downstream nature-like fishway suggested that many of the species may migrate downstream to the deep, low gradient Red River of the North followed by spring and early summer migrations (late April through July) upstream to the Otter Tail River.

Many fish species decrease or stop feeding when water temperatures approach freezing even though the energetic cost of maintaining position in flowing water is high. Winter fish sampling has shown high densities of fish in deep, low velocity pools. This may be particularly important for large-bodied

Species	Maximum documented migration (miles)	Maximum documented upstream migration (miles)	Maximum documented downstream migration (miles)	Source
lake sturgeon *Acipenser fulvescens*	800	300	800 Lake Pepin to Minnesota Falls, MN	Nick Schloesser, personal communication Ron Bruch, personal communication
pallid sturgeon *Scaphirhynchus albus*	992 Missouri River	220 Missouri River	316 Missouri River	DeLonay et al. 2009
shovelnose sturgeon *Scaphirhynchus platorynchus*	331 Yellowstone & Missouri Rivers	158 Yellowstone & Missouri Rivers	121 Yellowstone & Missouri Rivers	Schmulbach 1974 Bramblett and White 2001 Brooks et al. 2009
paddlefish *Polyodon spathula*	1240 Missouri River	200 Missouri River	1240 Missouri River	Stancill et al. 2002 Rosen et al. 1982
American eel *Anguilla rostrata*	3440	3440 Sargasso Sea to Red River of the North		Aadland et al. 2005
blue sucker *Cycleptus elongatus*	109	109	49	Neely et al. 2009
greater redhorse *Moxostoma valenciennesi*	9.4		9.4	Bunt and Cooke 2000
channel catfish *Ictalurus punctatus*	452 Red River of the North	290 Red River of the North	452 Red River of the North	Hegrenes 1992
blue catfish *Ictalurus furcatus*	427 Upper Mississippi River	202 Upper Mississippi River	86 Upper Mississippi River	Tripp et al. 2011
flathead catfish *Pylodictis olivaris*	117 Grand &Missouri Rivers	117 Grand &Missouri Rivers	56 Grand &Missouri Rivers	Vokoun and Rabeni 2005
northern pike *Esox lucius*	21 Black River, WI		21 Black River, WI	Finke 1966
white bass *Morone chrysops*	102 Mississippi River	77 Mississippi River	102 Mississippi River	Brooks et al. 2009
smallmouth bass *Micropterus dolomieu*	53	18 Upper Mississippi River	53	Altena 2003 Langhurst and Schoenike 1990
sauger *Sander canadense*	372 Yellowstone River, MT	372 Yellowstone River, MT	372 Yellowstone River, MT	Jaeger 2004
walleye *Sander vitreus*	164 Missouri River, MT	164 Missouri River, MT	164 Missouri River, MT	Bellgraph 2006
Introduced Species				
Common carp *Cyprinus carpio*	403 Murray River, Australia	79 Murray River, Australia	403 Murray River, Australia	Jones and Stuart 2008
Silver carp *Hypophthalmichthys molitrix*	255 Upper Mississippi River	81 Upper Mississippi River	149 Upper Mississippi River	DeGrandchamp et al. 2008 Brooks et al. 2009
Bighead carp *Hypophthalmichthys nobilis*	286 Upper Mississippi River	101 Illinois River	72 Illinois River	DeGrandchamp et al. 2008 Peters et al. 2006

Table 3. Maximum upstream and downstream migration distances of various fish species.

fish. Flathead catfish in the Fox River, Wisconsin moved downstream to deep pools with snags where they went dormant and even become partly covered with silt, meanwhile fish that wintered in nearby lakes were largely immobile but not entirely stationary (Piett and Niebur 2011). Chris Domeier, DNR Fisheries, made similar observations of wintering flathead catfish in the Minnesota River. In 1800, Alexander Henry wrote of winter sturgeon habitat in a deep pool at the confluence of the Red Lake and Red River of the North in East Grand Forks, Minnesota (Gough 1988). Access to these wintering areas may be critical to the survival of large-bodied river fishes.

Drought Survival

Droughts are a common part of the hydrologic regime for most watersheds. Even large rivers such as the Red River of the North at Fargo (1976) and the Minnesota River at Montevideo (1930s) have entirely stopped flowing. During the 1988 drought many of the remaining deep pools in the Minnesota River became anoxic. Downstream migration to larger river reaches followed by recolonization as flows rise is a common strategy for stream fishes and is essential to surviving these occasional events.

Foraging Optimization

Fish have been shown to migrate as a means of optimizing foraging and growth (Werner and Mittlebach 1981). This may be prevalent and an explanation of juvenile migrations observed in a number of stream fishes including channel catfish, walleye, and other species (Aadland 2010). Upstream migration to headwaters, typical of many species, is an example of adaptation to minimize competition while taking advantage of these productive streams.

Flathead catfish wintering in a pool in the Minnesota River. *Credit DNR Fisheries.*

Predator Avoidance

Stream fishes may migrate to reaches with lower piscivore abundance as a means of predator avoidance. For example, Schlosser (1988) showed the presence of smallmouth bass altered habitat use by hornyhead chubs.

Recolonization

Droughts, winter anoxia, chemical spills, and other factors can cause the loss of species from watersheds by either mortality or migration away from uninhabitable areas. These events can result in the loss of all species or the retention of tolerant species that are able to survive the event. Survivors have a significant competitive advantage due to the release from competition and predation population constraints. Connected river systems enable immigration into an impacted watersheds and re-establishment of the metapopulation.

Dispersal

Following the last ice age, most of Minnesota waters were devoid of fish. Biodiversity in our rivers and lakes depended on connections through stream networks primarily from southern latitudes. Modern climate change may result in the same need for basin-wide access to the river network.

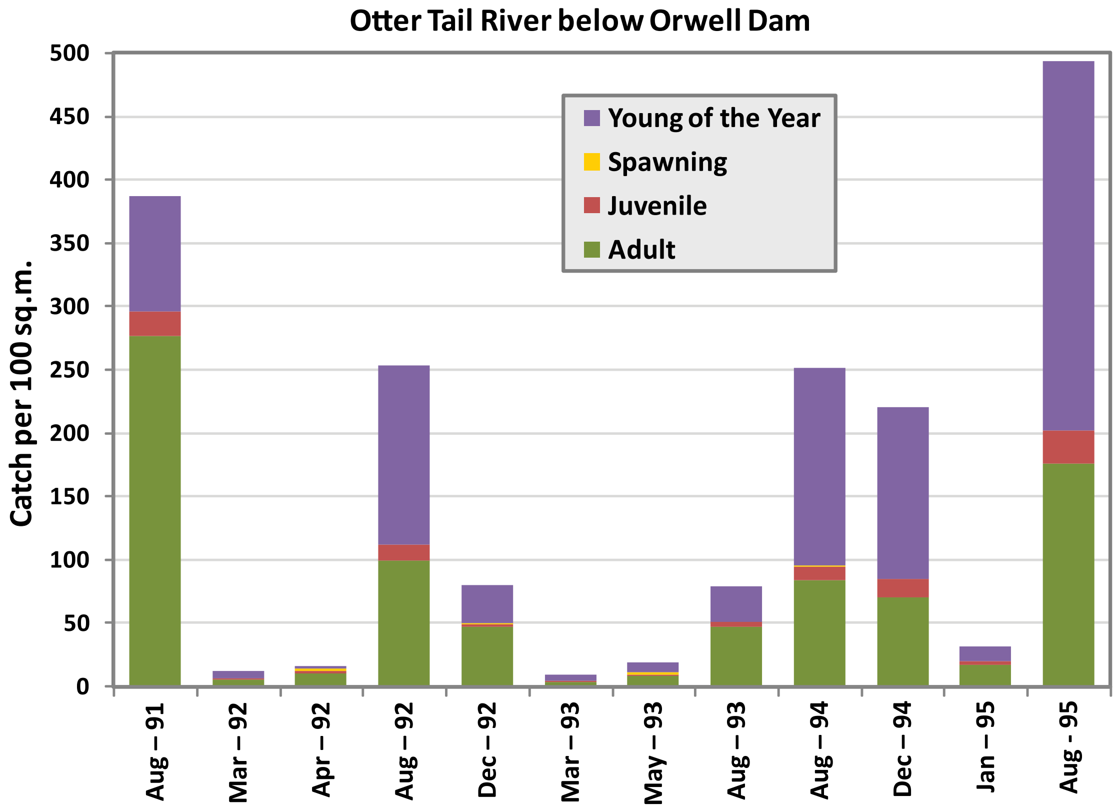
Effects Specific to Dams

Alter Water Quality

Dams can have additional effects on the native aquatic community by:

- interrupting sediment transport causing reservoir sedimentation and downstream incision,
- altering nutrient dynamics and causing cyanobacteria blooms,
- propagating introduced species,
- inundating important river habitat,
- altering flow regimes,
- altering temperature regimes,
- propagating fish diseases and parasites, and
- causing massive erosion when they fail.

Modifying thermal and flow regimes by impoundment are considered to be “major disruptions of continuum processes.” The concept of serial discontinuity explains the effect of dams,



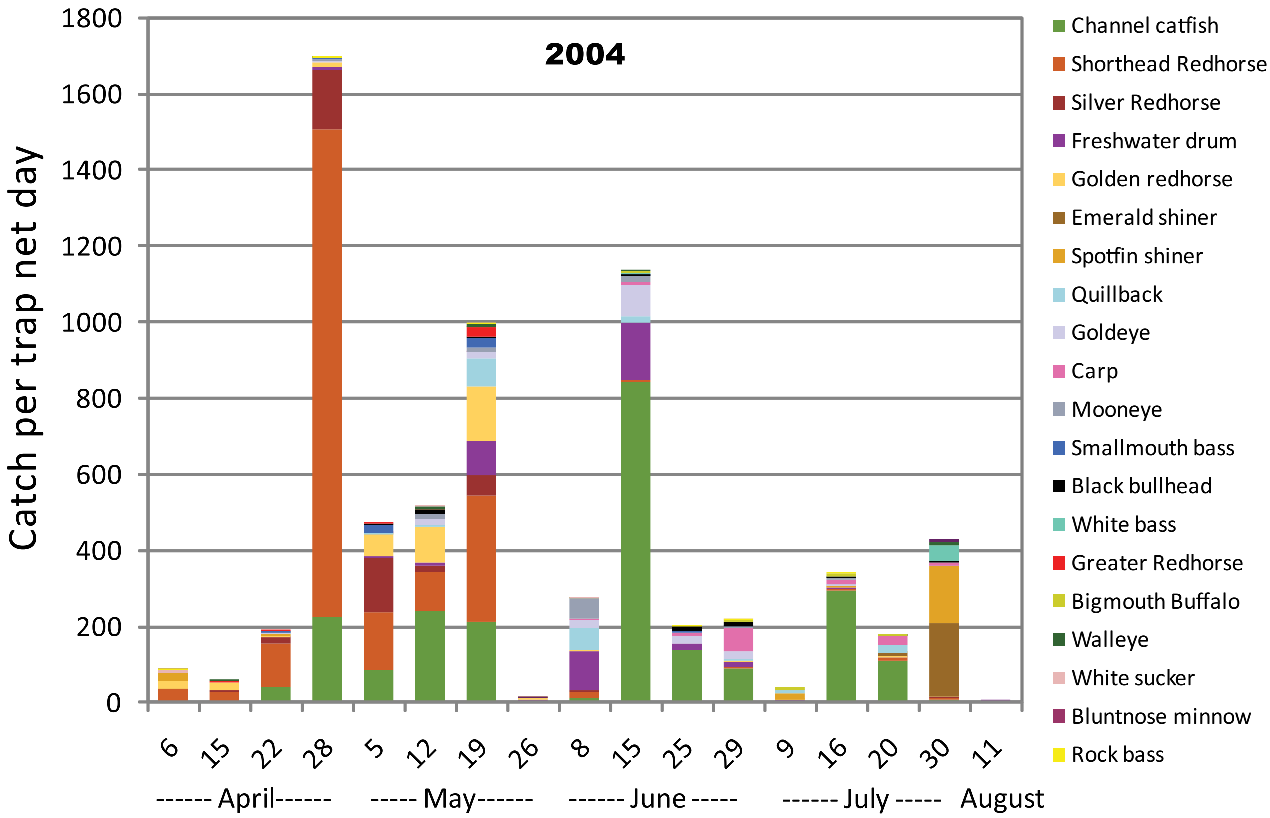


Figure 23. (top) Seasonal fish catch data from the Otter Tail River just below Orwell Dam. The data has been separated into life stages: young of the year, spawning, juvenile and adult. Spring migration in this river peaks in late April early May. Low fish abundances in the spring is pre upstream spawning migration followed by higher abundances in the fall and early winter before winter emigration. (bottom) Catch in the Breckenridge Fishway in 2004. Fish species in legend are listed in a general order of abundance. Low abundance species (<5 caught annually) not graphed include: northern pike, black crappie, bluegill, blackside darter, chestnut lamprey, pumpkinseed, sand shiner, brown bullhead, sauger, and stonecat.

which displace aquatic communities along the river continuum (Ward & Stanford 1983) (Figure 24). Changes in flow regime, water temperature, oxygen, turbidity, and the quality and quantity of food particles in the river downstream of impoundments shift the upstream-downstream patterns of biotic structure and function predicted by the River Continuum Concept (RCC). Basically the RCC explains that as the river grows in size the processes and the aquatic organisms change as the food source changes. The serial discontinuity concept predicts the way dams shift this expected continuum. The reach immediately downstream of the dam may essentially be reset to a condition that is more like that found upstream or downstream. This can be measured by 16 variables, including the ratio of coarse particulate to fine particulate organic matter, the relation of substrate size to biodiversity, and environmental heterogeneity. A dam may result in some conditions being more like those of

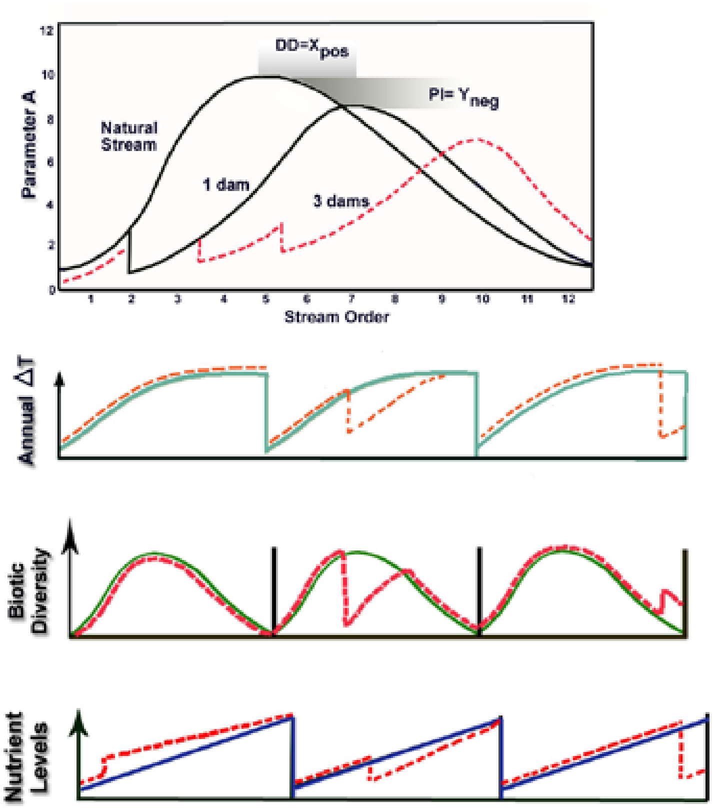


Figure 24. Theoretical framework for conceptualizing the impact of dams on selected ecological parameters. The relative influence of one and three dams is shown in the upper graph. Discontinuity distance (DD) is the positive downstream or negative upstream shift of a parameter a given distance (X) due to stream regulation. The change in parameter intensity (PI) is also defined. The relative changes in three of many potentially affected parameters as a function of stream order and postulated effects of locating a dam at different points on the continuum are shown in the lower three graphs (From Ward and Stanford 1983).

the headwaters (an upstream shift), while other conditions become more like those of downstream segments (a downstream shift) (Ward and Stanford 1983). Other characteristics may not fit either paradigm (Annear and Neuhold 1983).

Moreover, dams and reservoirs create lentic environments where production is based on plankton rather than the benthic algae and allochthonous material on which lotic production is usually based. When reservoir water is released to a stream, it carries with it the plankton that would otherwise be scarce in streams (Annear et al. 2004).

Alter Hydraulic Regimes and Habitat

Dams on rivers alter the water level in the impoundment above the dam and can affect water levels in the downstream channel and connected backwaters depending on operation and resultant flow regime. The degree of water level increase and duration in the impoundment is related to the specifics characteristics of the dam structure (size, height, outflow design) and its operation. Sediment transport, nutrient cycling, temperature effects, vegetation, plankton productivity and composition are all related to the size of the impoundment created by the dam. Backwater channels may be inundated or created depending on the specifics of the existing channel.

High gradient rapids and falls are often target locations for dams. For example, Fergus Falls, Internationals Falls, Pelican Rapids, Coon Rapids, Granite Falls, St. Anthony Falls, St. Croix Falls are all dam locations. Inundation of these fast water habitats is particularly damaging to those species that require these habitats for spawning. Many of these rapids-dependent species are now imperiled.

An Analysis of Barrier Effects on Native Fishes

Fish survey data from 32 dams and their watersheds throughout Minnesota was analyzed as a means of addressing the effects of barriers on native fishes in Minnesota. The presence/absence of fish species in the upstream versus downstream watersheds was used to determine relative vulnerability to barrier-caused extirpation. The dams assessed are, or were (have been removed), located in tributaries and mainstems of the Minnesota, Red River of the North, St. Croix, St. Louis, Missouri and Mississippi river watersheds. The fish records used in this analysis included hundreds of fish surveys conducted by the MN DNR, PCA, the Bell Museum, and others and involved 150 fish species, 134 native fish and 16 introduced fish species. Additionally, a detailed assessment of the Cottonwood River watershed was completed to address the effects of a dam on watershed scale fish diversity. The Attachment is the full report for this scientific investigation and includes the complete methods, results, and discussion. The primary results of this study are presented here.

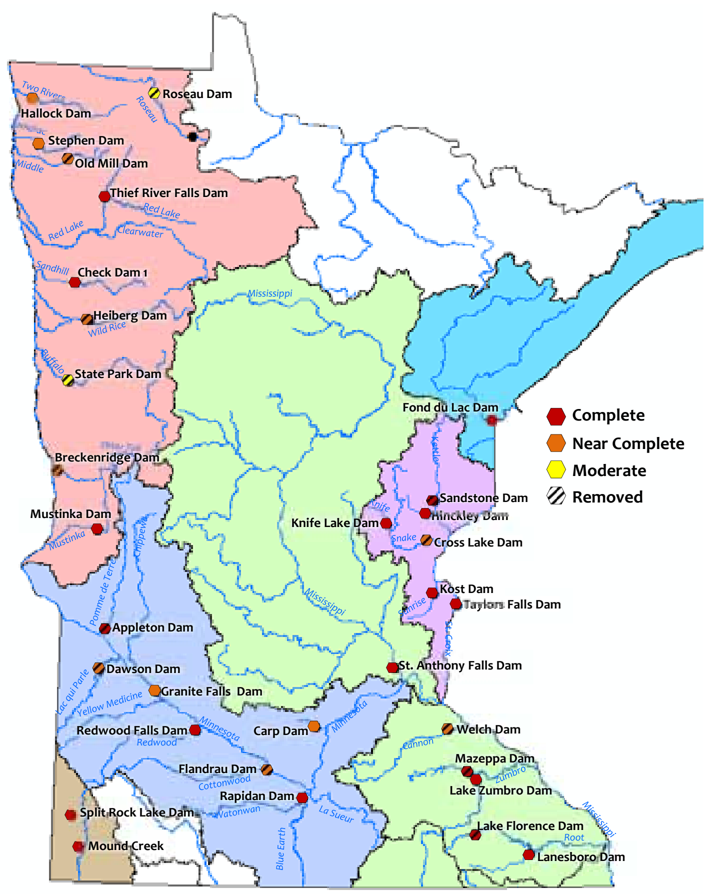


Figure 25. The locations, effectiveness, and current status of the 32 dams included in the barrier assessment.

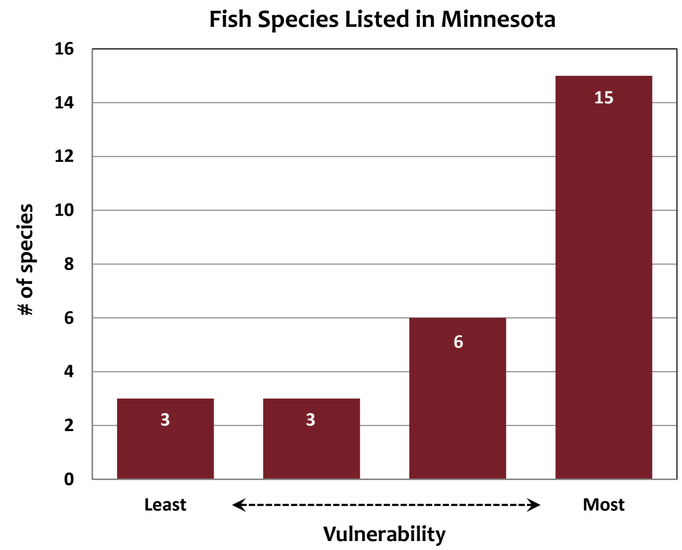
Result #1 Of the 32 barriers evaluated, an average of 37% of the species sampled in the watershed were absent from collections upstream of the barrier (Figure 25, Table 4 and 5).

Table 4. Summary of Barrier Assessment		
Barrier Effectiveness	**# of Dams Assessed**	**Average % Absence**
Complete	19	41%
Near Complete	9	37%
Moderate	4	20%
Overall Average	***32***	***37%***

Result #2 Most native species were found to be vulnerable to extirpation by dams. Nearly half, 68 of 134 native species, were absent upstream of at least half of the barriers for which they were assessed (Table 6). A total of 27 (20%) native fish species were absent upstream of every barrier (100%) for watersheds where they were found.

Table 6. Summary of Barrier-caused Extirpation Fish Data	
# of dams/watersheds analyzed	32
# of native fish species present in these watersheds	134
# of introduced species present in these watersheds	16
# of native species absent above every dam for watersheds in which they were present (Of which 2 are Federally listed, 12 are listed in Minnesota, and 14 are sensitive and pollution intolerant species)	27 (20%)
# of native species absent above at least half the barriers for watersheds in which they were present	68 (50%)

Result #3 Imperiled (special concern, threatened, and endangered), sensitive, stream-dependent species were most affected by barriers. Conversely, tolerant, habitat generalists, lake-oriented, headwater, and widely stocked species, including common carp, were among the species least affected by barriers (Figure 26).



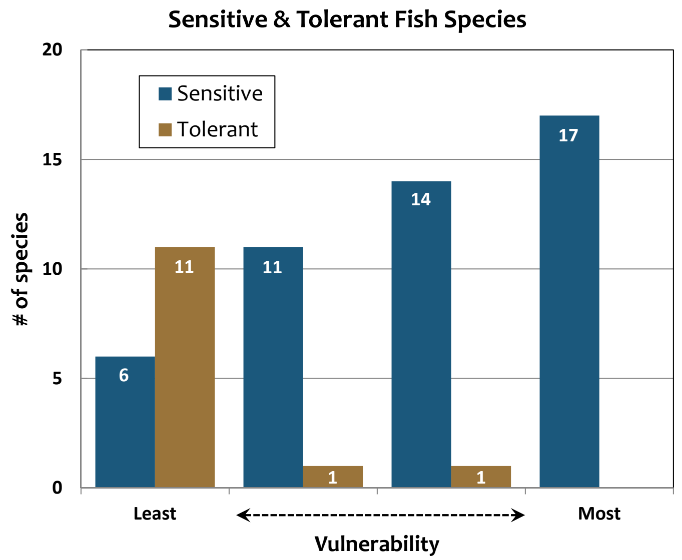


Figure 26. (top) Number of listed species in Minnesota in percent absence quartiles. (bottom) Number of sensitive and tolerant native fish species (including naturalized common carp) in percent absence quartiles.

Result #4 The influence of a dam near the mouth of the Cottonwood River extended virtually watershed-wide, over 150 miles upstream to the headwaters (Figure 27). This was determined by comparing species richness in the watershed when the dam was present to species richness after the dam was removed. After the dam was removed, species richness increased by an average of 35% in the watershed and this increase extended to the upper headwaters.

Result #5 Species richness increased significantly after dams were removed or modified for fish passage. This was demonstrated by eleven of the barriers in this assessment that were subsequently removed (Table 6). An average of 66% (up to 89%) of the species that had been absent recolonized above these 11 barriers following removal. More specifically, a total of 34 species returned to the reach upstream (between Welch and Byllsby dam, the next upstream barrier) after the Welch Dam was removed on the Cannon River. The return of numerous species following removal substantiates the direct impact of barriers on aquatic biodiversity.

For the full report on this scientific investigation, including the results and discussion, see the Attachment - Barrier Effects on Native Fishes of Minnesota.

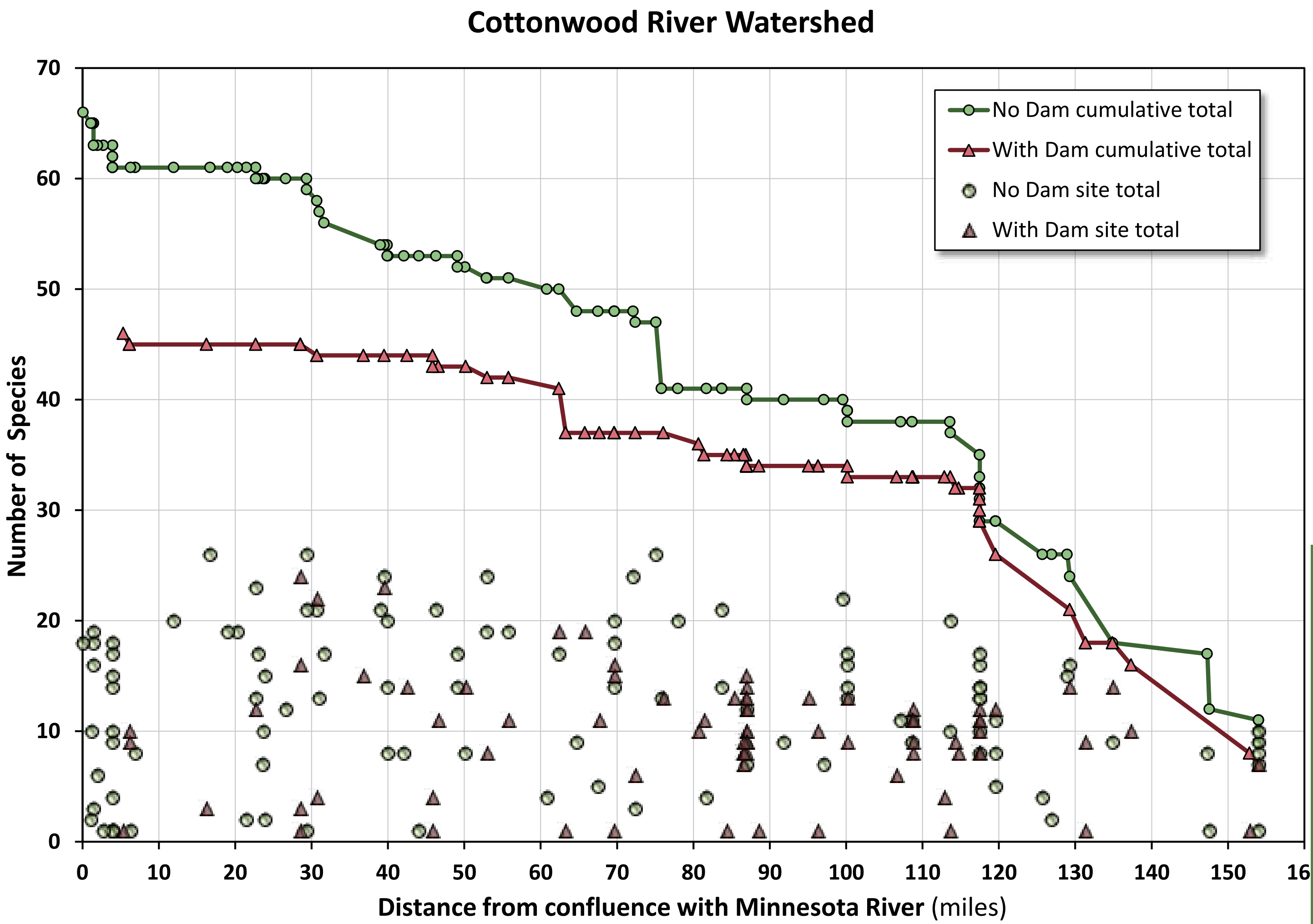


Figure 27. Number of species found in the Cottonwood River watershed. Points are the total number of species collected at a site. The line is the cumulative total. Species richness correlated with distance from the mouth of the Cottonwood River.

Watershed	Barrier Name Year Built, Year Removed	Dam height at low flow (ft) Barrier Effectiveness	Watershed area (mi^2) upstream of dam / total % of watershed upstream of dam	Total # of native species observed in watershed Additional introduced species	# of Native MN species absent upstream of barrier (% of total)
Red River of the North Basin					
Otter Tail River	**Breckenridge Dam** 1935. Replaced with rock ramp in 2007	8 Near Complete	1,910 / 1,952 97.8%	75 1	9 (12%)
Mustinka River	**Mustinka Dam** 1940	18 Complete	163 / 861 18.9%	30 1	15 (50%)
Buffalo River	**State Park Dam** Pre-1893, 1937 Removed in 2002	3.5 Moderate	325 / 975 33.3%	58 1	21 (36%)
Wild Rice River	**Heiberg Dam** 1875. Removed in 2006	8 Near Complete	934 / 1,560 59.9%	61 1	16 (26%)
Sand Hill River	**Check Dam 1** 1955	10 Complete	308 / 420 73.3%	36 1	15 (42%)
Red Lake River	**Thief River Falls Dam** 1946	16.75 Complete	3,450 / 5,680 60.7%	64 3	13 (20%)
Middle River	**Old Mill Dam** 1886, 1938. Removed in 2001	8.5 Near Complete	225 / 779 28.9%	32 1	25 (78%)
Tamarac River	**Stephen Dam** 1975	12 Near Complete	283 / 397 71.3%	37 1	9 (24%)
Roseau River	**Roseau Dam** 1932. Replaced with rock ramp in 2001	5 Moderate	474 / 1,420 33.4%	44 1	10 (23%)
South Branch Two Rivers	**Hallock Dam** 1938	8 Near Complete	592 / 1,100 53.8%	42 1	13 (31%)
St. Croix River Basin					
St. Croix River	**Taylors Falls Dam** 1890, 1907	50 Complete	6,240 / 7,650 81.6%	106 5	31 (29%)
Snake River	**Cross Lake Dam** 1800s, 1938, 1963. Modified with rock ramp in 2013	2 Moderate	974 / 1,009 96.5%	68 1	2 (3%)
Knife/Snake River	**Knife Lake Dam** 1983	14 Complete	92/1,009 9.1%	68 1	33 (49%)
Kettle River	**Sandstone Dam** 1908. Removed in 1995	20 Complete	868 / 1,060 81.9%	64 5	22 (34%)
Grindstone River	**Hinckley Dam** 1955	10 Complete	77 / 1,060 7.3%	64 5	30 (47%)
Sunrise River	**Kost Dam** 1885	13 Complete	268 / 283 94.7%	64 2	19 (30%)

Table 5. Watersheds assessed for barrier effects on fish species richness. Barrier effectiveness is based on dam height and frequency of inundation by floods; Complete = complete barrier, Near Complete = near complete barrier that may be passable during large floods (10-year or larger), Moderate = moderate flood barrier that may be passable during moderate floods (2-year or larger).

Watershed	Barrier Name Year Built, Year Removed	Dam height at low flow (ft) Barrier Effectiveness	Watershed area (mi²) upstream of dam / total % of watershed upstream of dam	Total # of native species observed in watershed Additional introduced species	# of Native MN species absent upstream of barrier (% of total)
Lower Mississippi River Basin					
Mississippi River (upstream of Iowa border)	**St. Anthony Falls Dam** 1848, 1963	49 Complete	19,100 / 65,000 29.4%	127 8	64 (50%)
South Branch Root River	**Lanesboro Dam** 1868	28 Complete	284 / 1,250 22.7%	93 4	57 (61%)
North Branch Root River	**Lake Florence Dam** 1857. Removed in 1993	12 Complete	119 / 1,250 9.5%	92 4	65 (70%)
Zumbro River	**Lake Zumbro Dam** 1919	55 Complete	845 / 1.150 73.5%	89 4	27 (30%)
North Fork Zumbro River	**Mazeppa Dam** 1922. Removed in 2001	20 lowered to 10 Complete	174 /1,150 15.1%	89 4	65 (73%)
Cannon River	**Welch Dam** 1900. Removed in 1994	8 Near Complete	1,340 / 1,440 93.1%	82 5	19 (23%)
Minnesota River Basin					
Minnesota River	**Granite Falls Dam** 1911	17 Near Complete	6,180 / 16,200 38.1%	97 4	39 (40%)
High Island Creek	**Carp Dam** 1958	6 Near Complete	206 / 241 85.5%	47 1	30 (64%)
Blue Earth River	**Rapidan Dam** 1910	55 Complete	2,410 / 3,486 69.1%	66 1	26 (39%)
Cottonwood River	**Flandrau Dam** 1937, Was repeatedly damaged by floods & was removed in 1995	28 lowered to 12 Near Complete	1,310 / 1,313 99.8%	65 2	24 (37%)
Redwood River	**Redwood Falls Dam** 1902	34 Complete	630 / 665 94.7%	53 2	19 (36%)
Pomme de Terre River	**Appleton Dam** 1872. Removed in 1999	13 – 16 Complete	905 / 915 98.9%	65 1	17 (26%)
Lac qui Parle River	**Dawson Dam** 1913. Replaced with rock ramp in 2009	8 Moderate	472 / 1,156 40.8%	41 1	8 (20%)
Missouri River Basin					
Mound Creek	**South Dam** 1936	14 Complete	16.8 / 17.2 97.7%	29 1	9 (31%)
Split Rock Creek	**Split Rock Dam** 1937	24 Complete	45 / 320 13.9%	26 1	10 (38%)
Lake Superior Basin					
St. Louis River	**Fond du Lac Dam** 1924	78 Complete	3.600 / 3,634 99.1%	62 11	9 (15%)

Barrier	Native fish species absent in upstream watershed while dam was present then found upstream of dam site after removal or modification or when dam was breached	# of species returned
Breckenridge Dam Otter Tail River Built in 1935 Replaced with rock ramp in 2007	silver lamprey[L], longnose gar[L], goldeye[L,I], mooneye[L,I], stonecat[I], white bass, sauger, lake sturgeon[MN,L]*	8 species (89% of 9 absent species)
State Park Dam Buffalo River Built pre 1893 & 1937 Removed in 2002	silver lamprey[L], goldeye[L,I], spotfin shiner, carmine shiner[L,I], sand shiner, northern redbelly dace[L], blacknose dace, quillback[L], silver redhorse, channel catfish, green sunfish, smallmouth bass[I], sauger, freshwater drum	14 species (67% of 21 absent species)
Heiberg Dam Wild Rice River Built in 1875 Removed in 2006	goldeye[L,I], brassy minnow, emerald shiner, carmine shiner[L,I], finescale dace[L], quillback[L], silver redhorse, channel catfish, tadpole madtom, smallmouth bass[I], sauger, freshwater drum, lake sturgeon[MN,L]*	13 species (81% of 16 absent species)
Sandstone Dam, Kettle River Built in 1905 Removed in 1995	southern brook lamprey[MN,I], blackchin shiner[I], blacknose shiner[L,I], mimic shiner[I], northern redbelly dace[L], bluntnose minnow, tullibee, banded killifish[L], gilt darter[MN,L,I], blackside darter[L], slimy sculpin[I], emerald shiner	12 species (55% of 22 absent species)
Welch Dam Cannon River Built in 1900 Removed in 1994	paddlefish[MN,L,I], mooneye[L,I], gizzard shad, speckled chub[L,I], silver chub[L], mimic shiner[I], river carpsucker, highfin carpsucker[I], river redhorse[L,I], flathead catfish[L], Muskellunge[I], brook trout[I], sauger, lake sturgeon[MN,L]	14 species (74% of 19 absent species)
Minnesota Falls Dam Minnesota River Built in 1871 & 1904 Removed winter 2013	shovelnose sturgeon[L], lake sturgeon[MN,L], flathead catfish[L], paddlefish[MN,L,I], mooneye[L,I], American eel[MN,L], gizzard shad, highfin carpsucker[I], blue sucker[MN,L,I], black buffalo[MN,L,I], sauger, silver lamprey[L] Notes: Removal was very recent so sampling effort has been limited and focused on the large species. American eel made it around dam during 2007 flood.	12 species (31% of 39 absent species) preliminary
Lake Florence Dam North Branch Root River Built in 1857 Removed in 1993	slenderhead darter[L,I], banded darter[I], smallmouth bass[I], bluegill, greater redhorse[L,I], golden redhorse[L], black redhorse[MN,L,I], smallmouth buffalo, northern hogsucker[L,I], longnose dace[I], sand shiner, gravel chub[MN,L,I], spotfin shiner, largescale stoneroller, chestnut lamprey[L]	15 species (23% of 65 absent species
Flandrau Dam, Cottonwood River Built in 1937. Dam was damaged by floods in 1947, was rebuilt in 1960, damaged again in 1965 and 1969, finally was fully removed in 1995	shovelnose sturgeon[L], mooneye[L,I], gizzard shad, golden shiner, river shiner[L], mimic shiner[I], river carpsucker, highfin carpsucker[I], black buffalo[MN,L,I], yellow bullhead[L], brown bullhead, channel catfish, white bass, Iowa darter[I], logperch[L], sauger, carmine shiner[L,I], freshwater drum, Mississippi silvery minnow[MN,I], speckled chub[L,I], silver chub[L] Note: Returned either while dam was passable or after it was removed.	21 species (88% of 24 absent species)
Dawson Dam Lac qui Parle River Built in 1913 Replaced with rock ramp in 2009	bigmouth buffalo[L], greater redhorse[L,I], channel catfish, bluegill, walleye	5 species (63% of 8 absent species)
Appleton Dam Pomme de Terre River Built in 1872 Removed in 1999	emerald shiner, carmine shiner[L,I], quillback[L], silver redhorse, greater redhorse[L,I], channel catfish, white bass, banded darter[I], freshwater drum	9 species (53% of 17 absent species)
Carp Barrier Dam, Drywood Creek, a tributary of the Pomme de Terre River Built in 1930s, failed, built taller in 1971. Failed in 2001	spotfin shiner, spottail shiner[I], common shiner, golden shiner, quillback[L], white sucker, shorthead redhorse, channel catfish, stonecat[I], Iowa darter[I], Johnny darter, banded darter, freshwater drum	13 species (72% of 18 absent species)

Table 6. Native fish species that returned to the watershed upstream of dam barriers after the dams were removed or modified or while the dam was passable. MN = listed in Minnesota, L = listed in neighboring state or province, I = intolerant, * lake sturgeon were re-introduced since extirpation in the Red River Basin. The average does not include Minnesota Falls Dam since the after removal data is preliminary.

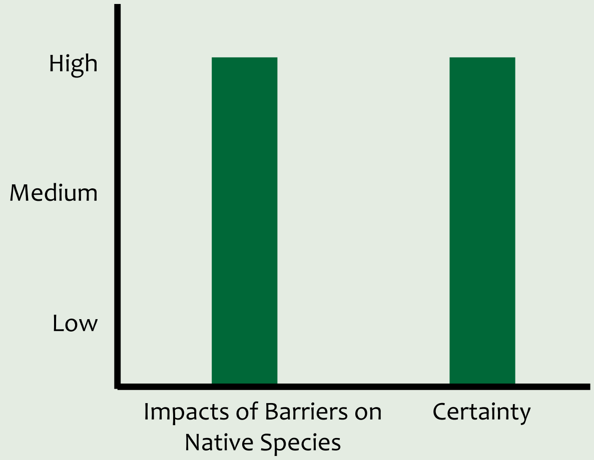
Summary:

Migration is a critical aspect of native fish biodiversity and ecology. Minnesota would be virtually devoid of life were it not for post-glacial invasions. The survival of many native fish populations depends on the ability and freedom to migrate to: access spawning areas, find refugia during drought-related low flows and hypoxia or other disturbance events, recolonize after these disturbance events, and find specific food resources needed at various life stages.

Based on an analysis of 32 dams across Minnesota, barriers have a dramatic impact on aquatic biodiversity, reducing upstream species richness by an average of 20% to 41% depending on the efficacy of the barrier. Loss of species richness due to barriers extends watershed-wide. Imperiled and sensitive species, such as paddlefish, blue sucker, black buffalo, and mooneye, were the most vulnerable to extirpation by barrier dams, while tolerant species, including common carp, were among the species least affected by barriers.

Barriers have direct negative effects on recreation as a number of game fish species were vulnerable to barrier related extirpation. Flathead catfish, sauger, speckled chub, shovelnose sturgeon and paddlefish are examples of the 27 species absent upstream of all barriers evaluated while lake sturgeon, channel catfish, white bass, and key forage species were absent upstream of most barriers in watersheds where they were present.

An average of 66% (up to 89%) of native fish species absent above barrier dams returned after the barrier was removed. This substantiates that the barriers were the cause of extirpation of these species. These data, along with published literature, demonstrate that barriers are among the most definitive causes of the extirpation and extinction of aquatic species.



A qualitative characterization of risk and certainty for barrier effects on native fishes in Minnesota.

Question 5. What are Alternative Approaches?

Overview: Separate from the use of barriers, there are two basic approaches for management of introduced species: (1) approaches targeting introduced species and (2) those focused on native species. Some consideration of elements within each of these approaches is important in guiding ecologically sound management to efficient and ultimately successful strategies.

Targeted controls on bigheaded carps, should they become established, include harvest by commercial fishermen. There is a growing demand for fish due to depletion of most global fisheries and increasing protein shortages. Reductions in harvest of flathead catfish and other predators on carp (all life stages)is another means of increasing population control and invasion resistance. Other targeted controls options may exist but require further research and development and need to be assessed to assure that there are no negative effects on native species and ecosystem processes.

Degraded, eutrophic, and impounded river networks have been shown to favor tolerant introduced species including bigheaded carps. Improving water quality and restoring healthy, free-flowing rivers are ecologically sound means of reducing habitat and resources for these species while reestablishing competitive advantages for native species. This would also increase competition and predation controls by native species.

Approaches Targeting Introduced Carps

When introduced fishes become invasive and are believed to be detrimental, there are a number of management actions that are typically proposed to minimize their dispersal and impacts. These actions include:

1) eradication attempts from specific waters or well-defined spatial areas,
2) population control by suppression (e.g., through harvest programs), and
3) containment of existing populations to prevent their further spread (Britton et al. 2010).

1) Eradication of Introduced Species

Eradication methods are often acute, being applied over a short time frame and aiming for rapid, total mortality in the target population (Britton et al. 2010). Conceptually, this is considered a desired goal as it inhibits opportunities for compensatory population responses that would negate eradication success (Britton and Brazier 2006).

Chemical treatments are used regularly in introduced fish eradications. Rotenone is one commonly applied fish toxicant. In practice, effective application of rotenone is highly variable according to species, rate of degradation in the target environment, water temperature, light exposure, water depth, absorption by suspended solids and benthic deposits, proper concentration and distribution evenness in the water, availability of underwater refuges, groundwater recharge and presence of aquatic macrophytes (Britton et al. 2010). Successful use of rotenone is primarily limited to small, easily accessible, closed lentic (still water) systems that are shallow and sparsely vegetated. Concern of losses of native fishes (collateral damage), has been addressed through removal of individuals prior to application and release after degradation of the active ingredient to non-toxic levels. Large river systems are not considered suitable for this eradication method.

As use of chemicals to eradicate introduced species has become more controversial to stakeholders, selection of an eradication method has to consider fish welfare and ethics (Huntingford et al. 2006) and the efficiency, selectivity, undesired effects and public acceptability of the method (Myers et al. 2000, Zavaleta 2002). An optimum eradication method would be one that is lethal to the target species only (Sorenson and Stacey 2004, Cotton and Wedekind 2007), however these remain rare.

2) Population Control by Suppression

Selective Pathogens

Koi herpes virus has been associated with die-offs of common carp. Since it is selective for common carp, koi, and goldfish, it may be an effective biological control for this species. Similar selective pathogens may exist for silver and bighead carp but have not yet been identified.

Triploidy

Research of life history intervention strategies, such as triploidy (triploid fish are reproductively sterile), could be pursued as a part of a comprehensive approach to dealing with these introduced species. However, triploid fish may be unlikely to be a useful control technique because they don't appear to engage in spawning activity.

Pheromones

Most fishes rely on pheromones to mediate social behaviors, with three main categories of pheromones being identified based on their function: anti-predator cues, social cues, and reproductive cues (Sorenson and Stacy 2004, Burnard et al. 2008). Chemically identified pheromones are highly potent, which combined with their specificity makes them potentially useful for controlling introduced fish populations (Sorenson and Stacy 2004). Their use should likely involve a variety of pheromones to supplement and increase the efficiencies of other control strategies (Gozlan et al. 2010); for example, to increase trapping efficiency (Twohey et al. 2003), disrupt or reduce reproductive success (Carde and Minks 1997; Wyatt 2003), disrupt movement and migrations (Li et al. 2003, Sorenson and Vrieze 2003), and promote the success of sterilized fish and repel fish from sensitive areas (Maniak et al. 2000).

Harvest

Physical removal typically uses techniques such as electric fishing and netting to capture target organisms. Harvesting may be effective in suppressing population abundance and reducing their recruitment (Britton et al. 2010). Timing of the deployment of control methods may be important to prevent compensatory responses in the target populations, as was revealed in an operation to control introduced Eurasian perch in New Zealand (Ludgate and Closs 2003). Removal of adult Eurasian perch, which cannibalize younger fish, actually increased juvenile perch in the New Zealand ponds. Knowledge of bigheaded carps life history and reproduction strategies in rivers may provide the means to be able to target particular locations where netting (e.g., seining young-of-year in select backwaters) and electro-fishing efforts can exact maximum impact on the introduced populations. Efforts to suppress introduced fish populations through cropping may be improved by modeling the outcomes of timing, life stage targeting, and effects on population recruitment (Peterson et al. 2008). In addition, subsidies could be placed on bigheaded carps, to create incentives to harvest, and presumably increase the likelihood of population suppression. Care in administration of subsidies may need to be exercised such that incentives to spread the targeted species were not inadvertently created. Bycatch mortality would also need to be addressed with this management option.

This technique is especially pertinent today given exponential human population growth, the shortage of dietary protein, and the over-exploitation of global fisheries.

3) Containment

Construction of artificial barriers to prevent the spread of introduced non-native fishes has become a commonly proposed strategy and is the subject of much of this report. Given the profound impact on native species and the poor historical success of barriers in stopping common carp, barriers on rivers and streams should not be considered a viable means of dealing with introduced carp. Barriers across natural divides or man-made channels would be an acceptable strategy for containing introduced carp.

Given the profound impact on native species and the poor historical success of barriers in stopping common carp, barriers on rivers and streams should not be considered a viable means of dealing with introduced carp.

Approaches Focusing on the Native Species

The successful invasion of waters by introduced fishes is often coincident with environmental degradation (Britton et al. 2010). Efforts to restore aquatic habitats and native species to their former state can impede invasion (Nicol et al. 2004). Therefore, if our underlying goal is to protect native species and maintain the health of their populations and the environment that supports them, a series of management strategies and actions are likely necessary, as there are multiple causes of degradation to address. Gore and Shields (1995), Graf (2001), Poff et al. (1997), and Stanford et al. (1996) offer several recommendations for improving the physical condition of large American rivers, modified from Hughes et al. (2005):

- Take a basin and riverscape perspective to emphasize the movements of water, sediments, nutrients, wood, and biota along the river continuum and between rivers and their floodplains;
- Manage towards the dynamic nature of rivers and a naturally variable flow regime (including magnitude, frequency, duration, timing, and rate of change) versus an equilibrium or average state;
- Re-regulate dam storage and releases to simulate natural peak and low flows;
- Reduce fragmentation by operating or removing dams to rehabilitate physical conditions;
- Work with the river and its tendencies. Use Natural Channel Design (see Rosgen 2007) to design restoration projects which set channel stability and further allow the river to develop habitat heterogeneity;
- Include a number of flow and channel measures in all river-monitoring and management programs;
- Increase complexity in channel types, backwaters, floodplains, vegetation, large wood, substrate, and channel depths;
- Identify and protect those rivers or river reaches that remain in minimally disturbed condition;
- Discourage floodplain and riparian settlements by humans through removal of flood insurance programs, irrigation subsidies, levees, and revetments;
- Improve information transfer and synthesis among river ecologists, hydrologists, managers, policy makers, and the general public.

Ecosystem Restoration

The success of tolerant introduced species has frequently been a response to habitat degradation and fragmentation. In this respect they are a symptom of the declining health of ecosystems. Introduced species such as silver carp, bighead carp, and common carp are very successful in fragmented, impounded, channelized, and nutrient enriched river and stream systems.

It follows that restoration of stream habitat favors native species and reverses the competitive advantage for introduced species. Dam removal, remeandering channelized rivers, restoring perennial vegetation in riparian zones and improving conservation measures in agricultural lands can improve water quality, habitat, and ecosystem function favoring native species rather than introduced species.

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(top) Appleton Milldam. (bottom) The dam site 8 years after dam removal and river restoration. Nine of 17 absent fish species and 3 of 4 absent mussel species returned upstream after removal of the barrier. *Credit DNR SHP.*

Additional Strategies - Biological Controls

Native Predators on Bigheaded Carps

While some sources have suggested bigheaded carps have no natural predators in North America, a number of native species likely feed on bigheaded carp eggs, fry, juveniles, and adults. Unfortunately, empirically-based studies of bigheaded carp predators and stomach content analyses are largely lacking in the literature. Common, silver, bighead, grass, and black carp can all attain a large size reducing the number of native predators large enough to feed on adult fish. However, fish populations are often controlled at earlier life stages and small predators can have significant effects on large bodied fishes. It is worthwhile to consider likely predators on each life stage.

Eggs and Fry

Since silver, bighead, grass, and black carp are all pelagic spawners with fry developing in the water column, it follows that pelagic predators that feed in the water column are most likely to be predators on these life stages. Many species meet this criterion but several warrant special discussion here.

American shad have similar pelagic spawning behavior to the bigheaded carp species providing a well-studied surrogate for potential predators. Johnson and Dropkin (1992) and Johnson and Ringler (1998) found 15 and 22 fish species, respectively, that had eaten American shad larvae in the Susquehanna River. Species found in Minnesota that had eaten larval shad in the Susquehanna River included central stoneroller, creek chub, carmine shiners, spotfin shiner, spottail shiner, bluntnose minnow, banded killifish, rock bass, bluegill, largemouth bass, and smallmouth bass. Individual smallmouth bass ate up to 900 fry, while spotfin shiners averaged only 8 larvae per fish yet their abundance made their overall predation influence substantial.

Juveniles

As larval bigheaded carps develop, they move to low velocity habitats. A number of predators are likely to feed on juvenile carp. Goldeye were an important predator on pelagic Chinook salmon smolts in Lake Sakakawea, ND (Aadland, 1987). Like bigheaded carps, Chinook salmon in this reservoir are a fast swimming introduced species associated with the water column. Sauger, walleye, smallmouth bass, and other species also fed heavily on salmon smolts. Other water column predators include mooneye, hickory shad (historically abundant in the lower Minnesota River), and white bass which are all likely predators on bigheaded carp eggs and fry and juveniles.

(top) A longnose gar preying on a fathead minnow. (bottom) A bowfin, a potential predator of bigheaded carp that is able to inhabit anoxic backwaters by gulping air. *Credit DNR SHP.*

Juvenile silver carp have been associated with anoxic backwaters where they can avoid predators by respirating at the water-surface interface using a vascularized extension of the lower jaw (Duane Chapman, USGS, personal communications 2013). While most predators cannot survive these anoxic conditions, longnose gar, shortnose gar, and bowfin have lung-like swim bladders that allow them to gulp atmospheric oxygen and live in these same backwaters. Shortnose gar grow to 9 pounds and fed heavily on common carp up to 5 inches long (Shields 1957). Longnose gar grow to as large as 6 feet long and 50 pounds and also feed heavily on carp (Becker 1983). Bowfin grow to over 20 pounds and 3.5 feet long.

Sub-adults and Adults

Large predators such as northern pike and channel catfish may consume carp up to about a foot long after which they would grow out of the threat of all but the largest individuals of these species. Flathead catfish are capable of consuming common carp up to 30% of their own body weight (Davis 1985). Since flatheads can grow to well over 50 pounds, they are capable of eating all but the largest bigheaded carps. A flathead illegally caught in the Minnesota River in 1930 reportedly weighed 157 pounds. While flatheads are often associated with the bottom of deep pools, they become active at night and are capable of feeding throughout the water column as illustrated by the photo below. Adult flathead catfish in Milford Reservoir, Kansas fed primarily on gizzard shad (Layher and Boles 1980). Like bigheaded carps, gizzard shad are associated with the water column and are found primarily in quiet waters at or near the surface.

Flathead catfish have very little harvest protection in much of the Upper Mississippi River watershed. Harvest of flathead catfish is unlimited (no angler limit) in the Mississippi River waters of Iowa and Illinois. Minnesota waters of the Mississippi River bordering Wisconsin have a limit of 10 fish with no size restrictions while Wisconsin waters of the Mississippi allow 25 fish with no size restrictions. The Minnesota River has a more restricted harvest of two fish with one fish over 24 inches. Reduced harvest and protection of large individual flatheads may be a means of increasing predation on bigheaded carps should they become established here or reducing the likelihood of their establishment.

Flathead catfish stocked in Richardson Lake in Southern Minnesota reduced the abundance of common carp by 90% (Davis 1985).

Flathead catfish are capable of consuming common carp up to 30% of their own body weight. Since flatheads can grow to well over 50 pounds, they are capable of eating all but the largest bigheaded carp.

(top) A flathead catfish, a predator capable of eating adult bigheaded carps that are prevalent in the Mississippi River. *Credit DNR Fisheries.* (bottom) Two blue catfish sampled below Mel Price Dam (the downstream most dam on the Mississippi River). Blue catfish migrations are largely blocked by this dam. *Credit USFWS Fisheries.*

Blue catfish are known to feed on bigheaded carps as well (Chapman, personal communications 2013). Blue catfish may have historically existed in Minnesota waters but have very poor success passing the lock and dam system (Tripp et al. 2013). The world record blue catfish was taken below Mel Price Dam, the downstream-most dam on the Mississippi River, where concentrations of large blue catfish are prevalent. The previous blue catfish record was caught in the Missouri River by an angler using silver carp as bait. Several early blue catfish records have been questioned based on the similarity between blue and channel catfish. Based on the lengthy migrations documented for this species in free-flowing systems and the early fragmentation that occurred, blue catfish may have been historically presence in Minnesota. Building fish passage facilities on the locks and dams of the Mississippi river may reestablish their presence as well as other likely native predators on bigheaded carps.

Lake sturgeon have been shown to consume zebra mussels and may be an important biological control (Eggleton et al. 2003) Sturgeon recovery has even been linked to the abundant food supply provided by zebra mussels (Boyd Kynard, USGS, personal communications). Zebra mussels were the most prevalent food item in lake sturgeon over 35 inches long in Lake Oneida (Cornell University Poster). They were also an important food item of sturgeon in the St. Lawrence River (Guilbard et al. 2007). Barriers are a primary cause in the declines of lake sturgeon. Removal of barriers coupled with reintroduction has been a DNR strategy for sturgeon recovery since the mid-1990s. Lake sturgeon are increasing in abundance in the Red River of the North Basin, which includes zebra mussel infested lakes in the Otter Tail River Watershed.

Harvest management of important predators for the life stages and habitats of bigheaded carps is a potentially important strategy for suppression and control of these introduced species. Reconnection of river stretches to enhance native species access throughout the system will maximize their ability to suppress bigheaded carps at all vulnerable life stages.

Pelicans feeding on silver carp at Mel Price Lock & Dam 26. *Credit Bill Rudden.*

Summary:

Of the approaches *targeting introduced carps*, population suppression is likely the only viable long term strategy. Eradication methods are notoriously ineffective, lethal to native species, and are increasingly controversial to stake holders. Removal, selective pathogens, pheromones, and the use of life-history intervention strategies (for example: triploidy) may become effective when used in targeting life stages and population recruitment. All of these strategies will demand a continued and consistent investment. Harvest could become self-sustaining if the species and market became established.

Strategies to improve water quality and restore habitat and connectivity can reduce conditions favorable to introduced carps while *benefiting native species*. This, in turn, can increase competition or predation controls on introduced species. Predation controls can be exerted on all life stages by reestablishing the suite of native species. For instance, bluegills have been shown to limit common carp reproduction by consuming eggs and larvae. Many native riverine species have been shown to eat the vulnerable eggs and larvae of pelagic spawners. Ecosystem restoration allows rivers to develop and maintain habitat diversity and connectivity, which supports the native aquatic community. This strategy is inherently ecologically sound; the system is restored such that it becomes a self-maintaining governor of biodiversity.

Management Implications and Recommendations

Minnesota DNR's Responsibilities

Minnesota Statutes charge the Commissioner of the Department of Natural Resources with doing all things necessary to preserve, protect, and propagate desirable species of wild animals (MN Statute 97A.045, Subd 1.(a)). The commissioner also may, in consultation with the commissioner of agriculture and the executive director of the Board of Animal Health, control nonnative species posing a threat to wildlife, domestic animals or human health (MN Statute 97A.045, Subd. 1. (b)). In recognition of the profound impact of human activity on the interrelations of all components of the natural environment, Chapter 116D of MN Statutes outlines a declaration of the state's responsibilities in terms of environmental policy. This declaration includes preserving important existing natural habitats of rare and endangered species of fish, plants and wildlife, and includes necessary protective measures where appropriate (MN Statute 116D.02, Subd. 2. (10)). These statutes generally provide the DNR with the responsibility and the authority for addressing the concerns over the introduced carp. Given the complexity and potential consequences of the bigheaded carps issue, that duty must integrate considerations of the ecological, economic, and social aspects into a strategy for the long-term health of these systems and constituents.

Management Implications

The Department of Natural Resources is expected to evaluate legitimate risk and respond logically with ecologically sound strategies. The use of strategies known to damage native species is not a logical strategy when the protection of native species is a primary objective. Definitive damages to native species as a result of barriers far exceeds that attributed to bigheaded carps. Restoration and reconnection of natural habitats and improving water quality benefit native species while reducing conditions favorable to tolerant invasive species. An ecologically sound solution is likely to have the most successful outcome in the long term.

Potential Approaches

There are essentially two approaches to addressing aquatic introduced species:

(1) a focus designed to control the introduced species, based on their life history, with requirements that these approaches do not harm native species and

(2) a focus designed to alleviate constraints on native species, via their life history or habitat requirements.

Proposed Strategy

Based on a) the systematic review of current science, b) the conditions on the Minnesota River, and c) the dramatic need for maintaining native species habitat everywhere, we see a clear need to focus on option **(2)** an approach with the emphasis on maintaining and restoring healthy conditions for native species. This will not be easy or quick. Effective prevention and control of biotic invasions require a long-term, large-scale strategy, rather than a tactical approach that focuses only on battling individual invaders (Moody and Mack 1988, Simberloff et al. 1997). "An underlying philosophy of such a strategy should be to establish why nonindigenous species are flourishing in a region and to address the underlying causes rather than simply destroying the currently most oppressive invaders" (Mack et al. 2000). This document provides the background to establishing a scientific basis for addressing these causes and focuses on the long-term health of the native species and the ecosystem. System management, rather than species management, ought to be the focus (Mack et al. 2000). Fortunately, this is a major underpinning of the Department's current emphasis on watershed management.

Options to be Avoided

Barriers should not be an option on free flowing streams. Fragmenting river systems through construction of dams or other barriers, and the subsequent loss of connectivity and habitat (inundated or blocked upstream) are the major constraints on aquatic communities in the world today. Additional effects associated with introduced species, such as bigheaded carps, are actually promoted by dams, regardless of how well-intentioned. Bigheaded carps thrive in eutrophic lake systems connected to large rivers. Impounding large rivers creates lentic (lake-like) river segments and initiates a series of changes, including those in nutrient cycling, temperature regimes, and increases in cyanobacteria production that favor bigheaded carps. Existing carp barriers, discussed in Chapters 3 and 4, have been ineffective for controlling common carp, but have caused the upstream extirpation of up to 60% of the native fish species. If our goal is to protect native species and maintain the health of their populations, then maintaining (and improving) access to the entire stream network (especially to large sections of critical, rare, natural habitat, such as Patterson Rapids and Minnesota Falls rapids on the Minnesota River), must be a priority and part of a larger effort to restore watershed health.

Specific Alternative Approaches Supported by Science:

- Protection and restoration of connected networks and stable river channels, including their associated riparian areas, is a strategy that benefits native biodiversity by way of sediment regimes, nutrient cycling, water temperature regimes, and native species dispersal - all of which are fundamental characteristics of ecosystem resilience and health.
- Combinations of population suppression techniques that target vulnerable or key life stages and are guided by knowledge of life history requirements and behavior are viable options. Examples of these efforts include the use of pheromones to increase trapping or harvest of adults or seining of young-of-year fishes in nursery habitats whose location is based on validated modelling.
- Management strategies that are designed to maximize native predation. For example, restrictions on harvest of flathead catfish in the Minnesota, St. Croix, and Mississippi rivers would help to increase the abundance and size structure of these important predators on carp.
- Barriers are a potentially viable option on artificial connections including ditches and locks through continental or watershed divides or at natural barriers (e.g., St. Anthony Falls Lock) that would not damage native assemblages.

Final Remarks

There is currently an active fervor for stopping the spread of introduced carps. This document lays out specific decisions and alternatives to address the concern over carp using watershed management and targeting watershed health. The science behind these approaches is presented to make abstract ecological principles tangible, such as preservation of biodiversity. We conclude that the introduced carp are symptomatic of fragmented, eutrophic river systems; and addressing the root of this problem is the most viable, sustainable strategy. It will not be quick or easy; it *will* take time and a concerted effort. If Minnesota is serious about addressing concerns associated with introduced carps, economic investment is likely inescapable, regardless of what strategy is chosen. The benefit of investing in the above approaches, especially related to restoration, is that watershed health is the focus *and* the by-product.

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Minnesota Department of Natural Resources,
Division of Ecological and Water Resources

Appendix

Watershed Health Assessment (WHAF) Framework Results; Minnesota River Basin

April 2015

Appendix

Overview

The Watershed Health Assessment Framework (WHAF) uses five components to view natural systems. Like looking through different colored lenses, each of the five components brings a different perspective. This framework provides a consistent approach for exploring the complex landscapes that impact watershed health. A suite of watershed health index scores have been calculated that represent many of the important ecological relationships within and between the components. These scores are built on statewide GIS data that is compared consistently across Minnesota to provide a baseline health condition report for each of the 81 major watersheds in the state. The Watershed Health Assessments consist of health scores that rank the condition of Minnesota's watersheds from 0 (poor health condition, red) to 100 (good health condition, green). The index values are benchmarks, like blood pressure, heart rate or age; they show trends across the landscape that compare health condition and health risks. For more information about the WHAF, the health scores, the individual indices, how they are derived, and more, go to: http://www.dnr.state.mn.us/whaf/index.html**. All of the following graphics in this Appendix, were produced from within the WHAF, either from health score results or as screen grabs from Index Related Features.**

Figures:

Appendix

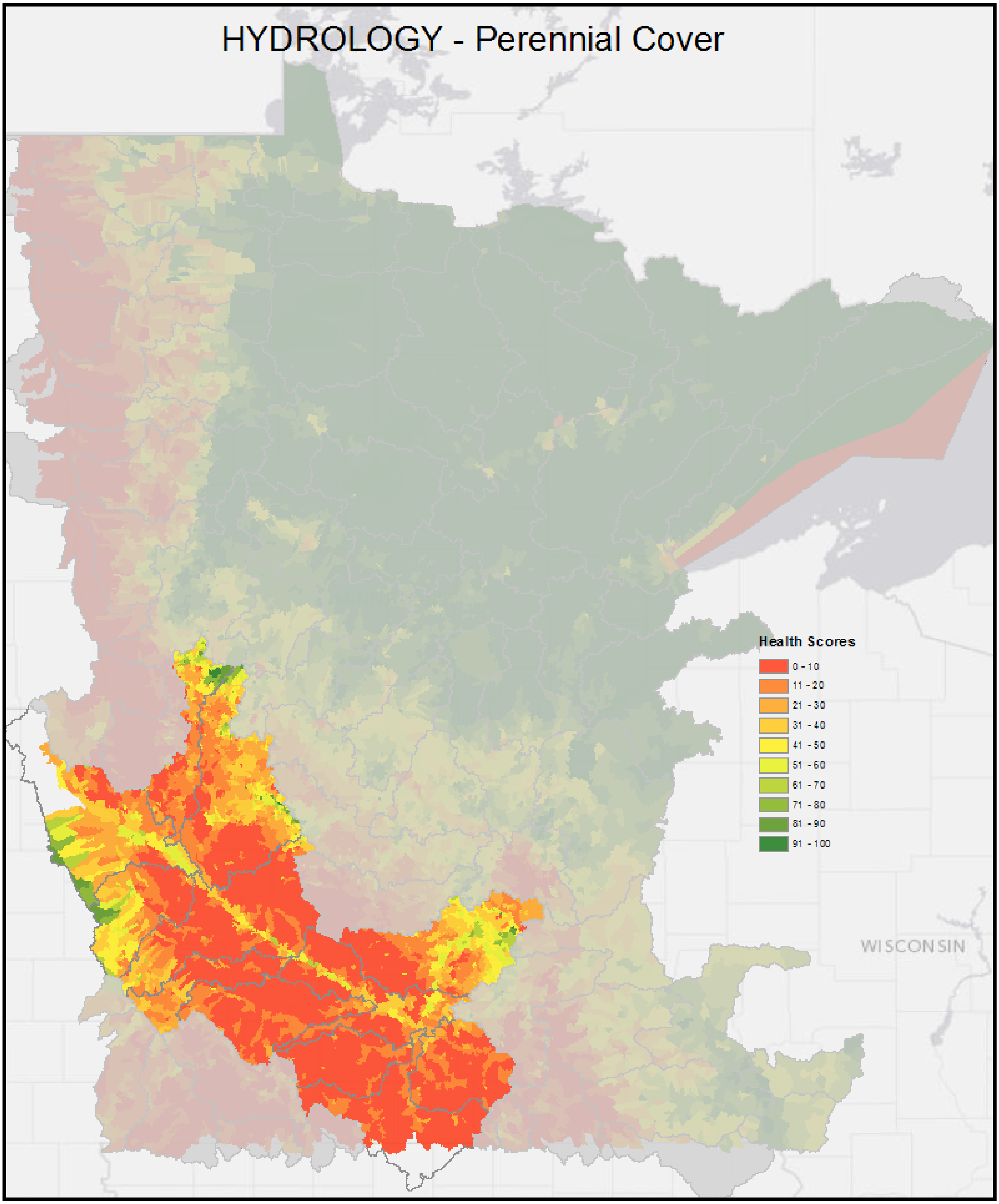


Figure 1. Map of perennial cover scores for lands in the Minnesota River Basin. Perennial cover is permanent vegetation that covers the landscape year-round. Permanent vegetation is removed from land when it is converted to cropland, or developed for human use, such as roads, buildings and homes. This index compares the amount of permanent vegetation that covered the watershed land surface in the 1890s to the amount of year-round vegetation that was measured in 2001.

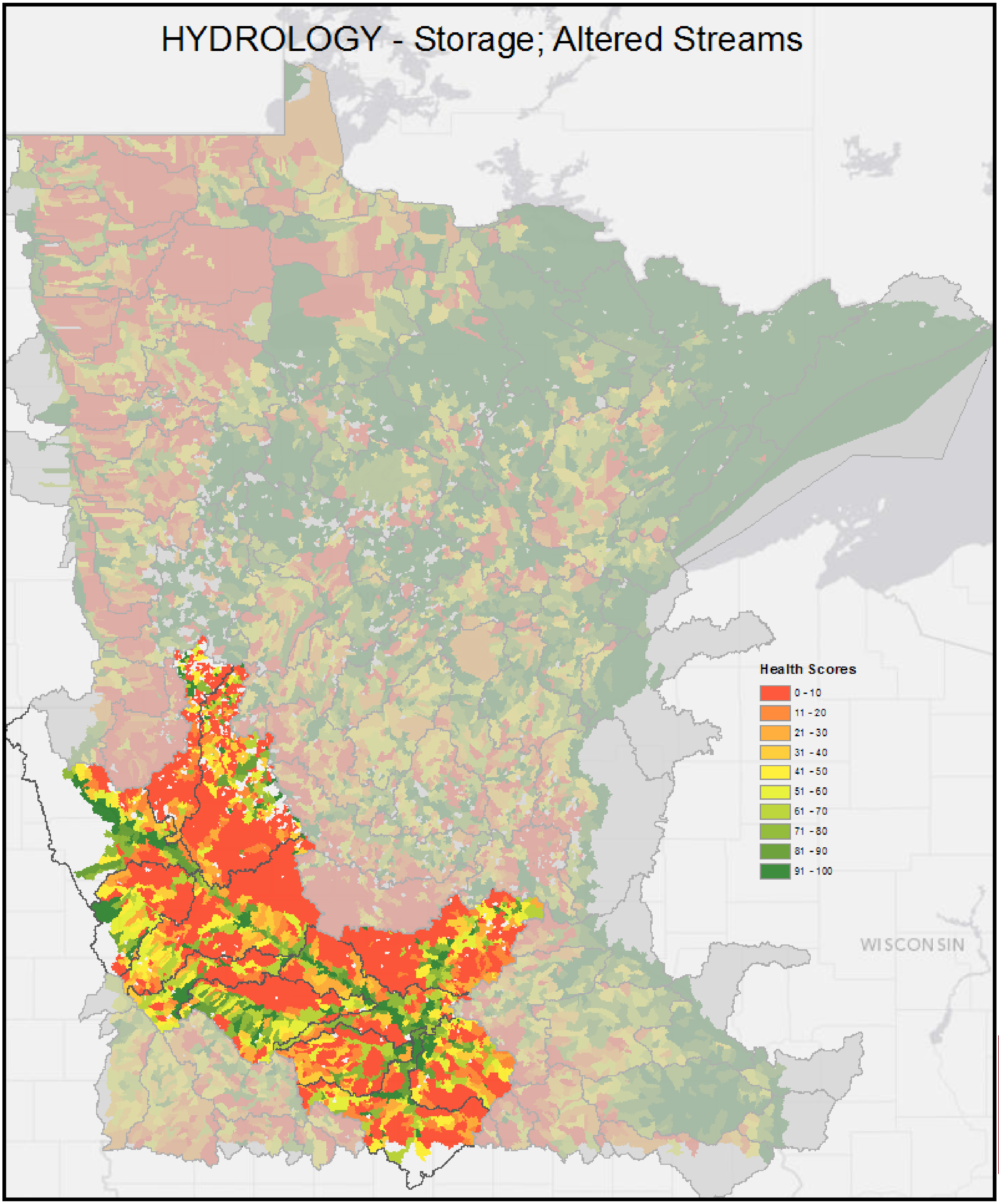


Figure 2. Scores for Hydrologic Storage Index in the Minnesota River Basin. This index represents the extent to which natural streams were straightened by human activity, thereby reducing the hydrologic storage of the land. It is based on the "altered watercourse" dataset and refers to the length of stream segments that were altered in relation to the length of those that meander naturally. This index does not represent data on the volume of water stored in these streams. The score, 0-100, represents the percent of stream length that remains unaltered.

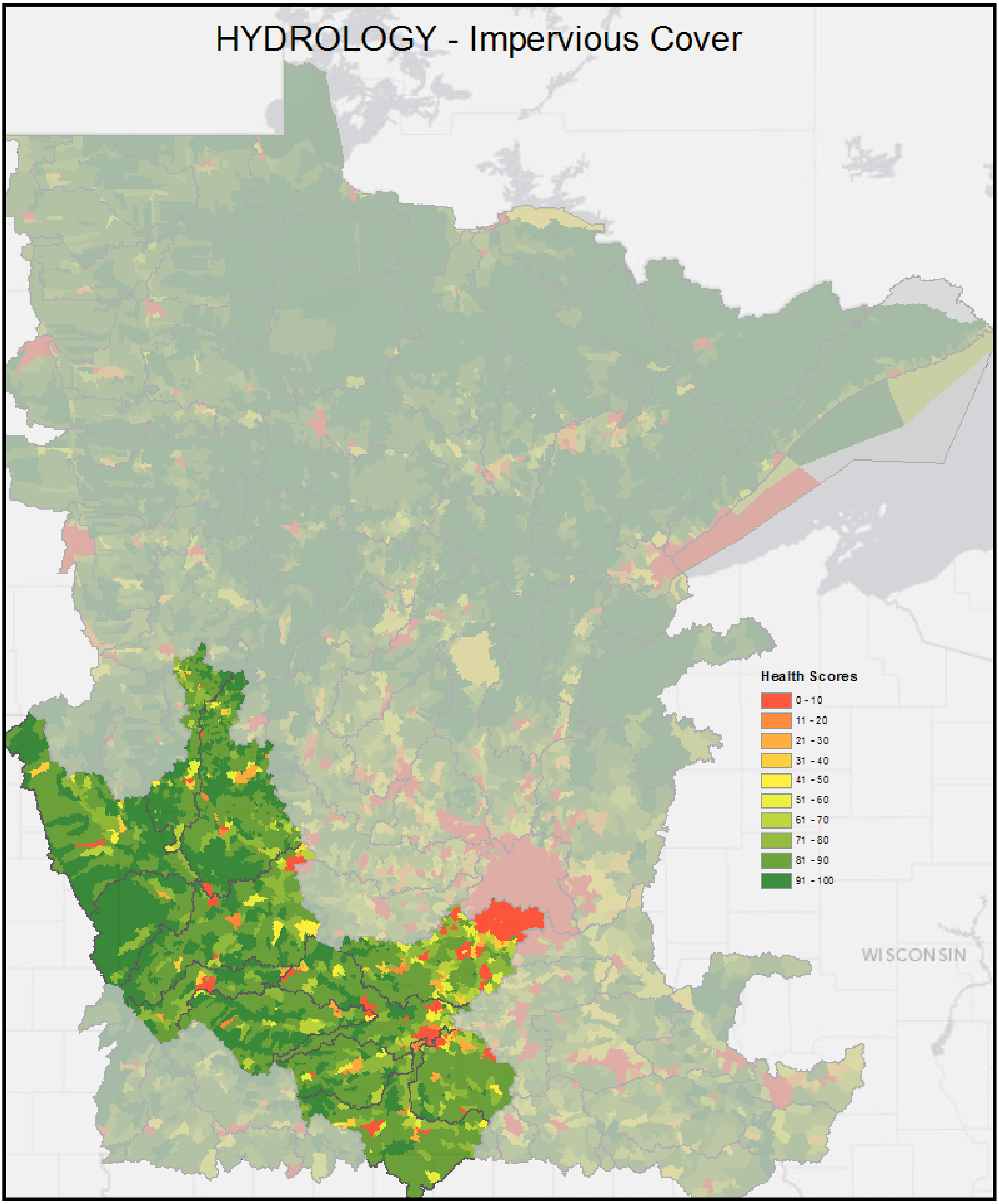


Figure 3. Map of impervious cover scores for lands in the Minnesota River Basin. Impervious cover refers to hard surfaces that do not allow water to pass through into the soil (i.e. roads, buildings, parking lots). Hard surfaces cause water to accumulate, carry impurities and fail to recharge groundwater. This index looks at what percentage of a watershed is covered in hard surfaces. Each small sub-watershed that is more than 4% impervious surface is considered impacted. The percentage of impacted subwatersheds within a major watershed was used to create the index.

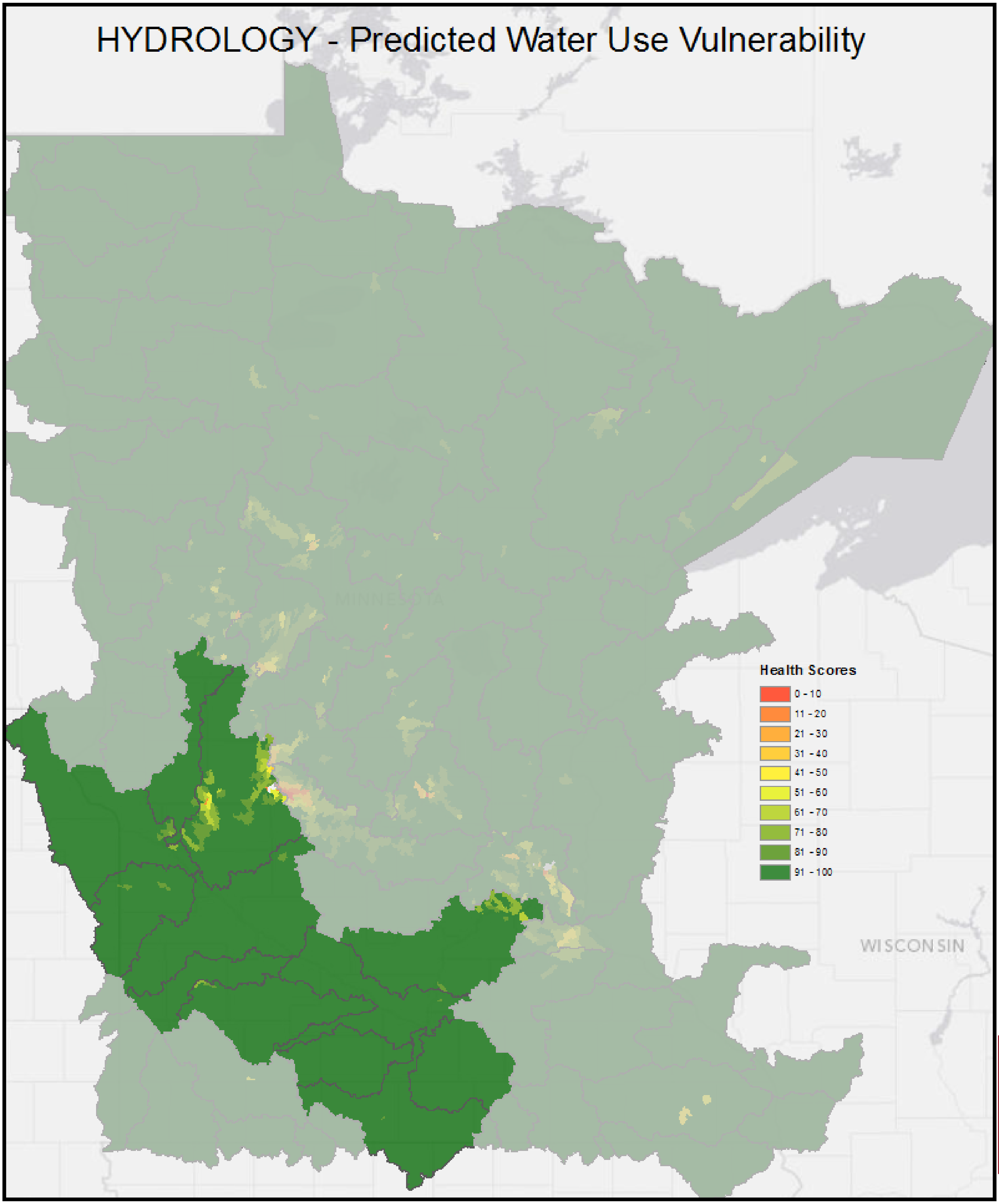


Figure 4. Scores for Water Use Vulnerability in the Minnesota River Basin. This index calculates the use of water in a given catchment and its upstream catchments as a percent of the surface water runoff. Scores are inversely related to the "water use vulnerability index", i.e. water use as percent of runoff. A score of 100 is given to catchments with 0 water use, and a score of 0 is given to catchments where average water use exceeds runoff.

Appendix

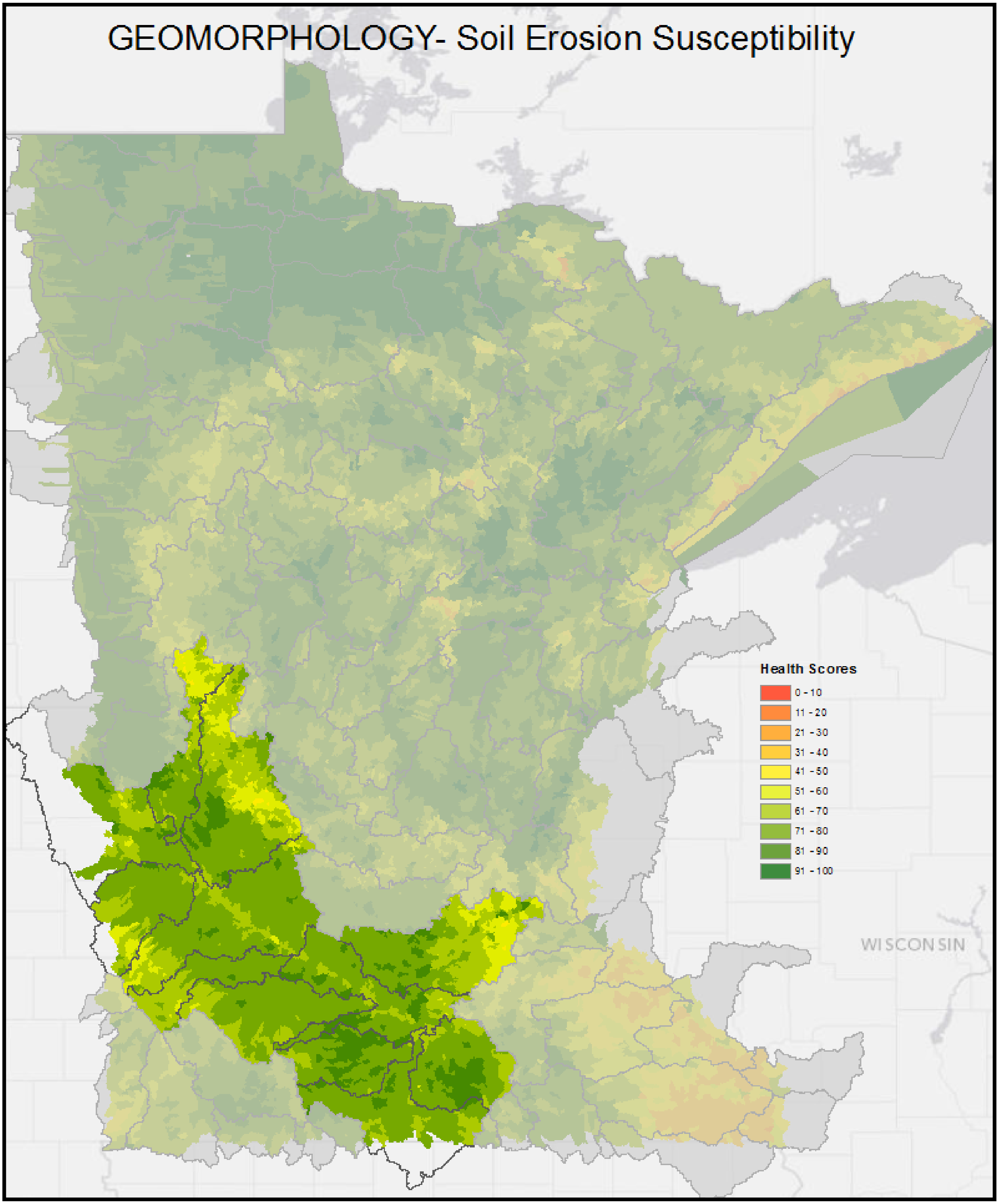


Figure 5. Soil erosion susceptibility in the Minnesota River Basin. Soil erosion is the loss of surface material due to water, wind or other natural forces. Different soil types are more or less erodible due to attributes like particle size and parent material. The Soil Erodibility Index calculates the amount of soil present in each watershed classified as an 'erodible' soil type, weighted by the steepness of the slope on which it is found. The index reflects only soil properties and not the land use or land cover in the watershed.

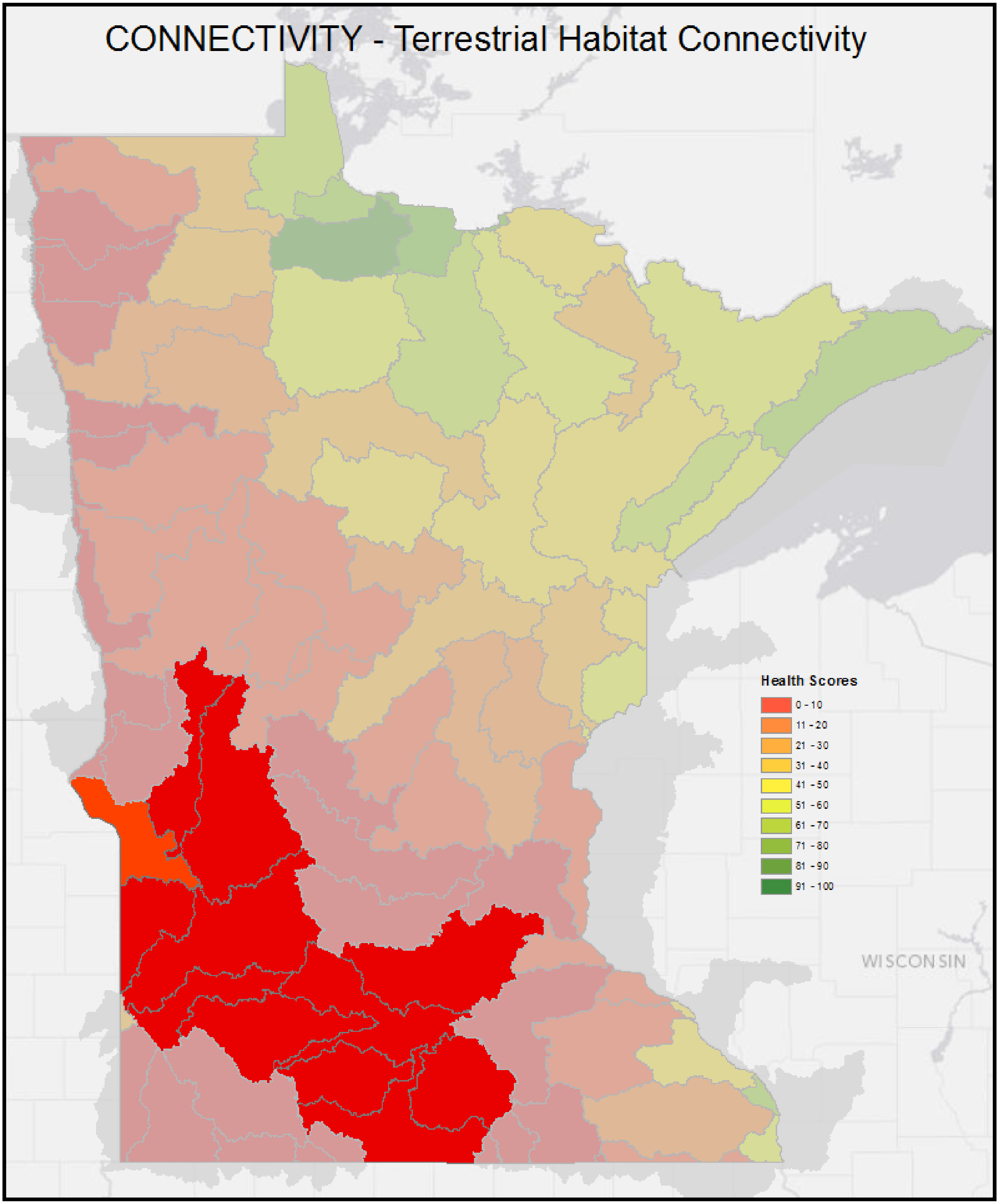


Figure 6. Scores of terrestrial habitat connectivity for lands in the Minnesota River Basin. The connections between patches of terrestrial habitat add value to the habitat and allow energy and organisms to move across the landscape. The Terrestrial Habitat Connectivity Index uses a computer model to rank the ability for organisms to move from one habitat patch to another based on the land cover type. A highway is difficult to cross, a prairie is not. The amount of land area that provides habitat and habitat connections is compared to the land area that is not suitable habitat.

Appendix

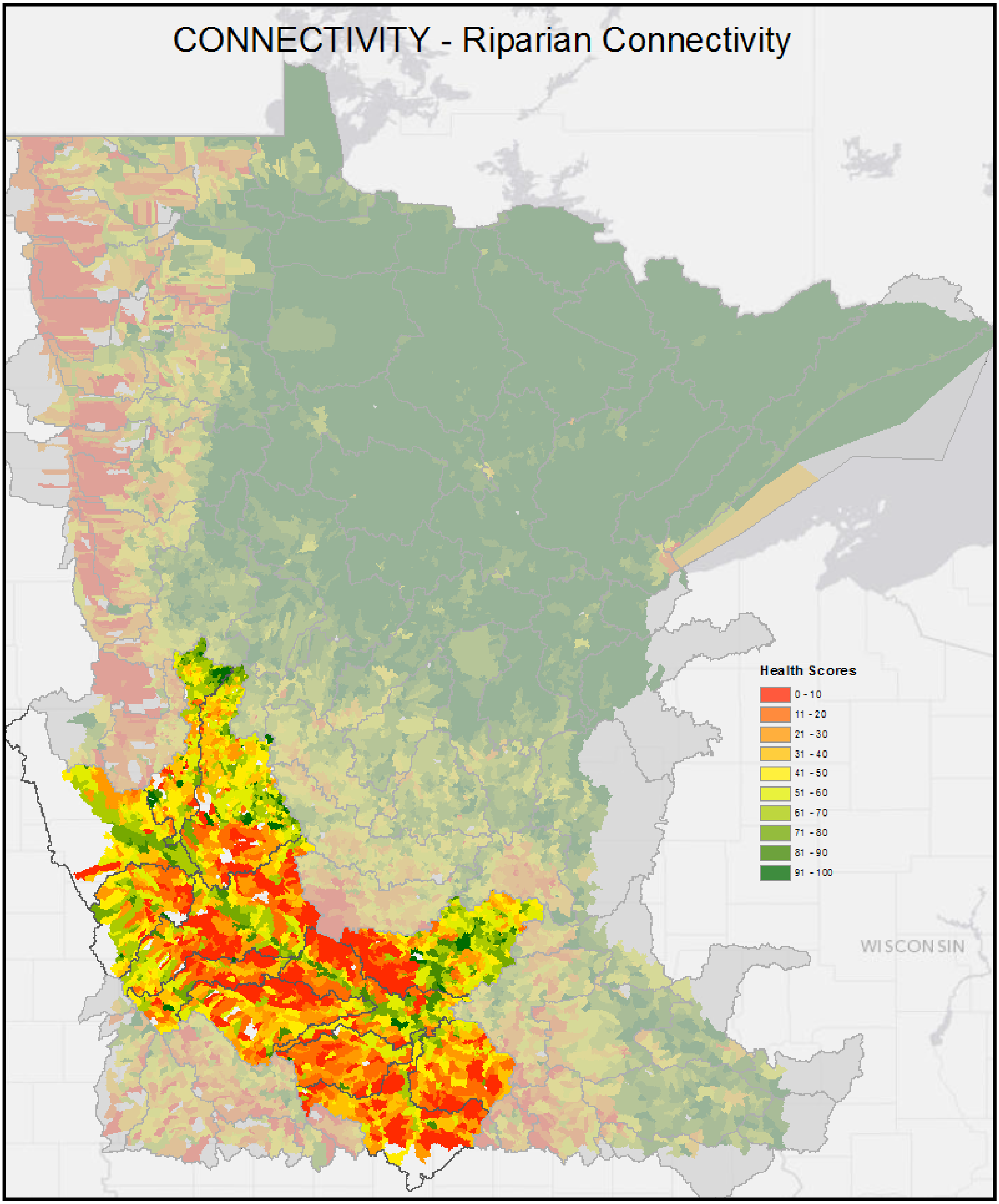


Figure 7. Map of riparian connectivity in the Minnesota River Basin. Riparian' refers to the land immediately adjacent to water features such as lakes and rivers. Access to this area is important to aquatic and terrestrial species particularly during seasonal high flow or flood events. Riparian lands are also important year round as travel corridors and habitat connectors, often providing the only remaining natural land cover in developed landscapes. The Riparian Connectivity Index compares the amount of cropped or developed land cover to the amount of open land in the riparian area.

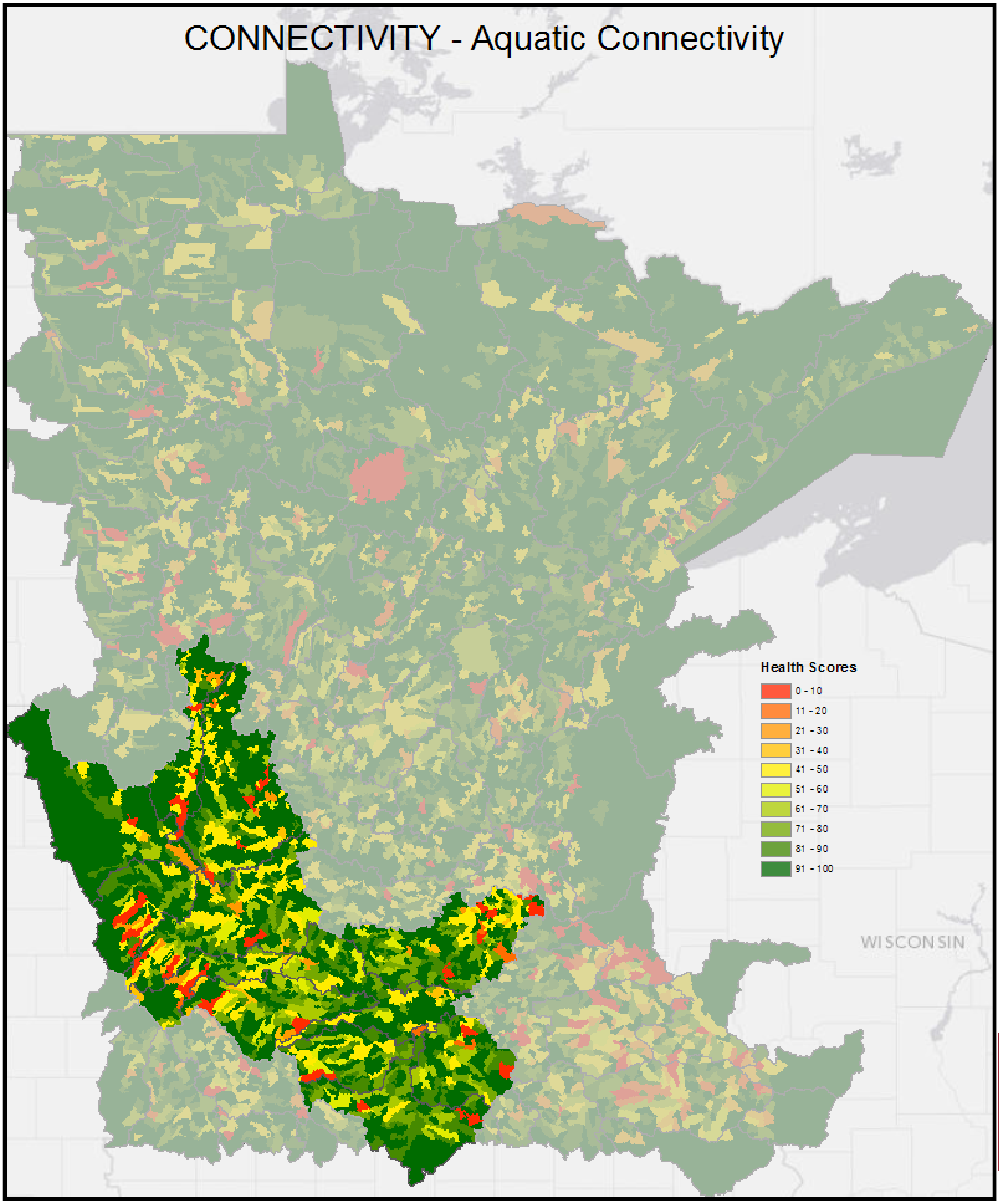


Figure 8. Aquatic connectivity in the Minnesota River Basin. Man-made structures can limit the ability of water, organisms and energy to flow through aquatic systems. The Aquatic Connectivity Index is based on the density of culverts, bridges and dams in each watershed. The higher the density of structures limiting the free flow of water, the lower the Aquatic Connectivity score.

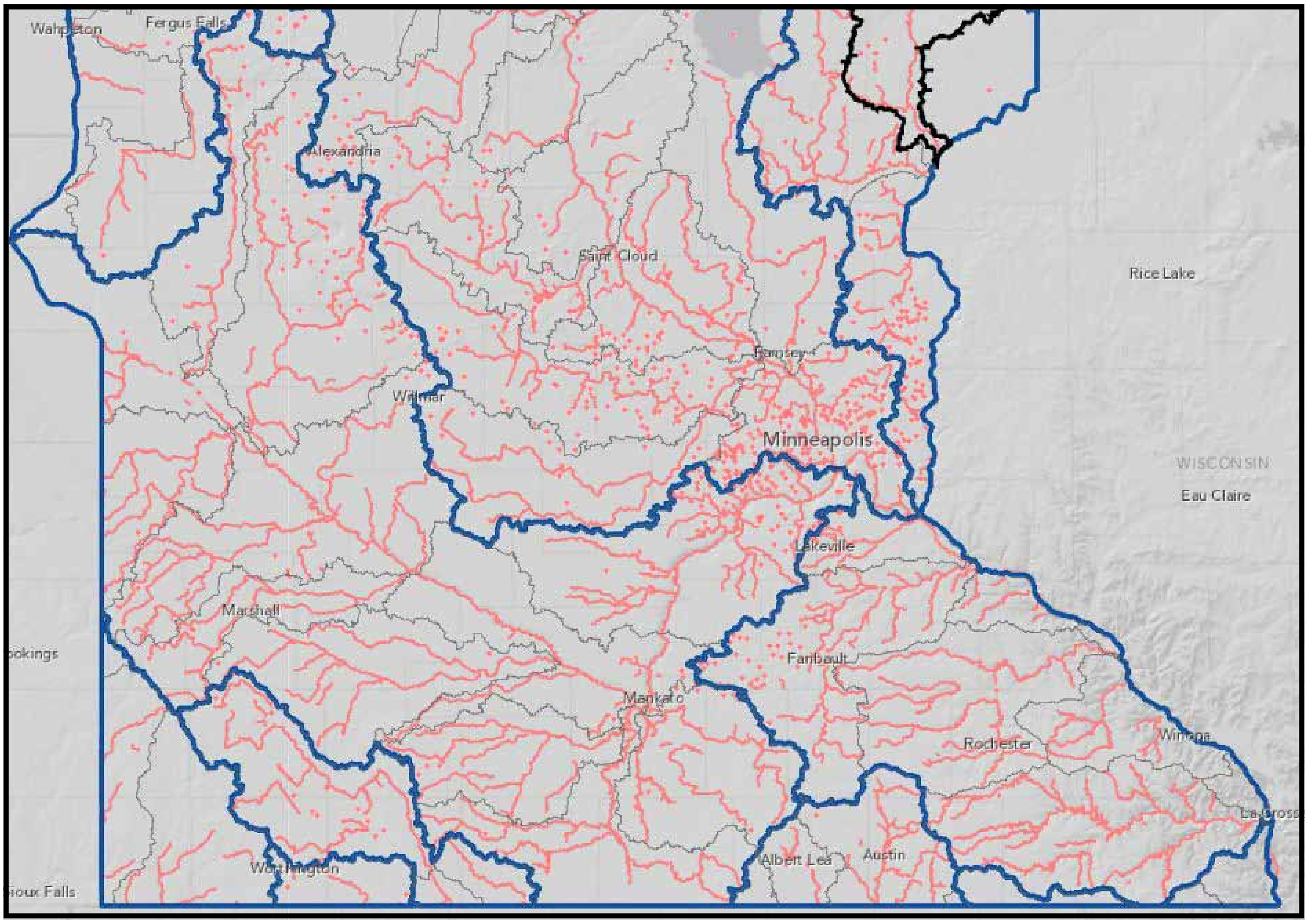


Figure 9. Map of Minnesota River Basin showing the Impaired Waters (in pink), as designated by Minnesota Pollution Control Agency. Many of the Impairments in this Basin are related to turbidity and/or fecal coliform.

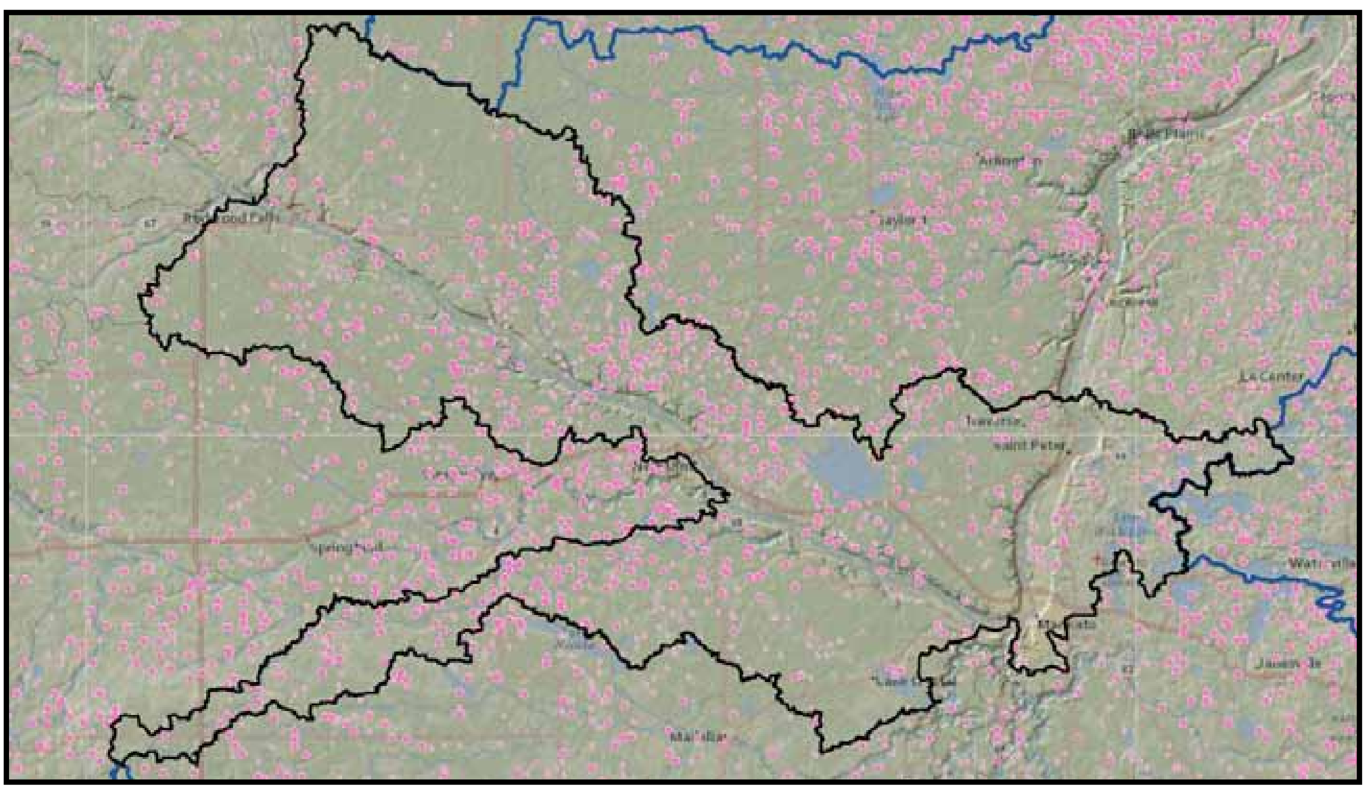


Figure 10. Feedlots in the Minnesota River at Mankato major watershed. There are a number of feedlots throughout the Basin.

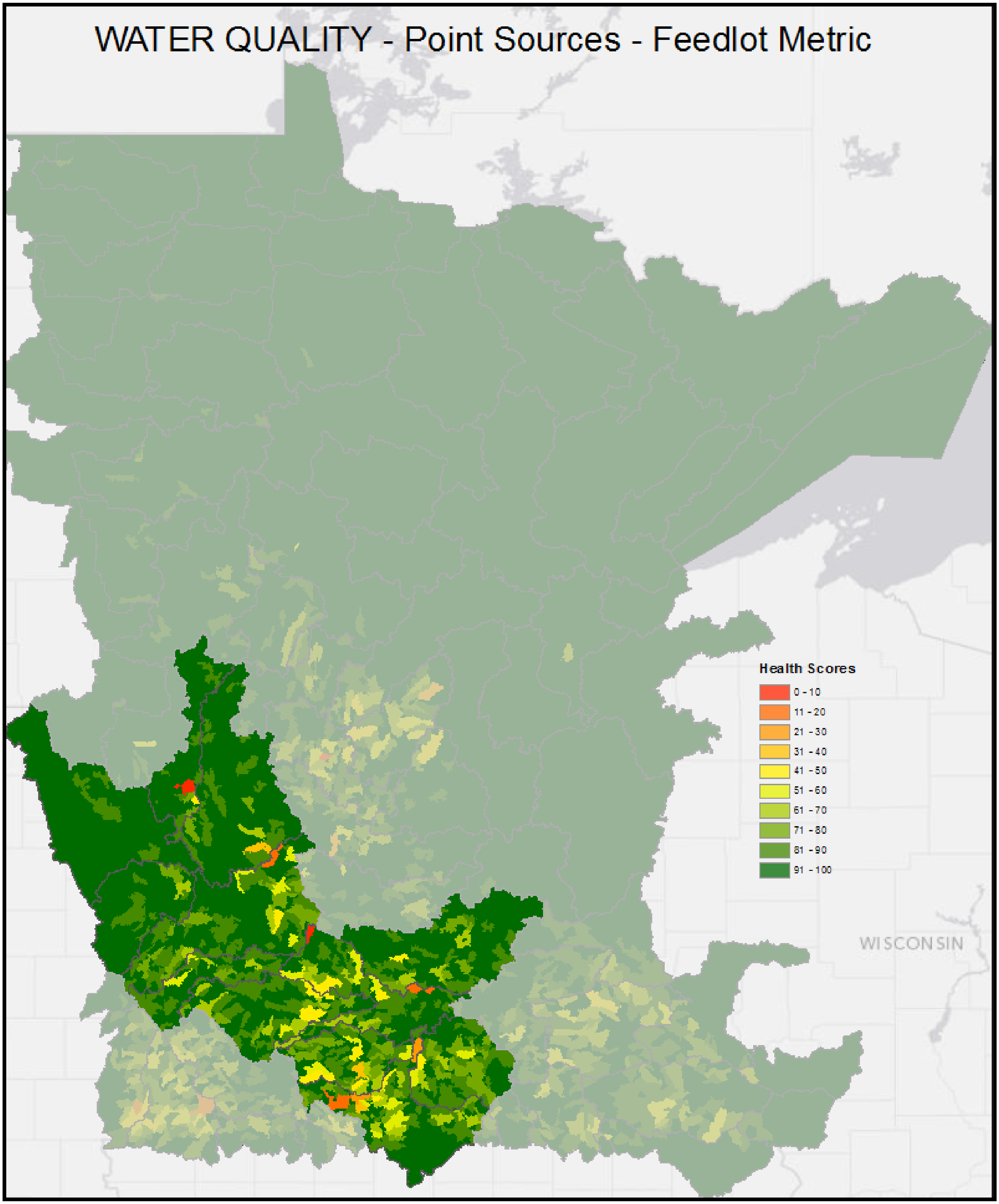


Figure 11. Scores for Animal Units Metric for lands in the Minnesota River Basin. This metric, which is part of the point-source index, scores catchments and watersheds based on the number of animal units in feedlots within their area. A score of 0 was given to the catchment with the highest number of animal units, and 100 to catchments without any animal units.

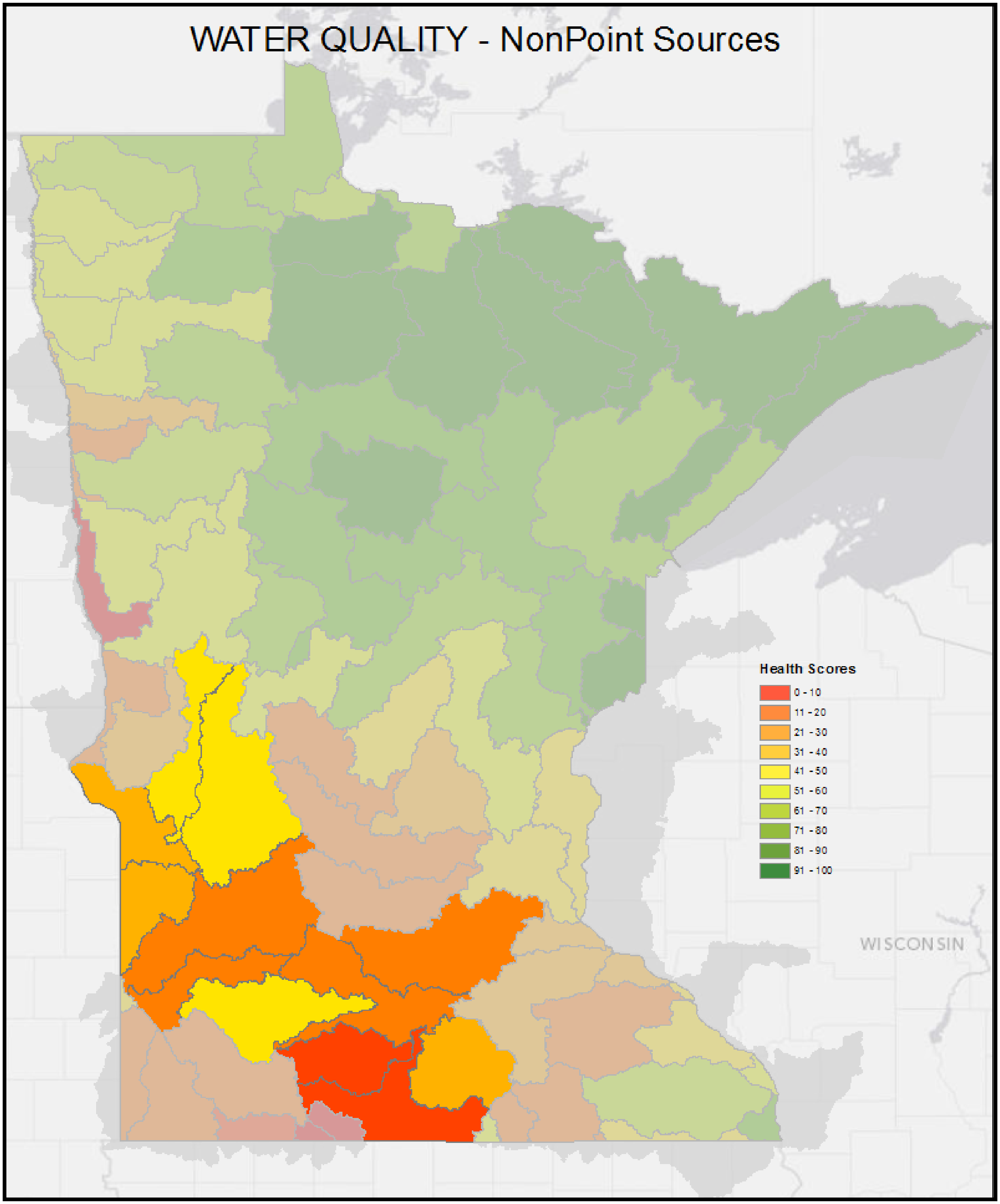


Figure 12. Scores of non-point source pollution for lands in the Minnesota River Basin. Distributed sources or potential sources of pollution to surface or groundwater that are not associated with a specific location are referred to as 'non-point sources'. For example, stormwater runoff carrying contaminants from urban or rural landscapes would be a non-point source. The Non-Point Source Index measures and combines two metrics: the rate of chemical application to cropland and the amount of impervious surface in the riparian zone.

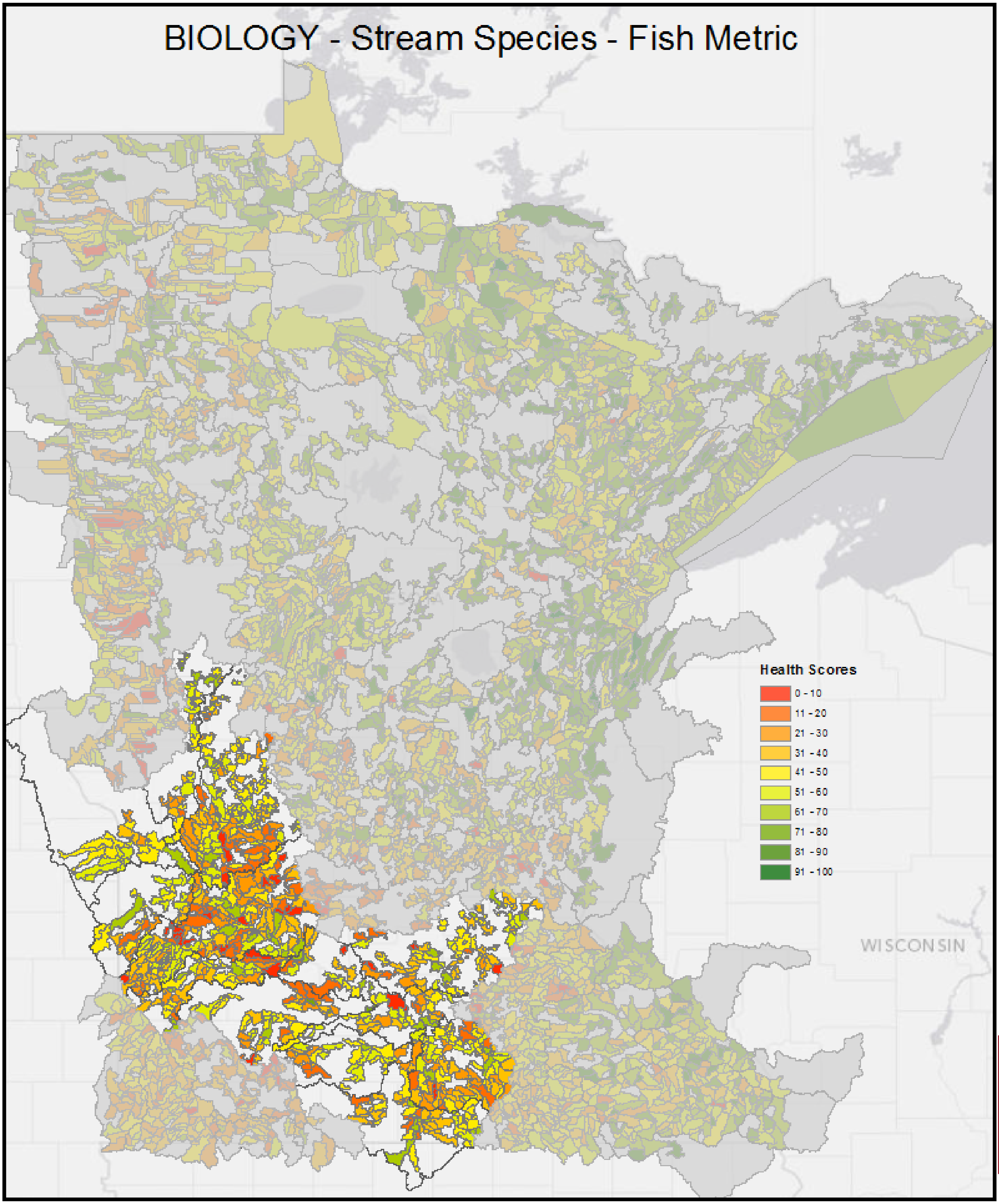


Figure 13. Fish Index (IBI-based) scores in the Minnesota River Basin. This index is based on the fish IBI (Index of Biotic Integrity) published by the Minnesota Pollution Control Agency. IBI site scores were transformed to a 0-100 scale, whereby the "threshold" score value determined by the PCA represents 50; site scores that are lower than the threshold value were transformed to a score between 0-50, while higher scores were transformed to a score between 50 and 100. Catchment scores represent an average of fish IBI scores in a given catchment.

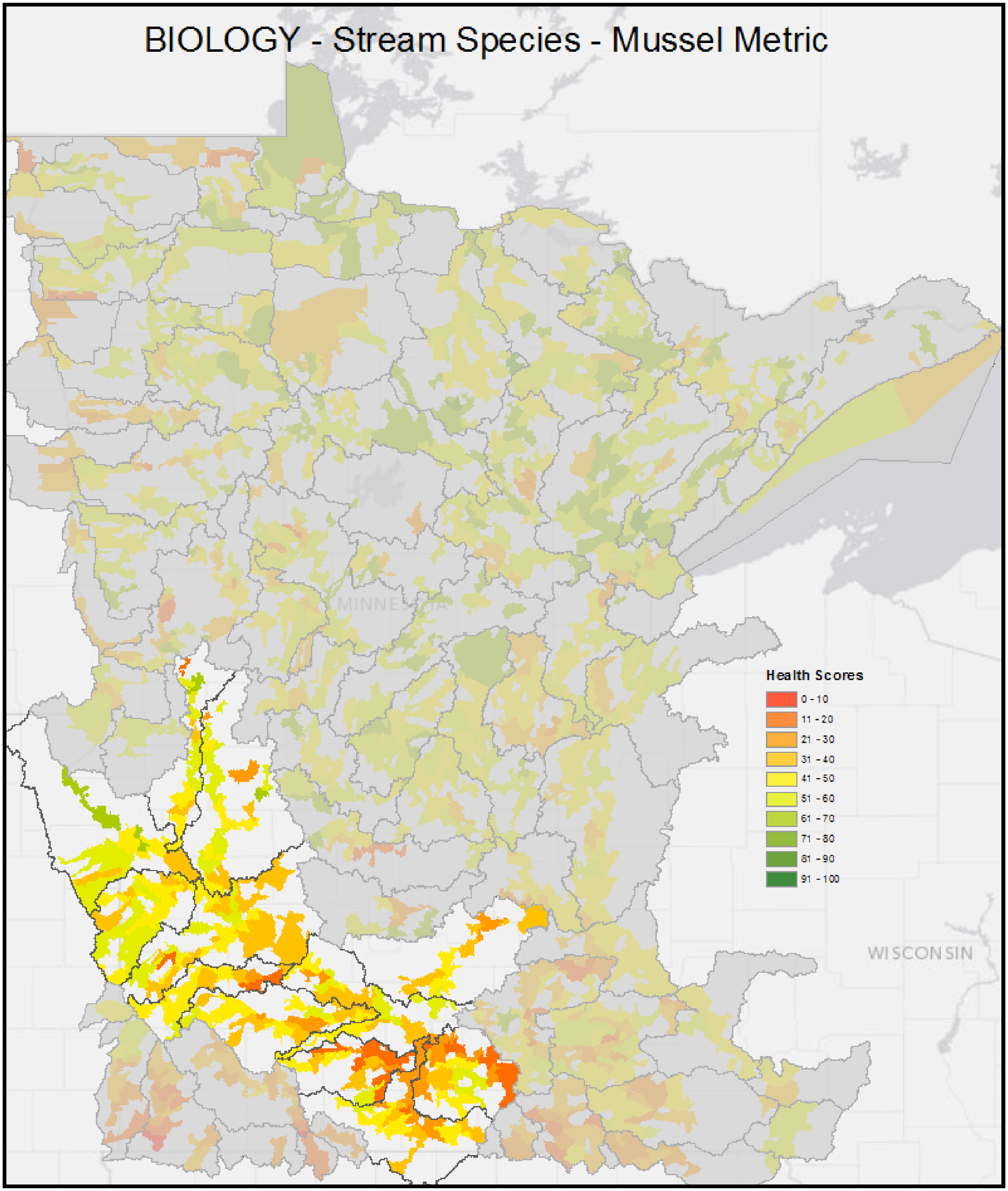


Figure 14. Scores for the Mussel Quality Metric in the Minnesota River Basin. This metric is based on the results of the MN DNR statewide survey of mussels in Minnesota's major streams and rivers. Each survey site has a site quality score that combines four underlying population measurements. These four measurements are scored on a 0-100 scale: count of live mussels per minute spent searching, recruitment (presence of juvenile mussels), percent sensitive mussel species, and percent of species present found live. These four metrics are averaged together to create a mussel site quality score. A low score is red, highest scores are green, as indicated by the legend.

Appendix

Minnesota Department of Natural Resources, *Division of Ecological and Water Resources*

Barrier Effects on Native Fishes of Minnesota

Luther Aadland

April 2015

MNDNR

Attachment

Abstract

To evaluate the effects of barriers on aquatic biodiversity, fish distributions upstream and downstream of 32 barrier dams on the mainstem or tributaries of the Mississippi, Minnesota, St. Croix, St. Louis, Missouri, and Red River of the North were assessed. Recolonization was assessed for eleven dams that were subsequently removed and had adequate post removal surveys. On average, species richness declined by 41% for complete barriers, 37% for near-complete barriers and 20% for barriers that are/were inundated at bankfull flows. A detailed assessment of the Cottonwood River Watershed indicated that a single barrier near the mouth of the river caused a watershed-wide loss of species richness.

Habitat generalists, tolerant (e.g., common carp, fathead minnow, black bullhead, white sucker) lake-oriented, headwater, and widely stocked species were the least likely to be absent upstream of barriers. Sensitive, stream-dependent, and imperiled species were the most likely to be absent upstream of barriers. Blue sucker, mooneye, paddlefish, sauger, shovelnose sturgeon, and flathead catfish were among 27 species absent upstream of all assessed barriers for watersheds in which they were present. A number of small-bodied species, like the carmine shiner, were also sensitive to fragmentation. Channel catfish and freshwater drum, hosts to 13 and 11 mussels were absent upstream of 61% and 64% of barriers, respectively.

Subsequent removal of 11 barriers resulted in upstream recolonization of an average of 66% of the species that had been absent. Removal also resulted in substantially higher catch per unit effort for a number of species, suggesting that an impact of fragmentation is reduced abundance of remaining riverine species. Removal of the Appleton Dam on the Pomme de Terre River resulted in recolonization of elktoe, deertoe, and plain pocketbook mussels; species that had been found only as dead shells in surveys prior to the dam's removal. These findings suggest that barrier dams, while often ineffective for control of common carp, are among the most profound and definitive causes of native biodiversity losses in Minnesota waters.

The fragmentation of North American Rivers is extensive with more than 87,000 U.S. dams over 6 feet high registered in the 2013 National Dam Inventory. Of these 1,078 dams are fragmenting Minnesota streams. Additional small dams, impassable culverts, and other barriers further fragment rivers and streams throughout the nation.

The effects of dams on fish migrations and the decline of migratory species have been acknowledged for over 300 years. In France, design of fish passage facilities began by the 17th century (McDonald 1887; Rajararnam and Katopodis 1984). In North America, conflicts between dam builders and commercial fisherman became intense by 1780 with the "shad wars" as new dams extirpated anadromous American Shad from East Coast rivers (Watson 1996). In Minnesota, Woolman (1895) recommended installation of fish passage for all dams. Most of this early awareness of barrier effects was centered on anadromous game species, such as salmon (those that migrate from the ocean to freshwater or upstream to spawn).

A number of more recent studies have associated barriers with the extirpation of strictly freshwater species and with reduced biodiversity in the North Central United States and Canada (Aadland et al. 2005; Santucci et al. 2005; Catalano et al. 2007). Santucci et al. (2005) found higher fish IBI scores, higher macroinvertebrate condition index scores, higher quality habitat, and more consistent compliance with water quality standards in free-flowing reaches of the Fox River, Illinois than was found in impounded reaches.

Migration of fish is associated with spawning; optimal foraging; seasonal changes in habitat needs and accessing winter habitat; and recolonization following drought or water quality related mortality. Migration may be especially critical in northern latitudes due to harsh winter conditions that can a) cause anoxia, reduction of habitat volume, super-cooled water, frazil and anchor ice and b) result in increased stress, prevalence of disease, and mortality. For example, the majority of species found in a west central Minnesota watershed were observed making seasonal migrations through fishways on the Otter Tail River and fish densities of all species in an upstream reach declined substantially in mid-winter suggesting downstream migration out of the reach (Aadland 2010).

As with fish, the role of dam construction in the decline of mussels has been acknowledged for over a century. In an assessment of mussels in Minnesota, Wilson and Danglade (1913) state, "A dam or natural fall, impassable for fish, may mean the entire absence of mussels in the river above." Dam construction has been cited as the primary cause of all recent (roughly 20 species) mussel extinctions in North America (Haag 2009). North America is analogous to tropical rainforests in terms of mussel species richness, with more species than any other continent, but 71.7% are listed as special concern, threatened, or endangered (Williams et al. 1993). The ecological implications of mussel declines are extensive due to their roles in stabilizing stream beds (Zimmerman and de Szalay 2007), increasing diversity of other benthic invertebrates (Gutierrez et al. 2003; Spooner and Vaughn 2006), and water filtration (Newton et al. 2011).

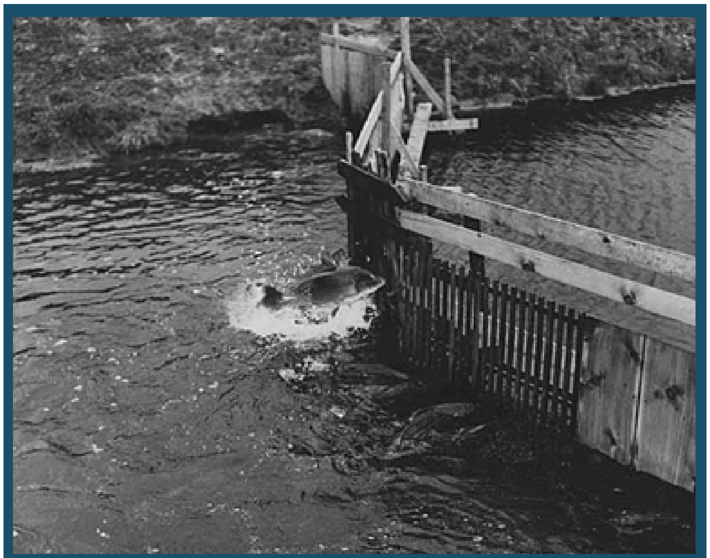
In addition to the loss of biodiversity, dam construction and fragmentation have also been shown to increase the prevalence and dispersal of aquatic introduced species. Johnson et al. (2008) found invasive species to be 2.4 to 300 times more likely to occur in reservoirs than in natural lakes. For example, the Illinois River has been channelized, has had severe water quality impairments throughout its history, and is entirely impounded by dams. It is also believed to have the highest densities of silver carp in the world, which became established in the river around 2000 (Sass et al. 2010).

For clarity, we are defining a species as **native** (indigenous) if its presence is the result of only natural processes, with no human intervention. In contrast, a species is **introduced** (non-native, alien, exotic, non-indigenous) if it is living outside its native range and has arrived there by human activity, either deliberate or accidental.

Diagnosis of barriers as the cause of reduced biodiversity is verified where barriers have been removed and species recolonize (Garvey et al. 2012). Kanehl et al. (1997) found moderate declines in carp abundance and major increases in smallmouth bass abundance following removal of the Woolen Mills Dam, Wisconsin. Removal of the Stronach Dam, Michigan resulted in recolonization of 8 species found only downstream of the dam and an increase in abundance of 18 of 25 species sampled (Burroughts et al. 2010). The removal of dams has increased recently due to structural instability of aging dams and increased awareness of the ecological damages associated with them (Aadland 2010).

The introduction of common carp in the 1880s and later declines in their popularity initiated construction of fish barriers as early as 1927 (Hoffbeck 2001). Subsequently, numerous carp barriers have been constructed across Minnesota including dams, electric barriers, screens, and high velocity culverts. These provide the opportunity to evaluate barriers targeting common carp in terms of effects on common carp and native assemblages.

Since the effects of introduced carp and other aquatic introduced species on native species is a primary cited concern, the evaluation of barriers on native species is fundamental to evaluating the efficacy of barriers as an introduced species deterrent. Nationally, most studies have focused on the effects of barriers on game species with relatively few evaluations of the effects of barriers on aquatic biodiversity.



A fish screen on Six Mile Creek near Lake Minnetonka in 1965. *Credit Minnesota Historical Society.*

The Methods

The Effects of Dams on Fish Diversity

As a means of addressing the effects of barriers on native fishes in Minnesota, the presence/absence of fish species in the upstream versus downstream watersheds of 32 dams throughout Minnesota was analyzed. The dams assessed are, or were, located in tributaries and mainstems of the Minnesota, Red River of the North, St. Croix, St. Louis, Missouri and Mississippi river watersheds (Figure 1). Geo-referenced fish records from the Minnesota DNR-Fisheries, MN DNR-Ecological and Water Resources, Pollution Control Agency, university collections, the Bell Museum, and other reliable sources were used to tabulate the presence and absence of fish above and below the barriers. Much of the data is available through the Department's "Fish mapper" tool (Fish Mapper website: www.dnr.state.mn.us/maps/fom/index.html) but more recent Stream Habitat Program and Fisheries records were acquired directly from Area Offices.

Dams that are frequently inundated and passable during high flow conditions were not included in this assessment. Of the 32 dams assessed, nineteen were complete blockages, nine were near-complete blockages (may be passable during 10 year or larger floods), and four were moderate blockages (may be passable during 2 year or larger floods). Two of the complete blockage dams were built on natural barriers, Redwood Falls and St. Anthony Falls. Fourteen of the dams have been subsequently removed or modified for fish passage and safety.

Major floods can inundate even relatively large dams making them passable for a brief yet key period of time; therefore, the results needed to be put in context for the occurrence of these large floods. Many dams also have experienced partial or complete failures during their existence - some dams have failed multiple times. Flood and failure events were considered in the analysis. Inundation may or may not create passable conditions for a long enough duration or at the right time of year for recolonization by a given species.

Only the downstream-most major barriers on the chosen tributaries were assessed. Several rivers had a series of closely spaced dams with little or no sampling effort in between them so the potential affect by each barrier could not be assessed.

Since fish records comprised a wide range of gear types and sampling effort, sample abundance was not quantified in the analysis and was handled as "present" or "absent". While presence/absence data handling was necessary, barriers can substantially reduce population size without extirpating the species entirely or major floods may allow a few individuals to pass. As a result, many species identified as "present" may not represent viable populations.

Unfortunately, for most cases, the historic pre-barrier species diversity and abundance is unknown because dams were built as early as the 1850s which pre-dates fish sampling by trained fisheries biologists or taxonomists.

For each barrier dam fish distributions were handled on a watershed basis upstream and downstream. If there were records of a species within the contributing watershed upstream of a barrier, it was considered "present". The exceptions to this were a couple of cases where a native species was known to have been stocked in a relatively isolated lake in the watershed but was absent from the rest of the basin, it was considered "absent".

Only species found in the river or tributary being assessed were included in the analysis as potential species for that tributary. Species found in larger mainstem rivers downstream were not included in the analysis for that tributary. This was done to avoid inclusion of species that may require larger river habitat that may not exist in the tributary. In several cases this limited the list of potential species where dams were close to the mouth of the tributary because few samples were collected between the barrier and the mouth.

Downstream effects on fish diversity were not quantitatively assessed due to the complexities of assessing effects attributable to a single barrier. Migration barriers have caused downstream basin-wide extirpations when they block access to critical spawning habitat. Large rivers, however, may have multiple tributaries that provide suitable spawning habitat so effects were evaluated only for the tributary watershed.

Distribution after removal or failure of a dam was also assessed for some structures to separate habitat or water quality effects from those attributable to the barrier. Since most dam removals have been relatively recent, several tributaries have had no surveys since removal. For most sites, significantly less sampling effort was available post-removal than for pre-removal. Pre and post dam construction records of species that were absent upstream following dam construction

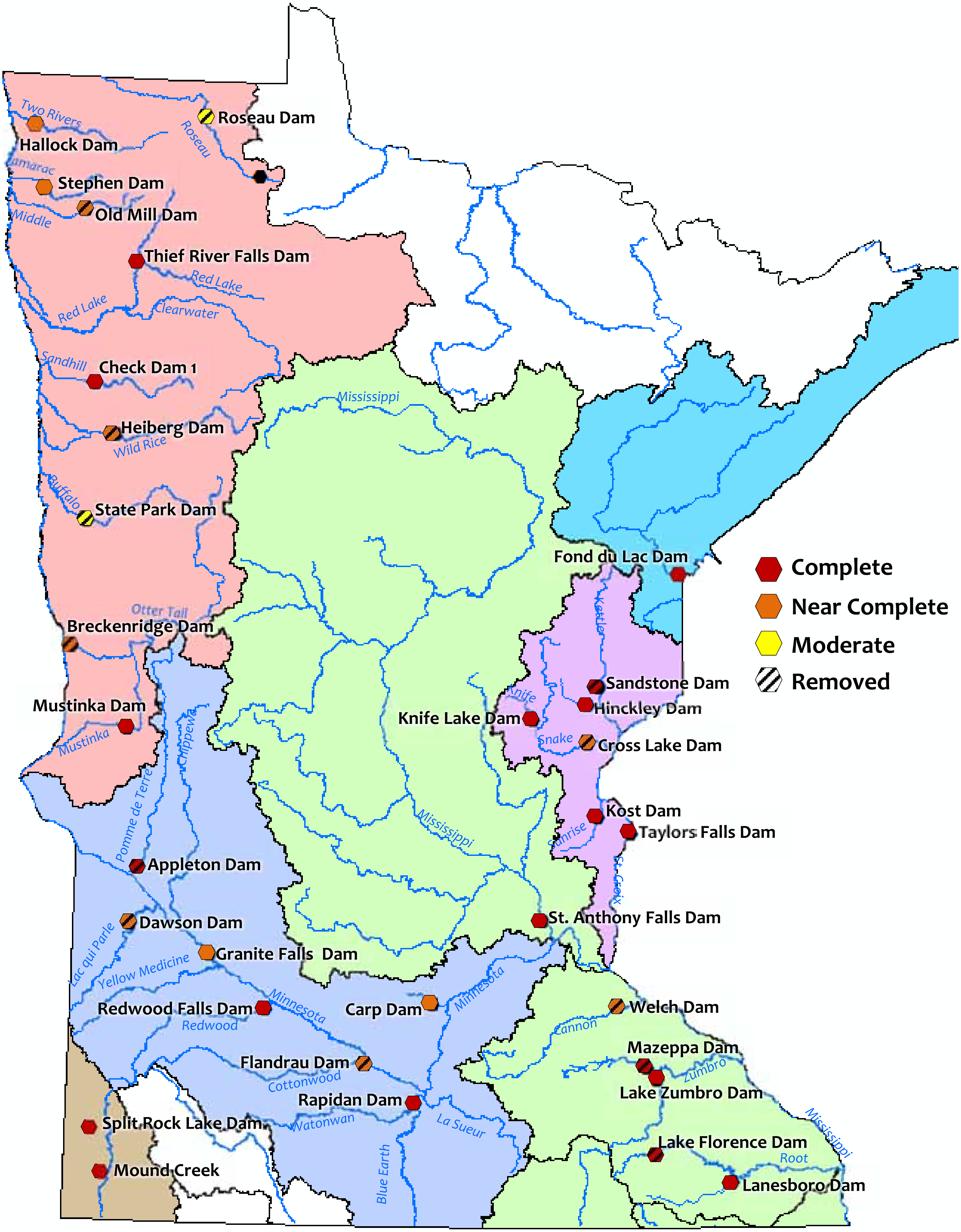


Figure 1. The locations, effectiveness, and current status of the 32 dams included in the barrier assessment.

were also considered as evidence that the barrier-caused the extirpation.

Relative vulnerability of species to barrier-caused extirpation was assessed as a percentage of watersheds where they were present in the watershed but were not found upstream of the barrier. This was put in the context of habitat, thermal regimes, introductions or stocking and other factors. Relative vulnerability was also assessed as a function of environmental tolerance (tolerant/intolerant species) and imperiled status (special concern, threatened, and endangered).

The Effects of a Dam on Watershed Scale Fish Diversity

To address relationships between watershed area, biodiversity, and barrier effects, a detailed assessment of the Cottonwood River Watershed was completed. Flandrau Dam, originally built in 1937 near the mouth of the Cottonwood River, blocked most of the watershed from the Minnesota River. The dam failed in 1947 and was rebuilt the following year but a number of fish surveys were conducted in 1948 during the time when the dam was passable. The dam also failed in 1965 and 1969 but was rebuilt each time and no available fish surveys were conducted upstream of the dam site during these dam breaches. The dam was finally removed in 1995. This dam and fish sampling history provided assessment of a short duration open river condition followed by nearly 50 years of fragmentation then a final period of surveys following the dam's removal. Watershed area and stream mile distance from the mouth of the Cottonwood River were measured for each site and associated with general habitat type and species composition.

Results and Discussion

Barrier Effects on Upstream Fish Diversity

Of the 32 barriers evaluated, an average of 37% (3% to 78%) of the species sampled in the watershed were absent from collections upstream of the barrier (Table 1 and Table 2). The fish records analyzed included a total of 150 species including 16 non-native and 134 species that are considered native to Minnesota. The extent of species absent upstream was higher among the more effective barriers.

Table 1. Summary of Barrier Effects on Species Richness

Barrier Effectiveness	# of Dams Assessed	Average % Absence
Complete	19	41%
Near Complete	9	37%
Moderate	4	20%
Overall Average	***32***	***37%***

The percentage of species absent above natural barriers at St. Anthony Falls (50%) and Redwood Falls (36%), which have likely been barriers for thousands of years, were within the range of that for complete barrier dams (15-73%). This suggests that barrier-caused extirpation can happen within a short time frame (decades). Rivers upstream of natural barriers tend to have lower species richness. It is unknown if absent species were never able to colonize upstream of the barrier or if some fish species were historically there then extirpated.

The absence of a species from surveys upstream of a barrier has several potential explanations:

1) The species was extirpated as a result of the barrier.
2) The species is present but was not collected in the surveys.
3) The upstream reach lacks suitable habitat for the species.

Significant sampling effort, a diversity of habitat upstream of the dams, and the abrupt upstream extent of the species at the dam site favors barrier-induced extirpation as the explanation of species absences for most sites and most species. However, a number of factors need to be considered in determining whether the upstream absence of a species is attributable to the barrier or if habitat, water quality, stream size, temperature regimes, hydrology, statistical probabilities, or other

Attachment

Watershed	Barrier Name Year Built, Year Removed	Dam height at low flow (ft) Barrier Effectiveness	Watershed area (mi²) upstream of dam / total % of watershed upstream of dam	Total # of native species observed in watershed Additional introduced species	# of Native MN species absent upstream of barrier (% of total)
Red River of the North Basin					
Otter Tail River	**Breckenridge Dam** 1935. Replaced with rock ramp in 2007	8 Near Complete	1,910 / 1,952 97.8%	75 1	9 (12%)
Mustinka River	**Mustinka Dam** 1940	18 Complete	163 / 861 18.9%	30 1	15 (50%)
Buffalo River	**State Park Dam** Pre-1893, 1937 Removed in 2002	3.5 Moderate	325 / 975 33.3%	58 1	21 (36%)
Wild Rice River	**Heiberg Dam** 1875. Removed in 2006	8 Near Complete	934 / 1,560 59.9%	61 1	16 (26%)
Sand Hill River	**Check Dam 1** 1955	10 Complete	308 / 420 73.3%	36 1	15 (42%)
Red Lake River	**Thief River Falls Dam** 1946	16.75 Complete	3,450 / 5,680 60.7%	64 3	13 (20%)
Middle River	**Old Mill Dam** 1886, 1938. Removed in 2001	8.5 Near Complete	225 / 779 28.9%	32 1	25 (78%)
Tamarac River	**Stephen Dam** 1975	12 Near Complete	283 / 397 71.3%	37 1	9 (24%)
Roseau River	**Roseau Dam** 1932. Replaced with rock ramp in 2001	5 Moderate	474 / 1,420 33.4%	44 1	10 (23%)
South Branch Two Rivers	**Hallock Dam** 1938	8 Near Complete	592 / 1,100 53.8%	42 1	13 (31%)
St. Croix River Basin					
St. Croix River	**Taylors Falls Dam** 1890, 1907	50 Complete	6,240 / 7,650 81.6%	106 5	31 (29%)
Snake River	**Cross Lake Dam** 1800s, 1938, 1963. Modified with rock ramp in 2013	2 Moderate	974 / 1,009 96.5%	68 1	2 (3%)
Knife/Snake River	**Knife Lake Dam** 1983	14 Complete	92/1,009 9.1%	68 1	33 (49%)
Kettle River	**Sandstone Dam** 1908. Removed in 1995	20 Complete	868 / 1,060 81.9%	64 5	22 (34%)
Grindstone River	**Hinckley Dam** 1955	10 Complete	77 / 1,060 7.3%	64 5	30 (47%)
Sunrise River	**Kost Dam** 1885	13 Complete	268 / 283 94.7%	64 2	19 (30%)

Table 2. Watersheds assessed for barrier effects on fish species richness. Barrier effectiveness is based on dam height and frequency of inundation by floods; Complete = complete barrier, Near Complete = near complete barrier that may be passable during large floods (10-year or larger), Moderate = moderate flood barrier that may be passable during moderate floods (2-year or larger).

Watershed	Barrier Name Year Built, Year Removed	Dam height at low flow (ft) Barrier Effectiveness	Watershed area (mi^2) upstream of dam / total % of watershed upstream of dam	Total # of native species observed in watershed Additional introduced species	# of Native MN species absent upstream of barrier (% of total)
Lower Mississippi River Basin					
Mississippi River (upstream of Iowa border)	**St. Anthony Falls Dam** 1848, 1963	49 Complete	19,100 / 65,000 29.4%	127 8	64 (50%)
South Branch Root River	**Lanesboro Dam** 1868	28 Complete	284 / 1,250 22.7%	93 4	57 (61%)
North Branch Root River	**Lake Florence Dam** 1857. Removed in 1993	12 Complete	119 / 1,250 9.5%	92 4	65 (70%)
Zumbro River	**Lake Zumbro Dam** 1919	55 Complete	845 / 1.150 73.5%	89 4	27 (30%)
North Fork Zumbro River	**Mazeppa Dam** 1922. Removed in 2001	20 lowered to 10 Complete	174 /1,150 15.1%	89 4	65 (73%)
Cannon River	**Welch Dam** 1900. Removed in 1994	8 Near Complete	1,340 / 1,440 93.1%	82 5	19 (23%)
Minnesota River Basin					
Minnesota River	**Granite Falls Dam** 1911	17 Near Complete	6,180 / 16,200 38.1%	97 4	39 (40%)
High Island Creek	**Carp Dam** 1958	6 Near Complete	206 / 241 85.5%	47 1	30 (64%)
Blue Earth River	**Rapidan Dam** 1910	55 Complete	2,410 / 3,486 69.1%	66 1	26 (39%)
Cottonwood River	**Flandrau Dam** 1937, Was repeatedly damaged by floods & was removed in 1995	28 lowered to 12 Near Complete	1,310 / 1,313 99.8%	65 2	24 (37%)
Redwood River	**Redwood Falls Dam** 1902	34 Complete	630 / 665 94.7%	53 2	19 (36%)
Pomme de Terre River	**Appleton Dam** 1872. Removed in 1999	13 – 16 Complete	905 / 915 98.9%	65 1	17 (26%)
Lac qui Parle River	**Dawson Dam** 1913. Replaced with rock ramp in 2009	8 Moderate	472 / 1,156 40.8%	41 1	8 (20%)
Missouri River Basin					
Mound Creek	**South Dam** 1936	14 Complete	16.8 / 17.2 97.7%	29 1	9 (31%)
Split Rock Creek	**Split Rock Dam** 1937	24 Complete	45 / 320 13.9%	26 1	10 (38%)
Lake Superior Basin					
St. Louis River	**Fond du Lac Dam** 1924	78 Complete	3.600 / 3,634 99.1%	62 11	9 (15%)

factors are responsible. Conversely, the presence of an individual does not necessarily indicate that the species is unaffected by the barrier or representative of a viable population. A number of species are routinely stocked, masking barrier effects on a population. Ultimately, historical pre-barrier records or those following removal of barriers indicate the ability of a species to exist or thrive in the river reach. These considerations warrant further discussion given their implications for barrier effects.

Considerations in Fragmentation Assessment

Historical Context of Fish Distribution Data It was not possible to comprehensively determine species distributions prior to watershed fragmentation since most of the watersheds evaluated had barrier dams by the mid- to late 1800s and did not have systematic fish surveys until the mid-1900s. Archeological surveys, some early explorers like Alexander Henry (1799 – 1808), George Featherstonhaugh (1835), and others who took detailed notes provide useful historical data on easily identified food fishes like lake sturgeon, walleye, channel catfish, and freshwater drum. Most species were not targeted until much later when biological surveys started. Woolman (1895) surveyed the upper Minnesota and Red River watersheds in the 1890s to 1910s. Surber (1923) primarily surveyed eastern Minnesota streams in the 1920s. However, most fish surveys did not occur until after 1940.

The late timing of initial surveys makes the early distribution data a baseline for a significantly impaired condition, not pre-human influence, in most watersheds. Land-use changes, dam construction, unregulated overfishing, and severe water pollution likely limited or extirpated many of the pre-settlement species prior to any surveys. The Mississippi River was an anoxic "dead zone" from the Twin Cities to Hastings from the about 1885 to the 1980s due to raw sewage effluent until the Clean Water Act and other legislation forced construction of water treatment plants. Release of raw sewage was typical for municipalities located on rivers and streams. The St. Louis, Rainy, and other relatively undeveloped watersheds were heavily polluted with paper mill effluent and massive logging drives. The Otter Tail River had repeated fish kills due to discharges of whey and other cheese by-products into the early 1990s. As a result, it is likely that many species absent from early records probably existed in Minnesota waters prior to these changes. Improved treatment of human waste does appear to be allowing some species to return to Minnesota waters.

Climate change will likely have implications for what species will be here in the future as it has in the past. As a result of the relatively recent glaciation of most of Minnesota and subsequent warming of waters over the past 14,000 years, most of our fish assemblage would have been invaders as thermal regimes and habitat changed. River systems of Northern Europe are less diverse than similar sized rivers in North America due, in part, to the north-south orientation of the Mississippi River that allowed recolonization from southern refugia compared to the East-West orientation of the Danube and other European rivers that would not have had southern un-glaciated refugia (Oberdorff et al. 1997). Under current anthropogenic climate change, southern species may expand into Minnesota waters while cold-water species may decline as thermal regimes change (Stefan and Hondzo 1991). Some species have already shown changes in abundance, northerly extent of range, and timing of spawning attributable to climate change (Schneider 2010).

Species Introductions and Stocking A number of the game and bait species native to Minnesota are widely stocked and this includes water to which they may not have historically been native to. Routine stocking likely masked the effects of fragmentation for walleye, channel catfish, smallmouth bass, and other species. Walleyes are migratory and likely susceptible to effects of fragmentation but are so widely and regularly stocked that these effects are very difficult to assess. Many of these occurrences do not represent viable populations or meta-populations as indicated by the need for ongoing stocking. Stocking is less common where natural reproduction occurs.

Habitat Type, Habitat Diversity, and Length of Free-Flowing River Fish distributions are defined by habitat, which is a function of geology, watershed size, slope, hydrology, climate, and other factors. Habitat also can be defined by temporal (diurnal, seasonal, annual), life stage (spawning, eggs, fry, juvenile, adult) and spatial (microhabitat, mesohabitat, watershed) scales. For many stream fish species, habitat overlaps large spatial areas and includes a diversity of microhabitat types for successful completion of life cycles (Aadland and

Kuitunen 2006). The length of stream required is likely to be dependent on the availability of the full suite of habitats needed to complete each life history stage. Since year to year climate and hydrology can dramatically affect habitat suitability and reproductive success, a network of connected habitats increases resilience to drought and poor spawning conditions.

Lake sturgeon may require 155 to as many as 620 miles of free-flowing river to maintain a healthy population (Auer 1996). Sturgeon have been observed visiting multiple spawning rapids before actually spawning. This likely increases reproductive success as the suitability of individual rapids varies with the flows and water temperatures each year. The fact that the St. Croix River has retained a viable lake sturgeon population upstream of the St. Croix Falls dam may be due to the availability of spawning rapids, large river habitat and considerable length of free-flowing river in the watershed upstream of the dam. However, a number of species have disappeared from the St. Croix and similar watersheds despite the presence of diverse habitats. Blue sucker maintained a presence upstream of the St. Croix Falls Dam until the 1970s but haven't been sampled there since.

Conversely, tolerant, generalized species are often able to maintain populations within much shorter river reaches. For example, common carp, black bullheads, fathead minnows, and a number of other tolerant lake species can complete life histories within a single isolated lake.

Stream and Watershed Size It is logical that large fish would require a minimum stream and watershed size but amazingly large fish are found in small streams and watersheds when they have access to them. For the largest fish species, presence in smaller streams may only occur during spawning and high flows or as juveniles. Large-bodied fish like flathead catfish risk stranding or predator attacks if present or trapped in small streams as flows recede. As shown in the picture below, a large flathead was found stranded in a riffle in the Yellow Medicine River in July 2009. This fish may have been killed by the eagle observed feeding on it. The presence of connected lakes or deep pools in a watershed can provide vital refugia for these large bodied fishes.

Interestingly, the smallest watershed assessed in this study, the Grindstone River (77 mi^2), had historical records of MN's largest fish species, the lake sturgeon (which can grow to 10 feet and 400 pounds), found in Grindstone Lake (20 mi^2 watershed). Since lake sturgeon spawn in rapids, these fish, at some point in their life, would have had to leave the lake and swim up the Grindstone River, which is about 20 feet wide at the lake outlet. Lake sturgeon have been observed spawning in the Moose Horn River where the drainage area is 112 mi^2. The largest paddlefish on record was 85 inches long and weighed 198 pounds. It was speared in Lake Okoboji, Iowa where they were once abundant, but are now extirpated, likely due to barrier dams. Paddlefish also spawn in rivers (riffles and rapids) so would have needed to ascend the Little Sioux River and the outlet creek, which is about 50 feet near the lake outlet (141 mi^2). These small streams and watersheds may be very important migratory pathways as well as spawning and nursery habitat for large-bodied fish, even though spawning adults may only be present briefly during high spring flows to spawn.

Watershed size and the location of the dam in the watershed also had statistical implications due to relative sampling effort. Several of the assessed barriers were near the mouth of the watershed being assessed so that most of the sampling effort and watershed area was upstream of the barrier. The limited number of samples downstream of the barrier results in a low number of potential species listed as "absent" upstream of the barrier (as it reduced the number of potential species considered present in the watershed). For instance, 99.8% of the Cottonwood River's watershed is upstream of the Flandrau Dam site, so only samples from a very short reach downstream of the dam and upstream of the Minnesota River confluence added species to the watershed total that were inferred to potentially exist upstream in the absence of the barrier. Despite the short segment of free-flowing river in the watershed downstream of the dam, 24 species (37% of the watershed total) were collected downstream of Flandrau Dam that were not collected above it.

Partial Barriers Four of the 32 dams assessed in this study are not complete barriers during moderate floods. Furthermore, some of these and others assessed have failed periodically over their history. The occasional flood flows and dam failures potentially allowed individuals of extirpated species to migrate upstream of the barrier. This may explain the relatively intact fish community upstream of the Cross Lake Dam on the Snake River. This dam was only 2 feet high but since it was built on natural rapids with steeper slopes over bedrock, velocities were high during major floods. The fact that only

A dead flathead catfish, apparently killed by a bald eagle, in a riffle in the Yellow Medicine River, July 2009. Fingerling flathead catfish have been caught at this site suggesting that the small river, though generally lacking deep water adult habitat, may be important for reproduction. *Credit DNR Stream Habitat Program.*

2 species were absent upstream of the dam may indicate that fish were able to pass this barrier recurrently during bankfull and higher flows. Lake sturgeon, extirpated above most barrier dams, have maintained a presence upstream of this dam. However, the photo below of sturgeon caught in rapids below the upstream Bean Dam suggest that historic sturgeon populations were much higher. Sturgeon were observed below the Cross Lake Dam unsuccessfully attempting to migrate upstream. It has since been modified for fish passage.

Locks & Dams The lock & dam system on the Mississippi River is a series of partial barriers that provide limited passage through the lock chambers or during high flows when the gates are open. Passage may vary by species and by lock & dam size and height. Tagged silver carp moved upstream through lock & dam #26 through #20, up to #19, during "closed" gate conditions almost as readily as during open gate conditions and were able to pass under gates that were not entirely closed (Brooks et al. 2009). Native species generally had much lower success in passing the dams. Paddlefish and blue catfish were impeded more than other fish species. The near-extirpation of skipjack herring and declines of other native species have been attributed primarily to the construction of the 36 foot-high Lock & Dam 19, which is a complete barrier except through the lock chambers (was completed in 1913 and is located at southern tip of Iowa).

Current fish assemblages of the Upper Mississippi River, and as a result potential assemblages of Minnesota tributaries, are likely limited by the lock & dam system and the associated habitat fragmentation and inundation (when compared to historic assemblages). This is especially significant when the anoxic dead zone between the Twin Cities and Hastings is considered since all current fish and mussel species in that reach would have needed to recolonize after sewage treatment plants improved water quality in the 1980s. The limited passage of native species through the Lock & Dam System likely allows more species to exist upstream that would not be present if they were complete barriers. Improved passage through these lock and dams would allow species like skipjack herring, American eel, paddlefish, and many others to increase in abundance. Conceptual designs for nature-like fish passage through the entire lock & dam system was proposed in 2006. Commercial fisherman described catching large "shovelnose sturgeon" over 50 pounds in Minnesota waters of the Mississippi (Mike Davis, DNR ecologist, personal communications). These likely would not have been shovelnose sturgeon, which do not get that large, but similar looking pallid sturgeon. Blue catfish (for which early records exist), pallid sturgeon and other species that may have been part a free-

A large pile of sturgeon speared in rapids below Bean Dam on the Snake River in 1912. *Photo taken by Herman Schmidt.*

flowing fish assemblage in Minnesota waters may also recolonize with improved connectivity of the Mississippi River.

Presence versus Viable Population The presence of a species upstream of a barrier does not confirm that the population is maintaining a viable population. For instance, surveys on the Red Lake River in 1996 and 2001 each collected a single channel catfish upstream of the Crookston Dam. Surveys following its removal (2005) in 2005 and 2011 collected 222 and 255 catfish respectively. Some long lived species can retain a presence long after functional extirpation. Lake sturgeon can live over 150 years. Large adults were caught in large lakes of the Red River Basin as late as 1947, which is over 50 years after suitable spawning habitat had been largely eliminated or blocked. Some mussel species have been aged to over 200 years so can also retain a presence many years after they can no longer reproduce due to the loss of their host species. Following the definitions used in this study, a single individual caught anywhere in the watershed upstream of a dam precluded the species from being considered “absent”.

Thermal Regimes Many tributaries to the Mississippi River in southeastern MN have cold-water headwater reaches with warm-water downstream reaches. Some of these streams have warm headwaters starting in the plains, followed by cold, groundwater-fed middle reaches through the bluffs, and finally warmer lower reaches near their confluence with the Mississippi. These thermal regimes dictate the presence, range and migratory boundaries of coldwater species during the summer months. During winter, all Minnesota waters are cold and may allow dispersal of these coldwater species to other groundwater-fed streams. Generally, headwater species associated with cold water were not absent above barriers assessed here.

Downstream Effects Downstream effects of barriers on fish diversity were not directly assessed due to the difficulty of determining whether a specific dam was the causative factor. The decline of many species, however, has been attributed to the loss of upstream spawning habitat. Since dams are frequently built in high gradient reaches (Minnesota Falls, Granite Falls, Rapidan, Taylors Falls, etc.) they not only block migrations but inundate these critical habitats. In addition, many are known to make seasonal spring migrations up smaller tributaries to spawn followed by downstream migrations back into the larger river. This short but critical presence in the stream makes them unlikely to be collected, especially since most stream surveys are done in late summer. By eliminating spawning habitat it is likely that many of the barriers assessed have substantial effects on downstream fish communities that, based on observed migration distances, may extend hundreds of miles.

Access to Refugia To maintain populations, species require available microhabitat for all life stages (spawning, fry, juvenile, and adult). They also need to be able to survive droughts and extreme winter conditions that may reduce or eliminate available habitat. Hydrologically stable streams and those with numerous lakes that maintain suitable dissolved oxygen levels through winter in their watersheds (such as the Otter Tail, Red Lake, and Cannon Rivers) generally retained more species upstream of barriers than those prone to low flows or that have few or no lakes. The lakes or stable base flows may provide habitat refugia during drought conditions that would not exist in stream reaches that stop flowing. Lakes that become anoxic in winter, like many in the agricultural watersheds of southern Minnesota, generally do not provide suitable refugia except for species tolerant of very low oxygen. Northern pike have been shown to migrate out of winterkill lakes and into connected streams as oxygen refugia (Tonn and Magnuson 1983). These lake–stream interactions may be very important to sustaining biodiversity in these watersheds.

Relative Vulnerability to Barrier-Caused Extirpation by Species

Of the 32 dams and 150 species evaluated, most native species were found to be vulnerable to extirpation by barriers. All 134 native fish species for which there were records were ranked according to vulnerability to barrier-caused extirpation. This was determined by the percentage of barriers upstream of which they were absent divided by the number of watersheds in which they were present (Table 3. and Table 4. starting on page 28). A total of 27 native fish species were absent upstream of every barrier (100%) for watersheds where they were found. Sixty-six native species were absent upstream of at least half of the barriers for which they were assessed. As already discussed, these results must be tempered by sample size and influence of the factors discussed previously.

The data suggest that imperiled species (special concern, threatened, and endangered) are particularly vulnerable to fragmentation by barriers (Figure 2). Species that have imperiled status in Minnesota and are imperiled or extirpated in adjacent states were most prevalent in the upper quartile of vulnerability (75-100 % absence) to barriers. This is consistent with other studies that have cited dams as a primary threat to imperiled species and native biodiversity (Rinne et al. 2005).

Species designated as "intolerant" or "sensitive" to impairment of water quality (EPA) were also vulnerable to barrier-caused extirpation while "tolerant" species were generally among the least vulnerable. The ability to survive anoxia in eutrophic lakes and agricultural watersheds allows tolerant species to maintain populations through winter and drought while other species must periodically migrate out of these watersheds or are killed. For example, black bullheads held in enclosures in Lake Christina, Minnesota were able to survive both rotenone treatment and anoxia by burying themselves in lake sediments (Thomas Carlson, retired DNR Shallow Lakes Biologist, personal communications). There may be interaction effects in addition to direct barrier effects that are responsible for this trend. The suppression or extirpation of sensitive species and decreased biodiversity due to barriers would give tolerant species a competitive advantage. Thus, tolerant species may actually benefit from fragmentation in some systems. Prominent tolerant species included common carp, fathead minnow, black bullhead, white sucker, and creek chub. These findings are

Table 3. Summary of Barrier-caused Extirpation Fish Data

# of dams/watersheds analyzed	32
# of native fish species present in these watersheds	134
# of introduced species present in these watersheds	16
# of native species absent above every dam for watersheds in which they were present	27 (20%)
# of species listed (Endangered, Threatened, Special Concern) in MN	27 (20%)
# of species listed in MN and neighboring states and province	69 (51%)
# of sensitive species	48 (36%)

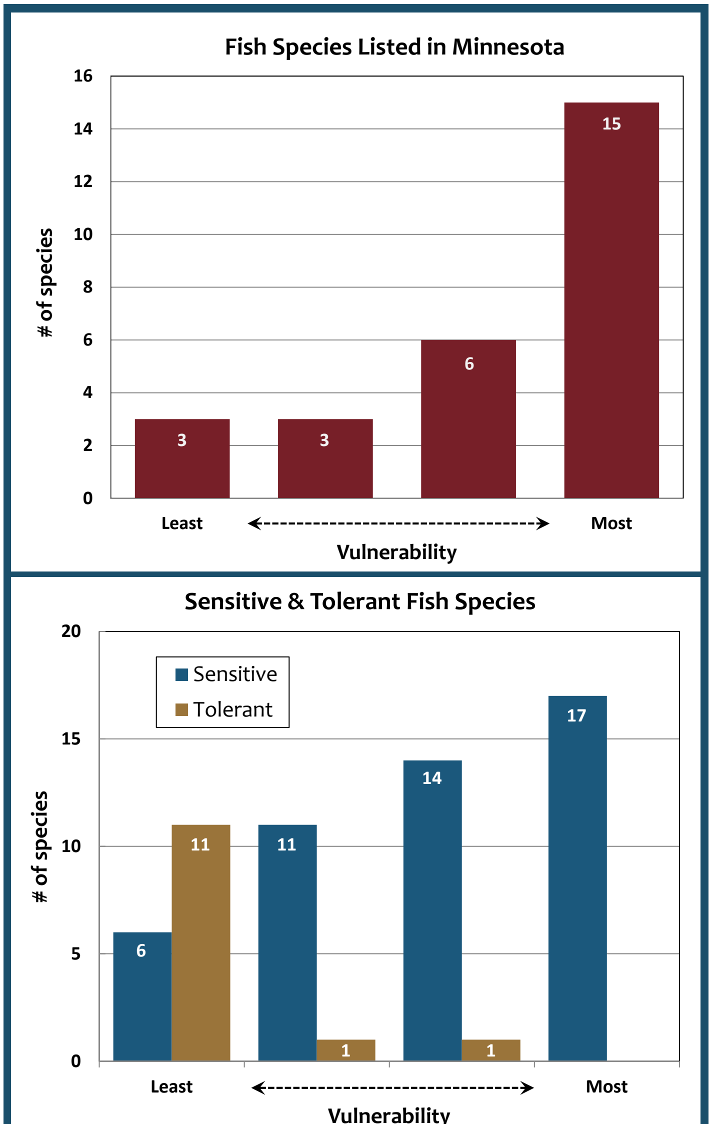


Figure 2 (top) Total number of listed species in Minnesota in percent absence quartiles. (bottom) Number of sensitive and tolerant native fish species (including naturalized common carp) in percent absence quartiles.

consistent with those of Santucci et al. (2005) in comparisons of free-flowing and fragmented reaches of the Fox River, Illinois.

While the absence of a species upstream of a dam does not prove that it was due to the barrier, historical records prior to the dam construction and later records following dam removal do substantiate a barrier effect. The likelihood of a species return to an upstream reach following a barrier removal can also be inferred by the presence of suitable habitat and comparisons to similar-sized connected streams and watersheds. Thirteen of the 32 dams were subsequently removed. Eleven dam removals have enough post-removal sampling effort to evaluate biodiversity effects, enabling greater certainty in defining barrier effects (Table 4). A summary of the species that returned following removal is shown in Table 4.

The general lack of spring surveys limits assessment of river reaches used for spawning but not for other life stages. Many species are known to ascend smaller rivers and streams in the spring followed by post-spawning downstream migrations back into larger river reaches. While juveniles of some species will remain near spawning areas as they mature, others will drift downstream as fry. Only 1 of 54 upstream surveys following removal of Flandrau Dam was done in May, with one in June, and none in April (a peak spawning month for many species). Most surveys were done in July, August or September. Some large-bodied species like flathead catfish and lake sturgeon that may only be present for a short but critical period in smaller river reaches are likely to be missed by summer surveys.

As expected, species known to migrate long distances and large-bodied fishes were among the most likely to be absent or extirpated upstream of dams. However, the list of species sensitive to fragmentation also included a number of small-bodied species as well as a disproportionate number of species listed as endangered, threatened or special concern in Federal, Minnesota and adjacent state listings.

The least likely species to be absent upstream of barriers were tolerant habitat generalists, stocked game and bait species, headwater fishes, and species that complete all life history stages in lakes. The absence of common carp upstream of barriers was relatively rare (25%) as it was for black bullhead (6%). Interestingly, these are two species typically targeted by fish barriers in Minnesota. Common carp were most likely to be absent upstream of complete barriers on cold-water streams, watersheds lacking lakes, or in watersheds that were relatively pristine.

Barrier Effects on Specific Fish Species

The sturgeons and paddlefishes of Order Acipenseriformes are the most vulnerable group in terms of extinction (85% of this group are critically endangered) because they are long distance migrants and their habitat needs are especially vulnerable to fragmentation (IUCN, 2004).

Lake sturgeon *Acipenser fulvescens* (*Special Concern in MN, WI, ON; Endangered in IA*) were absent above 80% of assessed dams (12 of 15). The exceptions were the St. Croix River upstream of St. Croix Falls Dam, and two of its tributaries, the Kettle River upstream of Sandstone Dam and the Snake River upstream of the Cross Lake Dam, which maintained the presence of lake sturgeon, but the species appears to be much less abundant than it was historically. The Cross Lake Dam may be passable for sturgeon during moderate floods helping to maintain a metapopulation, and the St. Croix, Kettle, and Snake rivers all have high quality spawning habitat connected to lakes and deep pools that would provide adult refugia and habitat from drought and winter conditions.

Lake sturgeon were extirpated from the entire Red River Basin and from the Minnesota River watershed upstream of Granite Falls where they were historically abundant to the headwaters of both watersheds. Dams in these basins inundated or blocked access to rapids where this species spawns like Rapidan (Blue Earth River), Minnesota Falls and Granite Falls (Minnesota River), Red Lake Falls (Red Lake River), and Fergus Falls (Otter Tail/ Red River).


A lake sturgeon caught below Minnesota Falls Dam before it was removed in 2013. *Credit Ken Peterson.*

Lake sturgeon will migrate hundreds of miles to spawn. A juvenile lake sturgeon tagged in Lake Pepin

Attachment

Barrier	Native fish species absent in upstream watershed while dam was present then found upstream of dam site after removal or modification or when dam was breached	# of species returned
Breckenridge Dam Otter Tail River Built in 1935 Replaced with rock ramp in 2007	silver lamprey[L], longnose gar[L], goldeye[L,I], mooneye[L,I], stonecat[I], white bass, sauger, lake sturgeon[MN,L]*	8 species (89% of 9 absent species)
State Park Dam Buffalo River Built pre 1893 & 1937 Removed in 2002	silver lamprey[L], goldeye[L,I], spotfin shiner, carmine shiner[L,I], sand shiner, northern redbelly dace[L], blacknose dace, quillback[L], silver redhorse, channel catfish, green sunfish, smallmouth bass[I], sauger, freshwater drum	14 species (67% of 21 absent species)
Heiberg Dam Wild Rice River Built in 1875 Removed in 2006	goldeye[L,I], brassy minnow, emerald shiner, carmine shiner[L,I], finescale dace[L], quillback[L], silver redhorse, channel catfish, tadpole madtom, smallmouth bass[I], sauger, freshwater drum, lake sturgeon[MN,L]*	13 species (81% of 16 absent species)
Sandstone Dam, Kettle River Built in 1905 Removed in 1995	southern brook lamprey[MN,I], blackchin shiner[I], blacknose shiner[L,I], mimic shiner[I], northern redbelly dace[L], bluntnose minnow, tullibee, banded killifish[L], gilt darter[MN,L,I], blackside darter[L], slimy sculpin[I], emerald shiner	12 species (55% of 22 absent species)
Welch Dam Cannon River Built in 1900 Removed in 1994	paddlefish[MN,L,I], mooneye[L,I], gizzard shad, speckled chub[L,I], silver chub[L], mimic shiner[I], river carpsucker, highfin carpsucker[I], river redhorse[L,I], flathead catfish[L], Muskellunge[I], brook trout[I], sauger, lake sturgeon[MN,L]	14 species (74% of 19 absent species)
Minnesota Falls Dam Minnesota River Built in 1871 & 1904 Removed winter 2013	shovelnose sturgeon[L], lake sturgeon[MN,L], flathead catfish[L], paddlefish[MN,L,I], mooneye[L,I], American eel[MN,L], gizzard shad, highfin carpsucker[I], blue sucker[MN,L,I], black buffalo[MN,L,I], sauger, silver lamprey[L] Notes: Removal was very recent so sampling effort has been limited and focused on the large species. American eel made it around dam during 2007 flood.	12 species (31% of 39 absent species) preliminary
Lake Florence Dam North Branch Root River Built in 1857 Removed in 1993	slenderhead darter[L,I], banded darter[I], smallmouth bass[I], bluegill, greater redhorse[L,I], golden redhorse[L], black redhorse[MN,L,I], smallmouth buffalo, northern hogsucker[L,I], longnose dace[I], sand shiner, gravel chub[MN,L,I], spotfin shiner, largescale stoneroller, chestnut lamprey[L]	15 species (23% of 65 absent species
Flandrau Dam, Cottonwood River Built in 1937. Dam was damaged by floods in 1947, was rebuilt in 1960, damaged again in 1965 and 1969, finally was fully removed in 1995	shovelnose sturgeon[L], mooneye[L,I], gizzard shad, golden shiner, river shiner[L], mimic shiner[I], river carpsucker, highfin carpsucker[I], black buffalo[MN,L,I], yellow bullhead[L], brown bullhead, channel catfish, white bass, Iowa darter[I], logperch[L], sauger, carmine shiner[L,I], freshwater drum, Mississippi silvery minnow[MN,I], speckled chub[L,I], silver chub[L] Note: Returned either while dam was passable or after it was removed.	21 species (88% of 24 absent species)
Dawson Dam Lac qui Parle River Built in 1913 Replaced with rock ramp in 2009	bigmouth buffalo[L], greater redhorse[L,I], channel catfish, bluegill, walleye	5 species (63% of 8 absent species)
Appleton Dam Pomme de Terre River Built in 1872 Removed in 1999	emerald shiner, carmine shiner[L,I], quillback[L], silver redhorse, greater redhorse[L,I], channel catfish, white bass, banded darter[I], freshwater drum	9 species (53% of 17 absent species)
Carp Barrier Dam, Drywood Creek, a tributary of the Pomme de Terre River Built in 1930s, failed, built taller in 1971. Failed in 2001	spotfin shiner, spottail shiner[I], common shiner, golden shiner, quillback[L], white sucker, shorthead redhorse, channel catfish, stonecat[I], Iowa darter[I], Johnny darter, banded darter, freshwater drum	13 species (72% of 18 absent species)
		Average = 66%

Table 4. Native fish species that returned to the watershed upstream of dam barriers after the dams were removed or modified or while the dam was passable. MN = listed in Minnesota, L = listed in neighboring state or province, I = intolerant (sensitive), * lake sturgeon were re-introduced since extirpation in the Red River Basin. The average does not include Minnesota Falls Dam since the removal was recent and post-removal data is limited.

was later caught below the Minnesota Falls dam in 2012, which is a distance of 300 miles. Lake sturgeon have been reintroduced to the Red River of the North since 1998. This has occurred concurrently with dam removal and fish passage projects to reconnect spawning rapids to the mainstem Red River and large lakes. Fish survey data confirm that this combined effort has been successful as sturgeon are becoming abundant in several of the large lakes.

A shovelnose sturgeon. *Credit DNR Fisheries.*

(top) A paddlefish caught in the Minnesota River near Granite Falls in 2005. *Credit DNR Fisheries.* (bottom) Paddlefish caught in 1957 just below Minnesota Falls Dam. *Credit Ken Peterson.*

Shovelnose sturgeon *Scaphirhynchus platorynchus* (*Federally Threatened*) were absent upstream of all assessed barriers (7). Shovelnose were absent upstream of Minnesota Falls Dam but returned to the rapids shortly after its removal. They were also absent upstream of Flandrau Dam but were caught about 25 miles upstream of the dam after its removal. Like other sturgeon species, shovelnose spawn in rapids and riffles over large substrates.

Paddlefish *Polyodon spathula* (*Threatened in MN and WI, Special Concern in ND, Extirpated in ON*) The paddlefish is a large river planktivore that spawns in riffles and rapids. Paddlefish were absent above all barriers assessed (4) but returned to the Minnesota River above Minnesota Falls Dam shortly after its removal in 2013 and to the Cannon River above Welch Dam following its removal in 1995. Fragmentation has been widely acknowledged as a primary cause of declines in this species (Unkenholz 1986). Paddlefish have been studied with particular attention as a planktivorous species which could be affected by bigheaded carp. The largest documented paddlefish, a 198 pound individual, was speared in Lake Okoboji, Iowa in 1916 where they were historically abundant. The species was extirpated from the lake, likely due to barrier dams on the Little Sioux River. Ironically an electric barrier recently installed on the outlet creek of Lake Okoboji, Iowa to prevent introduced carp from migrating into the lake also precludes reestablishment of paddlefish in the lake.

Restoration of the previously inundated Minnesota Falls should provide potential spawning habitat for paddlefish. Several paddlefish have been caught immediately downstream of the Minnesota Falls Dam over the years. Paddlefish have declined over their range due to dam construction that has blocked migrations and inundated spawning habitat.

Sauger *Sander canadensis* were absent upstream of all dams assessed (20). The closely related walleye may be nearly as sensitive to fragmentation, but widespread stocking masks possible barrier effects. Both species spawn in riffles and rapids in rivers or less commonly in clean wave-swept gravel or rubble shoals in lakes. Sauger returned to a number of river reaches following dam removal including: the Otter Tail after removal of Breckenridge Dam, the Cottonwood River after removal of Flandrau Dam, the Canon River after removal of Welch Dam, the Wild Rice River after

A sauger upstream of dam site after removal of Heiberg dam on the Wild Rice River. *Credit DNR Fisheries.*

An American eel. *Credit DNR Fisheries.*

removal of Heiberg Dam and the Minnesota River after removal of Minnesota Falls Dam. Walleyes similarly increased in abundance in these river reaches and successfully spawned in upstream reaches following removal of these dams.

American eel *Anguilla rostrata* (*Special Concern in MN, WI, SD, and ON*) were absent above 86% of assessed dams (6 of 7). This species is MN's only ocean-dependent species. These fish spawn in the Sargasso Sea then the catadromous (migrate from freshwater to the sea to spawn) females migrate back up the Mississippi River watershed. They have the unusual ability to occasionally pass some barriers by "swimming" out of water (usually in wet grass) and there is a single record in 1957 as far upstream as St. Anthony Falls prior to construction of the Lock. Another eel, caught by Area Fisheries staff made it past Minnesota Falls Dam in 2007, a year that lacked a flood large enough to inundate the dam. With the exception of these two individuals, they were absent above barriers for all of the assessed watersheds for which records exist. Since they spawn in the ocean, it follows that any complete barrier would extirpate them from the watershed. This has proven to be the case since American eel have declined over most of their range due to dam construction

A skipjack herring. *Credit Konrad Schmidt.*

Skipjack herring *Alosa chrysochloris* (*Endangered in MN and WI, Special Concern in SD*) was absent above all barriers assessed (3). This species was historically found in Bigstone Lake at the headwaters of the Minnesota River. They were largely extirpated from all Minnesota waters following construction of Lock and Dam 19 in 1913. This dam inundated Keokuk Rapids, which would have been an important spawning area for sturgeon, paddlefish and other rapid dependent species. It is also the tallest, 36 feet, lock & dam on the Mississippi. The loss of skipjack herring resulted in the near extirpation of elephant-ear *Elliption crassidens* and ebonyshell *Fusconaia ebena* mussels, for which skipjack herring are the sole host. Historically, ebonyshell mussels were the dominant mussel species in the Upper Mississippi and Lower Minnesota rivers of Minnesota. A few skipjack herring were caught in Lake Pepin in 1986 for the first time since 1928 and subsequently in 1993, 2001, and 2008. These fish would have had to pass through the lock chamber at Dam 19. The endangered skipjack herring and the dependent ebonyshell and elephant-ear mussels illustrate the importance of fish passage on the Mississippi River and the cascading fragmentation effects on biodiversity. Skipjack herring are also a piscivore that feed within the water column and may be an effective predator on bigheaded carp eggs, larvae, and juveniles.

Blue sucker *Cycleptus elongatus* (*Special Concern in MN, ND and SD, Threatened in WI*) were absent upstream of 100% of the barriers assessed (6). They maintained a population upstream of St. Croix Falls Dam on the St. Croix until the late 1970s. The large, relatively pristine watershed upstream of St. Croix Falls provides a suite of habitat, particularly rapids that this species prefers. Blue suckers maintained a metapopulation for a period of decades after

A blue sucker collected while electrofishing Minnesota Falls following removal of the Minnesota Falls Dam. *Credit DNR Stream Habitat Program.*

A longnose gar (left) and shortnose gar (right) caught upstream of Minnesota Falls Dam. Gar were absent from the reach above the dam prior to its removal. *Credit DNR Fisheries.*

the dam was built, but the species was ultimately lost from the reach by the late 1970s. Blue sucker were absent upstream of Minnesota Falls Dam, but an individual was caught following the 2011 flood that largely inundated the dam. The species was caught in numbers following removal of the dam in 2013. Blue sucker are a fast water species found predominantly in rapids.

Shortnose gar *Lepisosteus platostomus* and **longnose gar** *Lepisosteus osseus* (*Special Concern in SD*) both were absent upstream of 73% of barriers assessed (8 of 11). Gar may be an important predator on juvenile bigheaded carp (Duane Chapman, USGS, personal communications). The ability of juvenile bighead and silver carp to grow

A mooneye caught above Minnesota Falls dams site after dam removal. *Credit DNR Stream Habitat Program.*

vascularized lip extensions enable them to use atmospheric oxygen and inhabit warm, backwaters with low dissolved oxygen where most predators can't survive. Gar are also able to gulp oxygen due to lung-like vascularized swim bladders enabling them to live and hunt in these warm anoxic backwaters.

Mooneye *Hiodon tergisus* (*Concern in SD*) were absent upstream of all barriers assessed (15) while the closely related **goldeye** *Hiodon alosoides* (Endangered in WI) were absent above 92% of barriers (12 of 13). Both species returned to a number of river reaches following dam removal (Table 4). Mooneye and goldeye feed in the water column and at the surface on a variety of insects and small fishes. Their pelagic feeding behavior may equip them to be important predators on bigheaded carp fry and small juveniles.

Flathead catfish *Pylodictis olivaris* (*Concern in ND*) were absent upstream of all barriers assessed (11). They did return to the Canon River following removal of the Welch Dam and to the Mississippi River above St. Anthony Falls following construction the lock in 1963. Flathead catfish need deep pools, usually in larger rivers, for wintering but often migrate upstream to spawn in smaller streams. Flathead adults and fingerlings (indicating reproduction) have been found in the free-flowing Yellow Medicine River, which has an average flow of only 154 cfs and average August flows of only 66 cfs. Flathead catfish are the largest predatory fish in Minnesota and are capable of eating carp up to 30% of their body weight. Davis (1985) reported that stocked flatheads caused a 90% reduction in common carp abundance in Richardson Lake. It is known that these fish can grow very large, as a 157 pound flathead was illegally taken from the Minnesota River near Redwood Falls in 1930. Flatheads are capable of preying on adult carp and may be an important biological control.

A flathead catfish caught on the Minnesota River during Fisheries surveys. *Credit DNR Fisheries.*

A channel catfish caught on the Red River of the North. *Credit DNR Stream Habitat Program.*

Channel catfish *Ictalurus punctatus* absent upstream of 61% of assessed barriers (19 of 31), and **freshwater drum** *Aplodinotus grunniens* absent upstream of 64% of barriers (18 of 28), are two species that are especially important hosts for freshwater mussels. Freshwater drum are hosts for at least 11 species of native mussels, of which they are the sole hosts for 8 species (Figure 3). Channel catfish are hosts for at least 13 species of mussels and are the primary hosts for 6 species. Both fish species were extirpated from the Cottonwood watershed by Flandrau dam. Attempts to re-establish channel catfish by stocking failed.

A freshwater drum. *Credit DNR Stream Habitat Program.*

Following the removal of Flandrau Dam channel catfish and freshwater drum returned almost to the headwaters, 112 miles upstream of dam.

Small-bodied fish While tagging studies have shown that large-bodied fish are migratory, these results and fishway data indicate that many small fish species also migrate and are impacted by barriers.

Shiners & minnows

Shiners are a keystone forage species. Many shiner species are not tolerant of low dissolved oxygen, which may make them vulnerable to extirpation due to barriers. Their vulnerability to extirpation has obvious implications on the productivity of fisheries and for the bait industry. The following species were often absent upstream of barriers:

- **speckled chub** *Macrhybopsis aestivalis* (Threatened in WI) 100% of 11 barriers,
- **Mississippi silvery minnow** *Hybognathus nuchalis* (*Special Concern in MN*) 100% of 7,
- **gravel chub** *Erimystax x-punctatus* (*Threatened in MN, Endangered in WI, Extirpated from Canada*) 100% of 3,
- **silver chub** *Macrhybopsis storeriana* (Special Concern *in WI, SD, ND, and Canada*) 92%, 12 of 13,
- **slimy sculpin** *Cottus cognatus* 70%, 7 of 10
- **river shiner** *Notropis blennius* (Special Concern in SD) 70%, 7 of 10,
- **carmine shiner** *Notropis rubellus* (Threatened in *Canada, Concern in ND and SD*) 59%, 13 of 22, and
- **emerald shiner** *Notropis atherinoides* 52%, 12 of 23
- **spotfin shiner** *Notropis spiloptera* 44%, 12 of 27.
- **sand shiner** *Notropis stramineus* 40%, 12 of 30
- **spottail shiner** *Notropis hudsonius* 37%, 7 of 19.

Darters

Darter diversity is an important indicator of ecosystem health and a metric for the index of biological integrity.

The following species tended to be absent upstream of barriers:

- **western sand darter** *Ammocrypta clara* (*Threatened in IA, Special Concern in WI*) 100% of 7 barriers,
- **crystal darter** *Crystallaria asprella* (*Endangered in MN & WI, Extirpated from IA*) 100% of 6,
- **river darter** *Percina shumardi* (*Special Concern in ND*) 88%, 7 of 8,
- **mud darter** *Etheostoma asprigene* (*Special Concern in WI*) 75%, 3 of 4
- **gilt darter** *Percina evides* (*Threatened in WI, Special Concern in MN, Extirpated from IA*) 71%, 5 of 7,
- **banded darter** *Etheostoma zonale* 64%, 7 of 11.

Mussels Mussel surveys were more limited than those for fish but followed similar trends. Since most mussels require fish hosts, extirpation of the host will ultimately result in the extirpation of the mussel. However, due to the long life span of mussels, up to 200 years for one species (Haag and Rypel 2011), individuals may persist well after being functionally extirpated. Still, mussel diversity has decreased in many waters, particularly in the Minnesota River watershed where 23 of 41 species no longer exist. Unlike fish, historic mussel communities can be determined by the presence of dead shells. Like fish, poor water quality, sedimentation, and habitat alteration and changes in hydrology can adversely affect mussels.

The recolonization of 3 mussel species following removal of the Appleton Dam, on the Pomme de Terre River, is evidence that fragmentation was the cause of their extirpation. Pre-dam removal surveys found only dead shells of elktoe *Alasmidonta marginata*, deertoe *Truncilla truncate* and plain pocketbook *Lampsilis cardium* mussels upstream of the dam. Archeological surveys along the shores of Lake Christina, near the headwaters of the Pomme de Terre River, found plain pocketbook mussel shells indicating that this species was historically found in the headwaters of this watershed. Extirpation of these mussels upstream of the dam and their subsequent recolonization following the dam's removal may have different explanations based on the presence or extirpation of host fish species.

Freshwater drum, also extirpated upstream of the dam, are the sole host for deertoe mussels (Figure 3). The disappearance of this fish species would have led to the extirpation of this mussel species by the inability to reproduce. Return of the drum following removal of the dam is the likely explanation for the recolonization of deertoe mussels.

Rock bass and three sucker species (shorthead redhorse, white sucker, and northern hogsucker) have been identified as hosts (naturally infected; successful transference has not yet been determined) for elktoe mussels. Except for northern hog sucker, these species were present upstream of the dam. However, northern hogsucker and three additional sucker species (greater redhorse, silver redhorse, and quillback carpsucker) that were absent upstream of the dam recolonized following its removal. The return of these species may have been important in the recolonization of elktoe mussels. Functional mussel hosts need to be physiologically compatible, but habitat preferences and behavior also determine the success of mussel reproduction.

Plain pocketbook mussels also use species (walleye, black bass, and several sunfish species) that were present prior to the dam's removal. This suggests that the two latter species may have died out due to drought or other factors and lacked the ability to recolonize due to the dam. Like many rivers, the Pomme de Terre River has stopped flowing during droughts in several periods including the 1934, 1936, 1976, 1988, and 1989. Host fish cannot facilitate reproduction unless they can be infected by glocidia released by viable adults. Removal of the dam would have enabled both existing host fishes and extirpated hosts to become infected in downstream mussel beds and facilitate mussel recolonization of reaches upstream of the dam.

Watershed Scale Biodiversity Effects

Fish diversity was assessed along the Cottonwood River and its tributaries for periods with and without the presence of Flandrau Dam (see Figure 4).

Biodiversity effects of the dam extended to the entire watershed. Cumulative species richness and species per survey are shown in Figure 5. The species richness of the free-flowing Cottonwood River compared to the fragmented river was significantly greater based on a randomization t-test ($t = 2.998$, $\rho = .0016$).

In the absence of the dam, species richness increased by an average of 35% in the watershed and this increase extended to upper reaches of the watershed. For instance, channel catfish and freshwater drum were sampled in Double Lake (drainage area of 2.2 mi^2 , 112 miles upstream of the dam); these two species were not collected in any

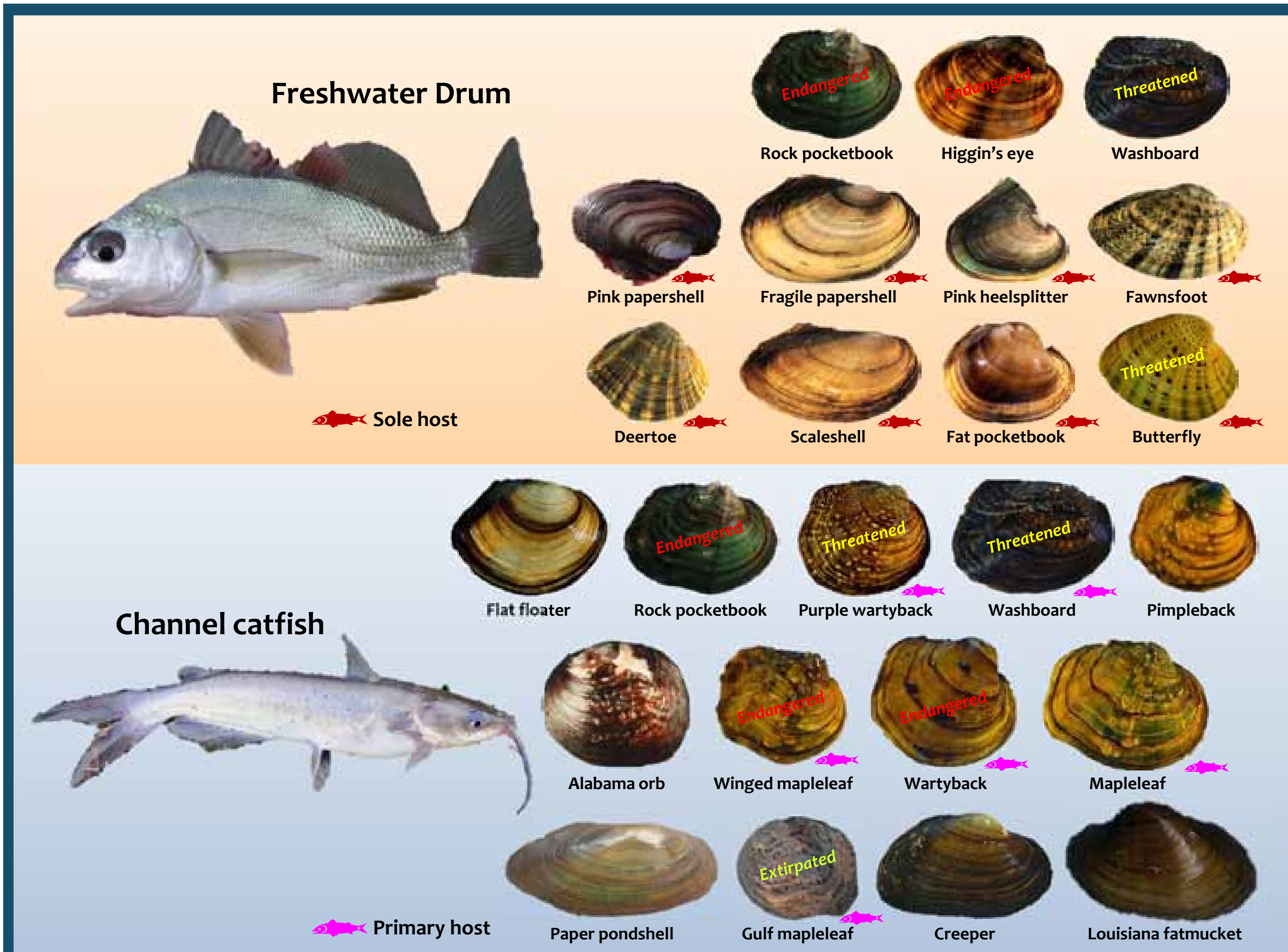


Figure 3. Freshwater drum and channel catfish and the mussels for which they are hosts.

samples upstream of Flandrau Dam prior to the dam's removal. The lake flows into Highwater Creek so these fish would have needed to ascend the creek, which is only about 10 feet wide at the lake's outlet. Removal of the dam also provided access to boulder rapids that are key spawning habitat for walleye, sauger, paddlefish, lake sturgeon, blue sucker, black buffalo and others.

Twenty-one of the twenty-four species that were absent upstream of Flandrau Dam were collected upstream of the dam site during the period when it was breached in 1948 or after it was removed in 1995 (Table 4).

Silver chub, Mississippi silvery minnow, and carmine shiner were present upstream of the dam in 1948 when it was breached, but have not yet been caught upstream of the dam site since removal. Land use changes like ditching, tiling, wetland drainage, use of nitrogen and phosphorous fertilizer, and pesticide use have caused significant habitat and water quality changes that may be unsuitable for these species. These minnows tend to migrate later in the spring and may still be blocked by low-head dams like Kuhar Dam near Lamberton, which is submerged during high spring flows, but would become a barrier as flows decrease. Rates of recolonization likely vary with species as well and these species are relatively rare. In addition to those already mentioned, flathead catfish, shortnose gar and longnose gar, speckled chub, and black buffalo, caught downstream of the dam, have not yet been collected upstream of the dam.

The presence or absence of species does not provide a full perspective of fragmentation effects since it does not show changes in abundance. A number of riverine species that were present in small proportions of the surveys while the river was dammed increased in prevalence (percent occurrence) when the main stem was free-flowing (Figure 6). For instance, the proportion of samples

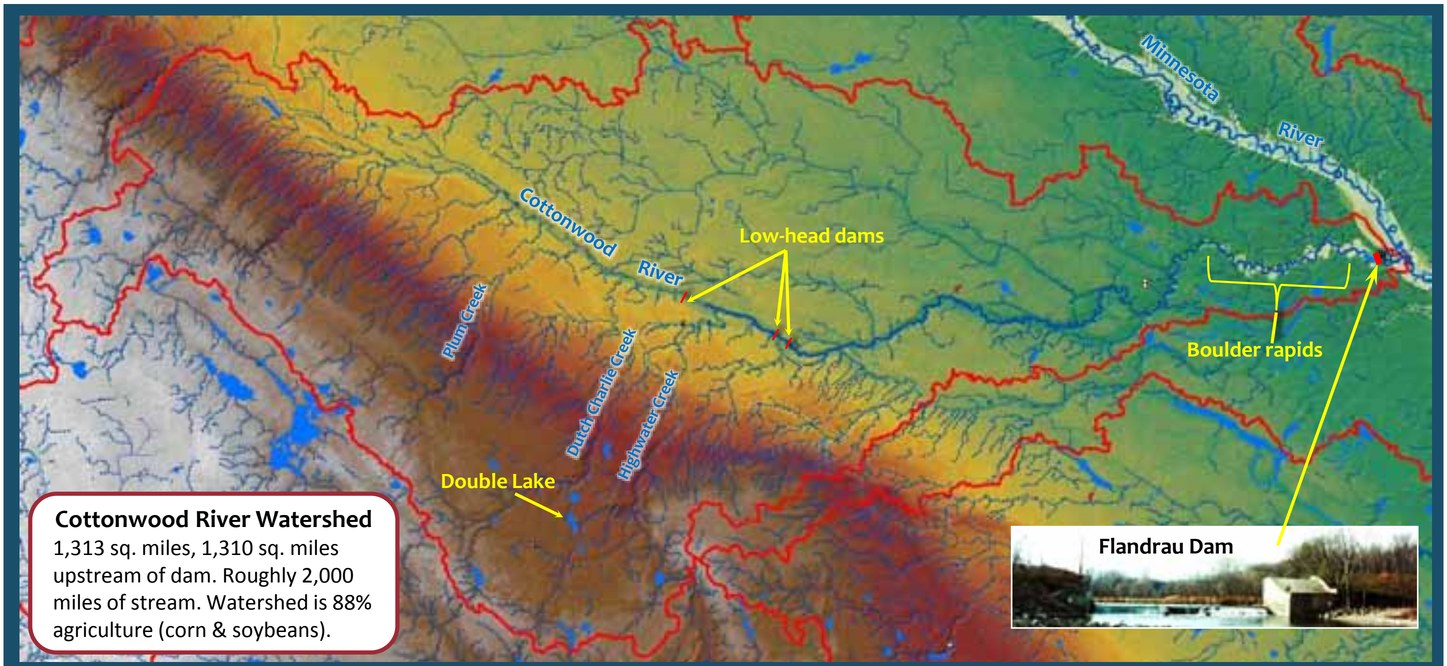
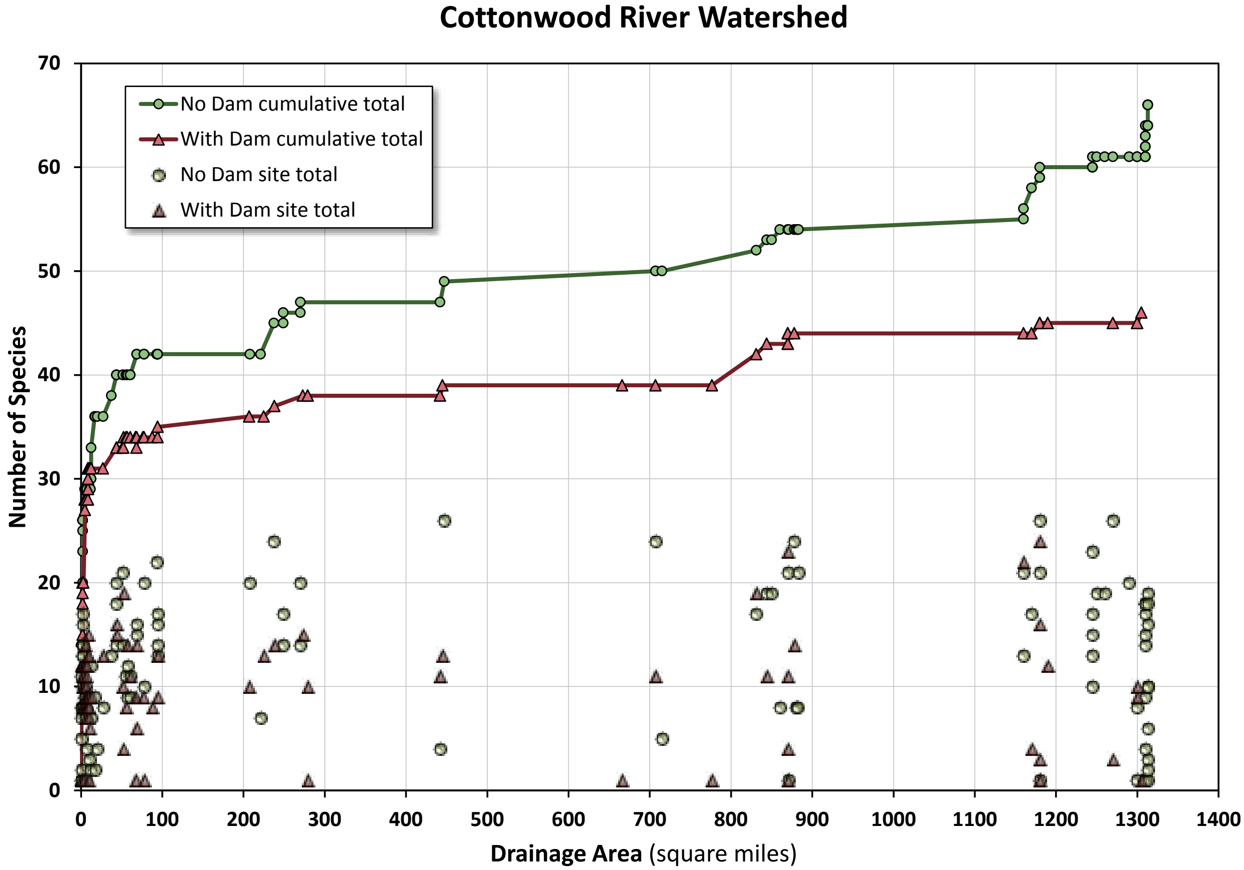


Figure 4. The Cottonwood River watershed.

in which river-oriented suckers were caught increased for all species. Percent occurrence of shorthead redhorse was 330% higher, silver redhorse 182% higher, golden redhorse 325% higher, northern hogsucker 236% higher, quillback 247% higher, and highfin carpsucker were 240% higher in the free-flowing compared to the dammed condition. Among facultative riverine game species, the proportion of samples in which smallmouth bass were caught was 88% higher in the free-flowing condition and walleye were 105% higher while sauger and channel catfish were absent in the dammed condition but were found in 8% and 24% of free-flowing samples. Abundant tolerant species like white sucker, fathead minnow, and black bullhead did not appear to be affected by fragmentation and tended to be present in virtually the same proportion of samples during the free-flowing and dammed condition.



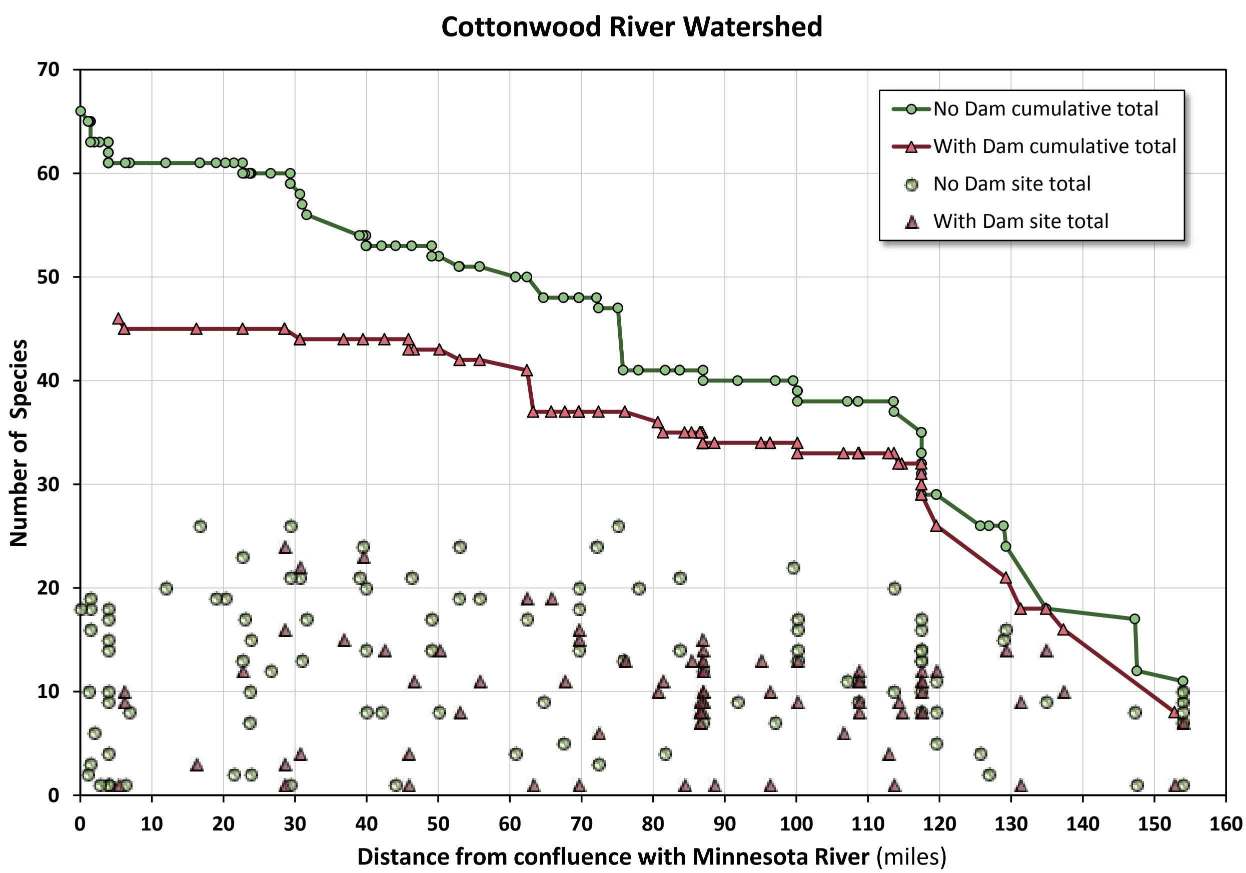


Figure 5. Number of species found in the Cottonwood River watershed. Points are the total number of species collected at a site. The line is the cumulative total. (top) Species richness is correlated with drainage area (bottom) Species richness correlated with distance from the mouth of the Cottonwood River.

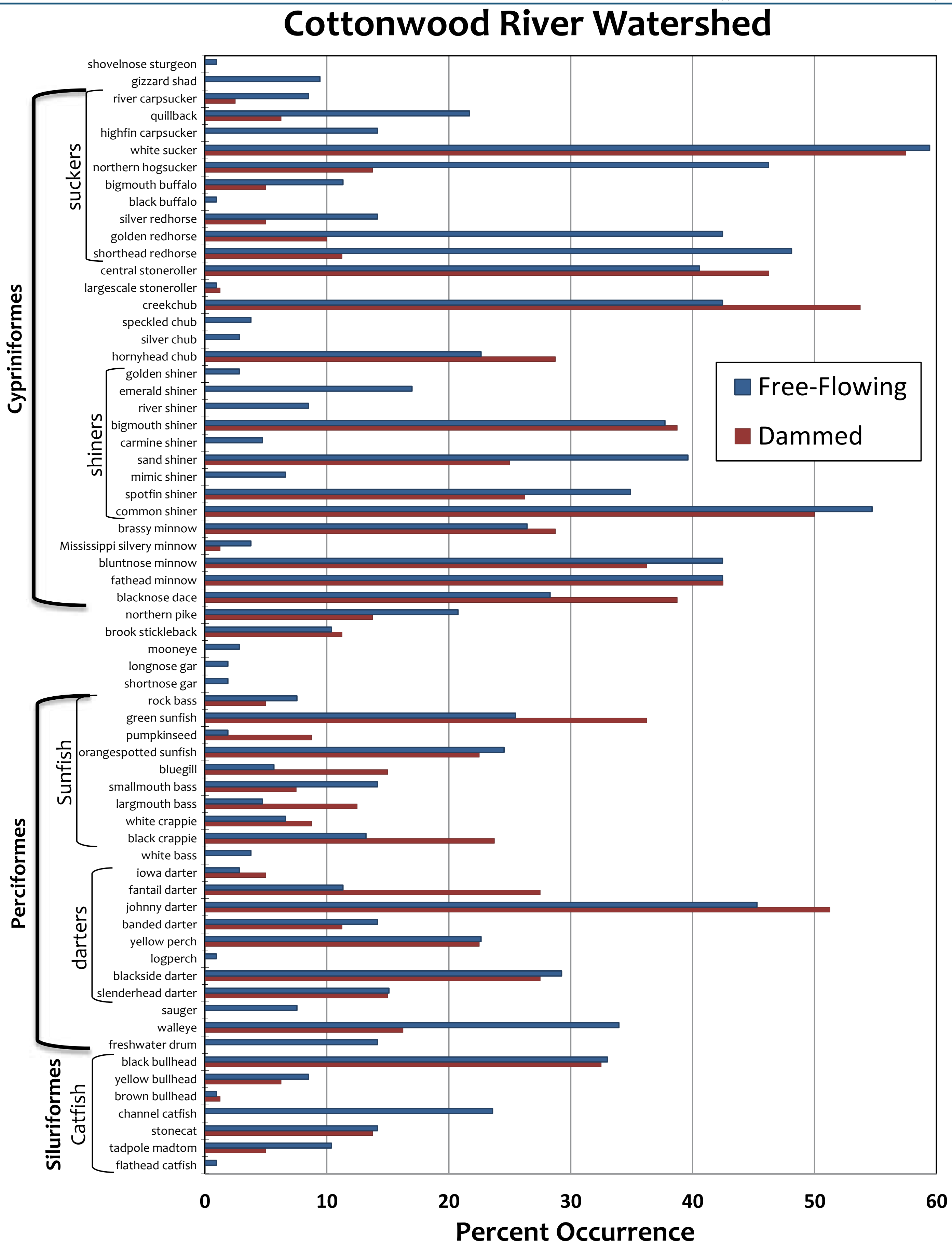


Figure 6. Percent occurrence of fish species from fish surveys in the Cottonwood River watershed separated into periods when Flandrau Dam was a barrier - dammed and when the dam was breached or removed - free flowing.

Summary and Conclusions

There are few impairments that have been shown to have as dramatic an influence on aquatic biodiversity as does the construction of barriers. To summarize:

1) Complete and near complete barriers reduced upstream species richness by an average of 41% and 37 % respectively.

2) Moderate barriers (may be passable during 2-year or larger floods) also reduced species richness by 20%. This is evidence that even partial barriers have an upstream impact.

3) Loss of species richness due to barriers extended watershed-wide.

4) Imperiled and sensitive species were the most vulnerable to extirpation by barrier dams.

5) Tolerant species, including common carp, were among the species least affected by barriers.

6) An average of 66% of species absent above barrier dams returned after the barrier was removed.

7) Based on this analysis and other studies the ability to migrate (or connectivity of migration pathways) is equally important to fish as it is to neotropical birds.

Ecological Implications of Dams The implications of barrier effects extend to fundamental elements of ecological health. Dams can have additional effects by interrupting sediment transport causing reservoir sedimentation and downstream incision, altering nutrient dynamics and causing cyanobacteria blooms, propagating non-native species, inundating important river habitat, altering flow regimes, altering temperature regimes, propagating fish diseases and parasites, and causing massive erosion when they fail. However, the effects on native species shown by this analysis are primarily due to the blockage of fish migrations since most of the reservoirs were relatively small in comparison the watershed-wide effects. Blocking seasonal fish migrations directly affects nutrient processing and water quality since fish carry these nutrients in their bodies and eggs. While this paper assessed barrier dams, any type of barrier that is effective in blocking fish migrations should be expected to cause significant declines in the diversity of fish and mussels.

This analysis has shown that barriers have direct negative effects on recreation as a number of game fish species were vulnerable to barrier related extirpation. Flathead catfish, sauger, white bass, yellow bass, and paddlefish were absent upstream of all barriers evaluated while lake sturgeon, channel catfish, and white bass were absent upstream of most barriers in watersheds where they were present. Smallmouth bass, in spite of being artificially maintained by stocking in some watersheds, were absent upstream of a number of barriers. The return of these species following dam removal supports fragmentation as the cause of their extirpation. Walleye may also be vulnerable to barrier extirpation, based on spawning habitat needs and the sensitivity of sauger (a close relative to walleye) to fragmentation, but walleyes are artificially maintained by extensive stocking.

Predatory game species are also affected by barrier effects on forage species. Several shiner and minnow species were frequently extirpated by barrier dams (again validated by their return following dam removal). Mimic shiner, emerald shiner, carmine shiner, weed shiner, silver chub, Ozark minnow, pugnose minnow, and river shiner were all absent upstream of half or more of the barrier dams in watersheds they were present.

The extirpation of native mussels that follows the loss of host fish species above dams eliminates the water filtration role of these mussels. Water filtration by mussels of the Upper Mississippi River has been estimated at 53.1 million cubic meters per day or 76 times the capacity of the Minneapolis - St. Paul metropolitan wastewater treatment plant, one of the largest in the USA (Newton et al. 2011). Mussels also stabilize stream beds (Zimmerman and de Szalay 2007) and increase the density and biodiversity of other benthic invertebrates (Spooner and Vaughn 2006; Gutierrez et al. 2003). Mussels are declining globally and this catastrophic loss in biomass may significantly alter river ecosystem functions (Spooner and Vaughn 2006). The recolonization of three extirpated mussel species following removal of the Appleton dam suggests that this trend is reversible for the species that have not yet gone extinct.

The Minnesota River The Minnesota River, one of the watersheds for which invasive species barriers are being considered, has been well documented for its water quality and sediment impairments. Nevertheless, the river between Granite Falls and its confluence with the Mississippi River is the longest reach of free-flowing, undammed river in Minnesota, a distance of 240 miles. Where free-flowing, the river mainstem and tributaries have a remarkable diversity of fish, with records of 98 native species. While the watershed has lost much of its mussel diversity, dam removal has proven to be an effective strategy in reestablishing extirpated species of fish and mussels.

While landuse impacts on water quality, hydrology, and channel erosion continue to degrade habitat in the Minnesota River and other watersheds, it is notable that where dams have been removed, the loss of biodiversity has actually been reversed and has resulted in substantial increases in species richness. This demonstrates the necessity of migration for reproduction, accessing changing habitat needs with seasons and life stage, and recolonization following drought, anoxia and water quality related mortality. Connectivity may be particularly important in watersheds subject to low winter flows, anoxia, and high summer water temperatures associated with drought since the fish and mussel assemblages of these streams depend on frequent recolonization.

Vulnerability to Fragmentation Tolerant native and introduced species have been successful in fragmented, degraded, and altered systems. These species can survive drought and often concurrent warm water temperatures and low dissolved oxygen, in addition to other water quality impairments. Tolerant species are often generalized and adapted to homogenized, silt laden microhabitat. Common carp were abundant upstream of most barriers, especially in eutrophic watersheds. This included barriers specifically designed to target carp. The extirpation of native species by barriers may actually increase the success of invasive species by eliminating competition and predation influences associated with a diverse, free-flowing river.

The high vulnerability of sensitive and imperiled species and relatively low vulnerability of tolerant species to extirpation by barriers has significant implications for ecosystem health and biological assessments. The Index of Biological Integrity, IBI (Karr et al. 1986), widely used as a measure of biological health and water quality, uses metrics that include the number of sensitive species, darter species, and sucker species as positive metrics. This study supports the usefulness of the IBI as a measure of biological health but suggests that fragmentation may significantly reduce scores. A fragmented system is more likely to be dominated by tolerant species that can survive periods of poor water quality, while a free-flowing system allows periodic recolonization by sensitive species.

Since 1) protection of native species is a primary objective of invasive species management and 2) this and other studies suggest that barriers are the single most definitive cause of declines in native biodiversity, barriers on naturally connected rivers and streams should not be considered a viable invasive species control strategy. Rather, reconnecting rivers by removing barriers has been shown to increase the diversity and resilience of native species while decreasing the prevalence of invasive species. Restoration of free-flowing, resilient ecosystems is likely to be the most effective means of increasing native biodiversity and preventing dominance by non-native species.

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Most Vulnerable Species : 75% to 100% Absence

Common Name *Scientific Name*	Adult Habitat *Spawning Habitat*	Adult Feeding Habits	% Absent Upstream of Barriers # Absent / Sample size	Conservation Status *Tolerance* Management (if any)
shovelnose sturgeon *Scaphirhynchus platorynchus*	pools in rivers **rapids in rivers & streams**	benthic invertivore	**100%** 7/7	**Federally Threatened** Intermediate
paddlefish *Polyodon spathula*	pools in large rivers **riffles & rapids in rivers**	planktivore/benthic invertivore	**100%** 4/4	**T** (MN, WI), **SCP2** (ND), **Ext** (ON) Intolerant
mooneye *Hiodon tergisus*	pools in rivers, connected lakes **pelagic, rivers**	surface & water column invertivore/piscivore	**100%** 15/15	SU (SD) Intolerant
skipjack herring *Alosa chrysochloris*	pools in rivers, connected lakes **pelagic, rivers**	surface & water column invertivore/piscivore	**100%** 3/3	**E** (MN, WI), **S3** (SD) Intermediate
gizzard shad *Dorosoma cepedianum*	pools in rivers, connected lakes **pelagic, rivers**	surface & water column planktivore/invertivore	**100%** 12/12	Intermediate
Mississippi silvery minnow *Hybognathus nuchalis*	pools & backwater in rivers & streams **glides, riffles, hornyhead chub nests**	benthic invertivore	**100%** 7/7	**SC** (MN) Intolerant
gravel chub *Erimystax s-punctatus*	riffles in coolwater rivers **glides, riffles**	herbivore, filamentous algae, diatoms	**100%** 3/3	**T** (MN), **E** (WI), **Ext** (ON) Intolerant
speckled chub (shoal chub) *Macrhybopsis aestivalis*	sandy riffles in rivers **semi-pelagic**	benthic invertivore	**100%** 11/11	**T** (WI) Intolerant
Topeka shiner *Notropis topeka*	streams **sunfish nests**	generalized invertivore	**100%** 2/2	**Federally Endangered**, **T** (IA), **SC** (MN), **S3** (SD) not rated
channel shiner *Notropis wickliffi*	pools in rivers **glides, riffles**	generalized invertivore	**100%** 3/3	Intermediate
ghost shiner *Notropis buchanani*	eddies & backwaters in rivers **glides, riffles**	generalized invertivore	**100%** 3/3	Intolerant
pugnose minnow *Opsopoeodus emiliae*	clear vegetated streams **under objects**	omnivore	**100%** 4/4	**SC** (IA, WI, ON) Intolerant
longnose sucker *Catastomus catastomus*	streams, Great Lakes, brackish water **riffles and shoals**	benthic invertivore	**100%** 1/1	**T** (SD) Intermediate
blue sucker *Cycleptus elongatus*	rapids in rivers **glides, riffles & rapids**	benthic invertivore	**100%** 6/6	**T** (WI), **SC** (MN), **SCP1** (ND), **S3** (SD) Intolerant
black buffalo *Ictiobus niger*	runs & pools in coolwater rivers **backwaters & floodplains**	benthic invertivore	**100%** 3/3	**T** (MN, WI), **SC** (ON), **PSC** (Canada), **SU** (SD) Intolerant
spotted sucker *Minytrema melanops*	clearwater rivers **glides, riffles & rapids**	benthic invertivore	**100%** 5/5	**SC** (Canada, ON) Intolerant
slender madtom *Noturus exilis*	riffles in streams **under rocks**	generalized invertivore	**100%** 1/1	**E** (MN, WI) Intolerant
flathead catfish *Pylodictis olivaris*	deep pools in rivers **nests in cavities**	piscivore, top predator	**100%** 11/11	**SCP3** (ND) Intermediate

Table 4. Fish species listed by percent absence upstream of dam barriers analyzed and listed in Table 1. Table is sorted by percent absence. Fish habitat and feeding data from Aadland & Kuitunen 2005 and Becker 1983. Conservation status: **E** = Endangered, **T** = Threatened, **SC** = Special Concern, **Ext** = Extirpated from Minnesota DNR (MN), Iowa DNR (IA), Wisconsin Natural Heritage Working List (WI), North Dakota Game & Fish Department (ND) Species of Conservation Priority, **SCP**, Levels 1 - 3), South Dakota Game Fish & Parks (SD, State Rank **S1 - S5**), U.S. Fish & Wildlife Service, and Government of Canada (Canada, Ontario=ON, **PSC**=Proposed Special Concern). Species tolerance ratings from US EPA.

Most Vulnerable Species : 75% to 100% Absence

Common Name ***Scientific Name***	**Adult Habitat** ***Spawning Habitat***	**Adult Feeding Habits**	**% Absent Upstream of Barriers** # Absent / Sample size	**Conservation Status** *Tolerance* **Management** (if any)
pirate perch *Aphredoderus sayanus*	sluggish streams, backwaters, wetlands **nest in vegetation**	generalized invertivore	**100%** 1/1	**SC** (MN, IA, WI) **Intermediate**
plains topminnow *Fundulus sciadicus*	streams **vegetation**	generalized invertivore	**100%** 1/1	**T** (MN), **S3** (SD) **not rated**
starhead topminnow *Fundulus dispar*	vegetated streams & backwaters **vegetation**	generalized invertivore	**100%** 1/1	**E** (WI) **Intolerant**
threespine stickleback *Gasterosteus aculeatus*	streams, lakes, and brackish bays **nest in shallow water**	generalized invertivore	**100%** 1/1	**E** (Canada) **Intermediate**
yellow bass *Morone mississippiensis*	pools in rivers, connected lakes **glides & riffles in streams**	planktivore, piscivore	**100%** 5/5	**SC** (MN) **Intermediate**
white perch *Morone americana*	rivers, lakes, and brackish bays **broadcast in rivers**	piscivore	**100%** 1/1	**Intermediate**
western sand darter *Ammocrypta clara*	sandy riffles in rivers **glides & riffles, sand**	generalized invertivore	**100%** 7/7	**T** (IA), **SC** (WI) **Intolerant**
crystal darter *Crystallaria asprella*	sandy riffles in rivers & streams **glides & riffles**	generalized invertivore	**100%** 6/6	**E** (MN, WI), **Ext** (IA) **Intolerant**
sauger *Sander canadensis*	pools in rivers, lakes **glides, riffles & shoals**	piscivore	**100%** 20/20	**Intermediate** Occasionally stocked game species
goldeye *Hiodon alosoides*	pools in rivers, connected lakes **pelagic, rivers**	surface & water column invertivore/piscivore	**92%** 12/13	**E** (WI) **Intolerant**
silver chub *Macrhybopsis storeriana*	pools in rivers **semi-pelagic**	benthic invertivore	**92%** 12/13	**SC** (WI, Canada) **S2** (SD), **SCP2** (ND) **Intermediate**
highfin carpsucker *Carpiodes velifer*	runs & pools in rivers & streams **backwaters**	omnivore	**91%** 10/11	**Intolerant**
bullhead minnow *Pimephales vigilax*	rivers & backwaters **underside of objects**	omnivore	**88%** 7/8	**Intermediate**
river darter *Percina shumardi*	riffles in rivers & streams **glides & riffles**	generalized invertivore	**88%** 7/8	**SCP3** (ND) **Intermediate**
American eel *Anguilla rostrata*	rivers (females) **Sargasso Sea**	piscivore	**86%** 6/7	**SC** (MN, WI, ON), **S3** (SD) **Intermediate**
silver lamprey *Ichthyomyzon unicuspis*	pools in rivers **glides, riffles**	parasite on fish	**82%** 14/17	**SCP3** (ND) **Intermediate**
lake sturgeon *Acipenser fulvescens*	pools in rivers, connected lakes **rapids in rivers & streams**	benthic invertivore	**80%** 12/15	**E** (IA), **SC** (MN, WI, ON) **Intermediate** Reintroduced in some waters
smallmouth buffalo *Ictiobus bubalus*	pools in rivers, lakes **backwaters & floodplains**	generalized invertivore	**80%** 8/10	**Intermediate**
black redhorse *Moxostoma duquesnei*	fast riffles & runs in streams **glides, riffles**	benthic invertivore	**80%** 4/5	**E** (WI), **T** (IA, Canada, ON), **SC** (MN) **Intolerant**
mud darter *Etheostoma asprigene*	rivers & backwaters **riffles on gravel or vegetation**	generalized invertivore	**75%** 3/4	**SC** (WI) **Intermediate**

Vulnerable Species : 50% to 74% Absence

Common Name ***Scientific Name***	**Adult Habitat** ***Spawning Habitat***	**Adult Feeding Habits**	**% Absent Upstream of Barriers** # Absent / Sample size	**Conservation Status** *Tolerance* **Management** (if any)
longnose gar *Lepisosteus osseus*	pools in rivers, connected lakes ***vegetated backwaters & bays***	piscivore	**73%** 8/11	**S3** (SD) **Intermediate**
shortnose gar *Lepisosteus platostomus*	pools in rivers, connected lakes ***vegetated backwaters***	piscivore	**73%** 8/11	**Intermediate**
brook silverside *Labidesthes sicculus*	ubiquitous in rivers, connected lakes ***nearshore over vegetation or gravel***	surface & water column invertivore, fish fry	**73%** 8/11	**Intermediate**
gilt darter *Percina evides*	fast riffles in rivers & streams **glides & riffles**	generalized invertivore	**71%** 5/7	**T** (WI), **SC** (MN), **Ext** (IA) **Intolerant**
white bass *Morone chrysops*	pools in rivers, connected lakes **glides & riffles in streams, shoals**	planktivore, piscivore	**71%** 12/17	**Intermediate**
river shiner *Notropis blennius*	slow riffles in rivers & streams **glides, riffles**	generalized invertivore	**70%** 7/10	**S2** (SD) **Intermediate**
river carpsucker *Carpiodes carpio*	pools in rivers & streams ***near banks or backwaters***	omnivore	**70%** 7/10	**Intermediate**
slimy sculpin *Cottus cognatus*	riffles in rivers & streams **nest under rocks in glides & riffles**	generalized invertivore	**70%** 7/10	**Intolerant**
southern brook lamprey *Ichthyomyzon gagei*	riffles in streams **glides, riffles**	do not eat, juveniles filter feed	**67%** 4/6	**SC** (MN) **Intolerant**
Ozark minnow *Notropis nubilus*	riffles in rivers & streams **glides, riffles, hornyhead chub nests**	omnivore, mostly vegetation	**67%** 4/6	**T** (WI), **SC** (MN) **Intolerant**
warmouth *Lepomis gulosus*	pools in low gradient streams, lakes ***nest near wood or vegetation***	generalized invertivore, piscivore	**67%** 2/3	**SC** (MN, Canada, ON) **Intermediate**
freshwater drum *Aplodinotus grunniens*	pools in river, lakes ***pelagic***	generalized invertivore, piscivore	**64%** 18/28	**Intermediate**
largescale stoneroller *Campostoma oligolepis*	slow riffles in rivers & streams **glides, riffles**	herbivore/benthic invertivore	**64%** 7/11	**SCP3** (ND) **Intermediate**
banded darter *Etheostoma zonale*	riffles in rivers & streams **glides & riffles**	generalized invertivore	**64%** 7/11	**Intolerant**
American brook lamprey *Lethenteron appendix*	riffles in streams **glides, riffles**	do not eat, juveniles filter feed	**63%** 5/8	**T** (IA) **Intolerant**
channel catfish *Ictalurus punctatus*	pools in rivers **nests in cavities**	piscivore, generalized invertivore	**61%** 19/31	**Intermediate** Occasionally stocked game species
bigmouth buffalo *Ictiobus cyprinellus*	pools in rivers & streams, lakes **backwaters & floodplains**	planktivore, benthic invertivore	**61%** 11/18	**SC** (Canada), **Intermediate**
mimic shiner *Notropis volucellus*	shallow pools in rivers & streams ***vegetation***	generalized invertivore	**61%** 14/23	**Intolerant**
quillback *Carpiodes cyprinus*	pools in rivers & streams **backwaters**	omnivore	**60%** 15/25	**S3** (SD) **Intermediate**

Table 4 (cont.). Fish species listed by percent absence upstream of dam barriers analyzed and listed in Table 1. Table is sorted by percent absence. Fish habitat and feeding data from Aadland & Kuitunen 2005 and Becker 1983. Conservation status: **E** = Endangered, **T** = Threatened, **SC** = Special Concern, **Ext** = Extirpated from Minnesota DNR (MN), Iowa DNR (IA), Wisconsin Natural Heritage Working List (WI), North Dakota Game & Fish Department (ND) Species of Conservation Priority, **SCP**, Levels 1 - 3), South Dakota Game Fish & Parks (SD, State Rank **S1 - S5**), U.S. Fish & Wildlife Service, and Government of Canada (Canada, Ontario=ON, **PSC**=Proposed Special Concern). Species tolerance ratings from US EPA.

	Common Name *Scientific Name*	Adult Habitat *Spawning Habitat*	Adult Feeding Habits	% Absent Upstream of Barriers # Absent / Sample size	Conservation Status *Tolerance* Management (if any)
Vulnerable Species : 50% to 74% Absence	**carmine shiner** *Notropis percobromus*	riffles in rivers & streams **glides, riffles**	omnivore	**59%** 13/22	**T** (Canada), **S2** (SD), **SCP3** (ND) Intolerant
	river redhorse *Moxostoma carinatum*	fast runs in rivers **glides, riffles**	benthic invertivore	**55%** 6/11	**T** (WI), **SC** (Canada, ON) Intolerant
	brook trout *Salvelinus fontinalis*	coldwater rivers & lakes **glides & riffles in rivers & streams**	generalized invertivore, piscivore	**54%** 7/13	Intolerant Widely stocked game species
	emerald shiner *Notropis atherinoides*	shallow pools in rivers & streams **glides, riffles**	generalized invertivore	**52%** 12/23	Intermediate
	northern brook lamprey Ichthyomyzon fossor	pools in streams **glides, riffles**	don't eat, juveniles filter feed	**50%** 2/4	**SC** (MN, ON), **PSC** (Canada) Intolerant
	red shiner *Cyprinella lutrensis*	ubiquitous in rivers & streams **sunfish nests in vegetated backwaters**	omnivore	**50%** 1/2	Tolerant
	redside dace *Clinostomus elongatus*	riffles & raceways in streams **glides, riffles, creek chub nests**	benthic invertivore	**50%** 3/6	**T** (ON), **SC** (MN, WI), **PSC** (Canada) Intolerant
	weed shiner *Notropis texanus*	pools in clearwater streams & lakes **unknown**	omnivore	**50%** 5/10	**E** (IA), **SC** (WI) Intolerant
	silver redhorse *Moxostoma anisurum*	runs, glides & pools in rivers & streams **glides, riffles**	benthic invertivore	**50%** 14/28	Intermediate
	Muskellunge *Esox masquinongy*	pools in rivers, lakes **vegetated backwaters & side channels**	piscivore, top predator	**50%** 6/12	Intolerant Widely stocked game species
	ninespine stickleback *Pungitius pungitius*	headwater streams, shoals of large lakes **nests of vegetation between rocks**	omnivore	**50%** 1/2	Intermediate
Somewhat Vulnerable Species : 25% to 49% Absence	**greater redhorse** *Moxostoma valenciennesi*	runs & glides in rivers & streams **glides, riffles**	benthic invertivore	**47%** 7/15	**T** (WI) Intolerant
	mottled sculpin *Cottus bairdii*	riffles in rivers & streams **nest tunnel under rocks in riffles**	generalized invertivore, piscivore	**46%** 6/13	Intolerant
	spotfin shiner *Cyprinella spiloptera*	slow riffles in rivers & streams **crevices, glides, riffles**	generalized invertivore	**44%** 12/27	Intermediate
	blackchin shiner *Notropis heterodon*	shallow pools, clearwater streams, lakes **vegetation**	generalized invertivore	**43%** 6/14	Intolerant
	burbot *Lota lota*	rivers (pools) & lakes **pelagic over gravel or rocks**	piscivore	**42%** 8/19	**T** (IA), **SCP2** (ND) Intermediate
	slenderhead darter *Percina phoxocephala*	fast riffles in rivers & streams **glides & riffles**	generalized invertivore	**42%** 8/19	**SX** (SD) Intolerant
	sand shiner *Notropis stramineus*	slow riffles in rivers & streams **glides, riffles**	surface and water column invertivore	**40%** 12/30	Intermediate
	redfin shiner *Lythrurus umbratilis*	pools in headwater streams **nests in glides & riffles**	benthic invertivore	**40%** 2/5	**T** (WI), **SC** (MN) Intermediate

Somewhat Vulnerable Species : 25% to 49% Absence

Common Name *Scientific Name*	**Adult Habitat** *Spawning Habitat*	**Adult Feeding Habits**	**% Absent Upstream of Barriers** # Absent / Sample size	**Conservation Status** *Tolerance* **Management** (if any)
orangespotted sunfish *Lepomis humilis*	pools in rivers, streams **nest in backwaters & bays**	generalized invertivore	**39%** 7/18	**SC** (ON), **PSC** (Canada) **Intermediate**
spottail shiner *Notropis hudsonius*	slow riffles, rivers & streams **glides, riffles**	generalized invertivore	**37%** 7/19	**Intolerant** Common bait species
shorthead redhorse *Moxostoma macrolepidotum*	runs & glides in rivers & streams **glides, riffles**	benthic invertivore	**34%** 11/32	**Intermediate**
blacknose shiner *Notropis heterolepis*	pools in clearwater streams & lakes **vegetation**	generalized invertivore	**33%** 5/15	**E** (SD), **T** (IA), **SCP3** (ND) **Intolerant**
suckermouth minnow *Phenacobius mirabilis*	slow riffles in rivers & streams **glides, riffles**	generalized invertivore	**33%** 2/6	**SC** (MN), **SH** (SD) **Intermediate**
golden redhorse *Moxostoma erythrurum*	runs & pools in rivers & streams **glides, riffles**	benthic invertivore	**33%** 10/30	**SH** (SD) **Intermediate**
stonecat *Noturus flavus*	riffles & runs in rivers & streams **glides, riffles**	generalized invertivore, piscivore	**33%** 10/30	**Intolerant**
trout-perch *Percopsis omiscomaycus*	pools in rivers, large lakes **glides & riffles in streams**	generalized invertivore	**33%** 5/15	**SCP2** (ND), **S2** (SD) **Intermediate**
rainbow darter *Etheostoma caeruleum*	fast riffles in rivers & streams **glides and riffles**	generalized invertivore	**33%** 4/12	**Intolerant**
blackside darter *Percina maculata*	slow riffles in rivers & streams **glides & riffles**	generalized invertivore	**33%** 10/30	**S2** (SD) **Intermediate**
bowfin *Amia calva*	pools in rivers, connected lakes **nest, vegetated backwaters**	piscivore	**31%** 4/13	**Intermediate**
northern redbelly dace *Chrosomus eos*	clear, headwater streams & ponds **filamentous algae**	herbivore	**29%** 6/21	**T** (SD), **SCP2** (ND) **Intermediate**
least darter *Etheostoma microperca*	clearwater streams, lakes & ponds **vegetation, roots or rubble**	generalized invertivore	**29%** 2/7	**E** (IA), **SC** (MN, WI) **Intolerant**
logperch *Percina caprodes*	fast riffles in rivers & streams, large lakes **glides, riffles, shoals**	generalized invertivore	**29%** 6/21	**SCP3** (ND), **S3** (SD) **Intermediate**
smallmouth bass *Micropterus dolomieu*	raceways in rivers, lakes **nest in backwaters and bays**	generalized invertivore, piscivore	**27%** 6/22	**Intolerant** Widely stocked game species
fantail darter *Etheostoma flabellare*	fast riffles in rivers & streams **glides & riffles**	generalized invertivore	**27%** 3/11	**Intermediate**
Iowa darter *Etheostoma exile*	shallow pools in rivers & streams, lakes **nest in riffles or in vegetation**	generalized invertivore	**26%** 7/27	**Intolerant**
chestnut lamprey *Ichthyomyzon castaneus*	riffles & pools in rivers & streams **glides, riffles**	parasite on fish	**25%** 4/16	**T** (IA), **SCP3** (ND), **PSC** (Canada) **Intermediate**
central stoneroller *Campostoma anomalum*	slow riffles in rivers & streams **glides, riffles**	herbivore/benthic invertivore	**25%** 5/20	**SCP3** (ND) **Intermediate**

Table 4 (cont.). Fish species listed by percent absence upstream of dam barriers analyzed and listed in Table 1. Table is sorted by percent absence. Fish habitat and feeding data from Aadland & Kuitunen 2005 and Becker 1983. Conservation status: **E** = Endangered, **T** = Threatened, **SC** = Special Concern, **Ext** = Extirpated from Minnesota DNR (MN), Iowa DNR (IA), Wisconsin Natural Heritage Working List (WI), North Dakota Game & Fish Department (ND) Species of Conservation Priority, **SCP**, Levels 1 - 3), South Dakota Game Fish & Parks (SD, State Rank **S1 - S5**), U.S. Fish & Wildlife Service, and Government of Canada (Canada, Ontario=ON, **PSC**=Proposed Special Concern). Species tolerance ratings from US EPA.

	Common Name *Scientific Name*	Adult Habitat *Spawning Habitat*	Adult Feeding Habits	% Absent Upstream of Barriers # Absent / Sample size	Conservation Status *Tolerance* Management (if any)
Somewhat Vulnerable	**finescale dace** *Chrosomus neogaeus*	cool, headwater streams & ponds **logs & branches in backwaters or bays**	generalized invertivore	**25%** 4/16	**E** (SD), **SCP3** (ND) Intermediate
Somewhat Vulnerable	**lake whitefish** *Coregonus clupeaformis*	deepwater lakes **glides& riffles in streams, lake shoals**	water column invertivore, piscivore	**25%** 1/4	Intermediate
Somewhat Vulnerable	**walleye** *Sander vitreus*	pools in rivers, lakes **glides, riffles & shoals**	piscivore	**25%** 8/32	Intermediate Widely stocked game species
Least Vulnerable Species : 0% to 24% Absence	**Northern pearl dace** *Margariscus nachtriebi*	pools in cool, headwater streams **glides, riffles**	omnivore	**24%** 4/17	**E** (IA), **T** (SD), **SCP1** (ND) Intermediate
Least Vulnerable Species : 0% to 24% Absence	**bigmouth shiner** *Notropis dorsalis*	shallow pools in rivers & streams **unknown**	generalized invertivore	**23%** 7/31	Intermediate
Least Vulnerable Species : 0% to 24% Absence	**pugnose shiner** *Notropis anogenus*	clearwater streams & lakes **vegetation**	herbivore, crustaceans	**22%** 2/9	**E** (IA, Canada, ON), **T** (MN, WI), **SCP3** (ND) Intolerant
Least Vulnerable Species : 0% to 24% Absence	**banded killifish** *Fundulus diaphanus*	backwaters in clear rivers, lakes **vegetation**	generalized invertivore	**21%** 3/14	**E** (SD), **SC** (Canada) Tolerant
Least Vulnerable Species : 0% to 24% Absence	**northern hogsucker** *Hypentelium nigricans*	fast runs in rivers & streams **glides, riffles & rapids**	benthic invertivore	**21%** 4/19	**SH** (SD) Intolerant
Least Vulnerable Species : 0% to 24% Absence	**white crappie** *Pomoxis annularis*	pools in river, & lakes **nest in backwaters & bays**	planktivore, piscivore	**21%** 4/19	Intermediate Widely stocked game species
Least Vulnerable Species : 0% to 24% Absence	**central mudminnow** *Umbra limi*	headwater streams **flooded ephemeral wetlands**	generalized invertivore, piscivore	**20%** 5/25	**S1** (SD) Tolerant
Least Vulnerable Species : 0% to 24% Absence	**tulllibee** *Coregonus artedi*	deepwater lakes **pelagic over lake shoals**	water column inverti-vore, piscivore	**20%** 2/10	Intermediate
Least Vulnerable Species : 0% to 24% Absence	**lake trout** *Salvelinus namaycush*	deepwater lakes **deep shoals**	piscivore	**20%** 1/5	Intermediate Occasionally stocked game species
Least Vulnerable Species : 0% to 24% Absence	**brassy minnow** *Hybognathus hankinsoni*	pools in rivers & streams **vegetated backwaters**	herbivore/benthic invertivore	**19%** 5/27	Intermediate
Least Vulnerable Species : 0% to 24% Absence	**yellow bullhead** *Ameiurus natalis*	clear rivers, streams, lakes, & ponds **nests in cavities**	generalized invertivore, piscivore	**17%** 4/23	**SCP3** (ND) Intermediate*
Least Vulnerable Species : 0% to 24% Absence	**tadpole madtom** *Noturus gyrinus*	pools in streams **under rocks**	generalized invertivore	**17%** 5/29	Intermediate
Least Vulnerable Species : 0% to 24% Absence	**rock bass** *Ambloplites rupestris*	pools in rivers & streams, lakes **nest in backwaters & bays**	generalized invertivore, piscivore	**17%** 5/29	Intolerant
Least Vulnerable Species : 0% to 24% Absence	**golden shiner** *Notemigonus crysoleucas*	pools in rivers, lakes & ponds **vegetated backwaters & bays**	omnivore	**17%** 4/24	Tolerant Widely stocked bait species
Least Vulnerable Species : 0% to 24% Absence	**southern redbelly dace** *Chrosomus erythrogaster*	clear, headwater streams & ponds **glides, riffles**	herbivore	**17%** 1/6	**S1** (SD) Intermediate
Least Vulnerable Species : 0% to 24% Absence	**blacknose dace** *Rhinichthys atratulus*	riffles and pools in rivers & streams **glides, riffles**	generalized invertivore	**17%** 5/30	Tolerant
Least Vulnerable Species : 0% to 24% Absence	**bluntnose minnow** *Pimephales notatus*	slow riffles in rivers & streams, lakes, ponds **underside of objects**	omnivore	**16%** 4/25	Tolerant

Attachment

Common Name *Scientific Name*	**Adult Habitat** *Spawning Habitat*	**Adult Feeding Habits**	**% Absent Upstream of Barriers** # Absent / Sample sizee	**Conservation Status** *Tolerance* **Management** (if any)
common shiner *Luxilus cornutus*	pools in rivers & streams **glides, riffles, hornyhead chub nests**	omnivore	**16%** 5/32	**Intermediate** Common bait species
brown bullhead *Ameiurus nebulosus*	rivers, streams, lakes, & ponds **nests in cavities**	generalized invertivore, piscivore	**13%** 3/23	**Intermediate***
johnny darter *Etheostoma nigrum*	ubiquitous in rivers, streams & lakes **nest in backwaters in vegetation**	generalized invertivore	**13%** 4/32	**Intermediate**
green sunfish *Lepomis cyanellus*	pools in rivers & streams, lakes **nest in backwaters & bays**	generalized invertivore, piscivore	**12%** 3/25	**Tolerant**
brook stickleback *Culaea inconstans*	shallow pools in streams, wetlands **nests in vegetation**	omnivore	**10%** 3/30	**Intermediate**
bluegill *Lepomis macrochirus*	pools & backwater in river, lakes **nest in backwaters & bays**	generalized invertivore	**10%** 3/30	**Intermediate** Widely stocked game species
longnose dace *Rhinichthys cataractae*	fast riffles in rivers & streams **glides, riffles**	generalized invertivore	**10%** 2/21	**Intolerant**
northern pike *Esox lucius*	pools in rivers & streams, lakes **vegetated backwaters & wetlands**	piscivore, top predator	**9%** 3/32	**Intermediate** Widely stocked game species
hornyhead chub *Nocomis biguttatus*	ubiquitous in streams **gravel nests in glides, riffles**	benthic invertivore	**8%** 2/25	**SCP3** (ND), **S3** (SD) **Intolerant** Common bait species
largemouth bass *Micropterus salmoides*	pools & backwaters in rivers, lakes **nest in backwaters & bays**	top predator, piscivore	**7%** 2/30	**Intermediate** Widely stocked game species
yellow perch *Perca flavescens*	pools in rivers & lakes **vegetation & brush**	generalized invertivore, piscivore	**7%** 2/30	**Intermediate**
black bullhead *Ameiurus melas*	rivers, streams, lakes, & ponds **nests in cavities**	generalized invertivore, piscivore	**6%** 2/31	**Tolerant***
white sucker *Catostomus commersonii*	rivers, streams, & lakes **glides, riffles**	omnivore	**6%** 2/32	**Tolerant** Widely stocked bait species
creek chub *Semotilus atromaculatus*	pools in rivers and streams **glides, riffles**	generalized invertivore, piscivore	**6%** 2/32	**Tolerant**
pumpkinseed *Lepomis gibbosus*	pools in rivers, lakes **nest in backwaters & bays**	generalized invertivore	**5%** 1/22	**Intermediate**
black crappie *Pomoxis nigromaculatus*	pools in rivers, lakes **nest in backwaters & bays**	planktivore, piscivore	**3%** 1/29	**Intermediate** Widely stocked game species
fathead minnow *Pimephales promelas*	rivers, streams, lakes, & ponds **underside of objects**	omnivore	**3%** 1/31	**Tolerant** Widely stocked bait species
lake chub *Couesius plumbeus*	Great Lakes **streams and shoals**	omnivore	**0%** 0/1	**SC** (MN), **S1** (SD) **Intermediate**
Northern longear sunfish *Lepomis peltastes*	clearwater lakes **nest in bays**	generalized invertivore	**0%** 0/3	**T** (WI), **SC** (MN) **Intolerant**

Least Vulnerable Species : 0% to 24% Absence

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