Risk Assessment and Decision Analysis Support for Invasive Mussel Management for the St. Croix Basin and Adjacent Upper Mississippi River

Final Report

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Introduction

The purpose of this project is to (1) assess the vulnerability of surface waters within the St. Croix River Basin to invasion and establishment of persistent populations of invasive mussels, (2) develop a risk-based decision framework to assist in the control and management of invasive mussels, and (3) evaluate the effectiveness of alternative technologies in controlling or reducing the rate of mussel spread throughout the basin, and Navigation Pools 2−4 of the Upper Mississippi River (UMR). The potential effects of zebra and quagga mussel establishment on populations of the endangered Higgins eye pearlymussel (*Lampsilis higginsii*) and winged mapleleaf mussel (*Quadrula fragosa*) are also addressed as part of this study.

The project report begins with a brief summary of the life history, biology, and ecology of the dreissenid mussels. This information provides background for the subsequent presentation of the risk-based decision support model for mussel establishment. Preliminary estimates of the risk of establishment are provided for 70 lakes and stream segments within the St. Croix Basin for which adequate data were available. The report continues with an evaluation of the model and a recommended approach for using the model to evaluate the effectiveness of alternative mussel control strategies. The report concludes with recommendations for future work to refine the model and identification of data needed to accomplish the recommended modifications to the risk-based decision model.

Background

The zebra mussel (*Dreissena polymorpha*) and the quagga mussel (*Dreissena bugensis*) are freshwater mussels native to the Black and Caspian Sea region of Asia (Figure 1). These species were apparently introduced to Lake St. Clair in the Great Lakes via ship water ballast in the early to mid-1980s (Mackie et al. 1989). Following this introduction, these two invasive mussel species have spread throughout the Lawrentian Great Lakes (McMahon et al. 1993). Relevant to this project, zebra mussels have subsequently spread into the Illinois and Mississippi Rivers and into smaller lakes and rivers in the states that surround the Great Lakes. These mussels have spread extensively throughout the surface waters of the eastern United States (Figure 2).

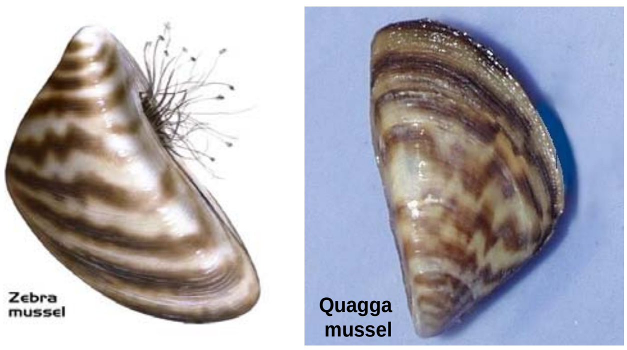

Figure 1. Zebra (*Dreissena Polymorpha*) and Quagga (*Dreissena bugensis*) mussels.

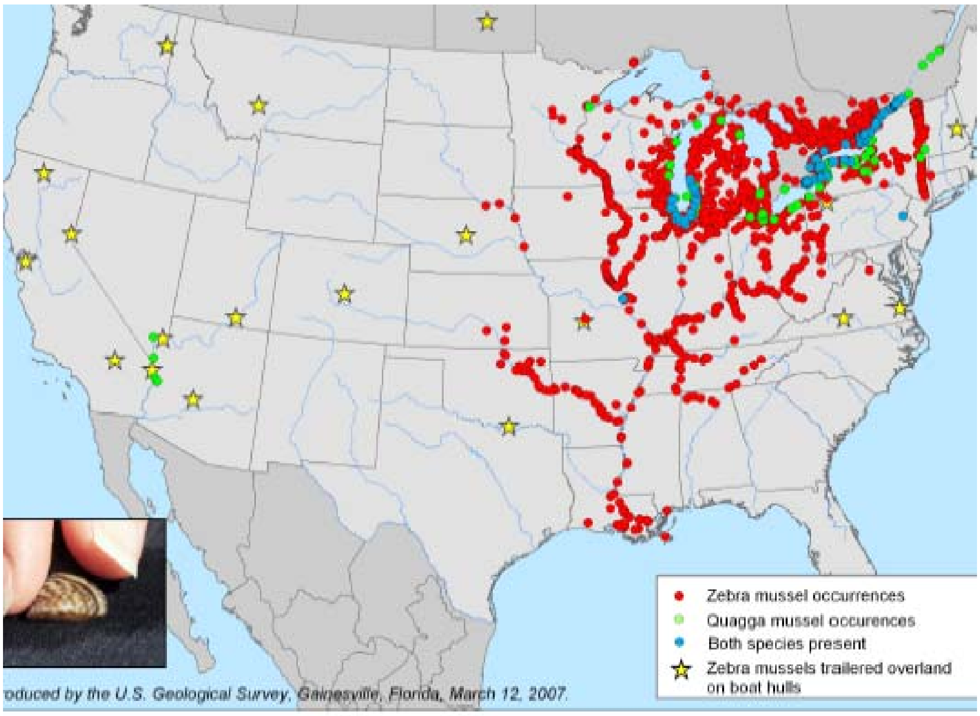

Figure 2. Distribution of zebra and quagga mussels in the United States in 2007. (Source: USGS Aquatic Nuisance Species web site).

Biology and Ecology of the Zebra and Quagga Mussels

The life cycle of the zebra mussel consists of larval, juvenile, and adult stages (Figure 3). Following fertilization, the subsequent larval stages (trochophore, straight-hinged veliger, and umbonal veliger) are planktonic. After an initial non-feeding phase (~2−9 days), the larvae develop intestines and a swimming organ (velum), and begin feeding. Veligers develop to a pediveliger stage characterized by the initial development of a foot. The pediveligers grow to shell lengths of ~200–240 μm and then settle onto available substrate (e.g., sediments, infrastructure, and other mussels) and undergo metamorphosis to the juvenile stage (Mackie et al. 1989, Sprung 1993, Mackie and Schloesser 1996). Juvenile mussels can then grow into sexually mature adults. However, as few as 2 percent of juvenile zebra mussels survive to adulthood (Miller et al. 1992).

The adults release eggs and sperm into the water to continue the cycle. In North America, zebra mussels can mature in their first year (8 to 10 mm shell length) and have exceedingly high fecundities. Individual females can produce from 30,000 to 1,610,000 eggs/year (Mackie et al., 1989; Borcherding, 1992). Zebra mussels spawn when the water temperatures are approximately 14º to 20º C. Spawning can continue to late summer or early fall. In some regions, the reproductive process occurs later, with synthesis of gametes peaking in spring and spawning beginning in late summer (Haag, and Garton 1992). Egg and sperm release decrease in late September to mid-October in the northeastern United States (Claudi and Mackie 1994). Zebra mussels can apparently reproduce at temperatures as low as 2.5°C (Mills et al. 1999).

The recorded life spans of zebra mussels vary substantially. Zebra mussels appear to live 3–5 years in Polish lakes, 3.5 years in British reservoirs, 6–7 years in Swiss lakes, and 6–9 years in some Russian reservoirs (Ackerman et al. 1994). Adult zebra mussels live 2–3 years in temperate climates such as the St. Croix Basin (ERDC Environmental Laboratory, U.S. Army Corps of Engineers, Vicksburg, Mississippi).

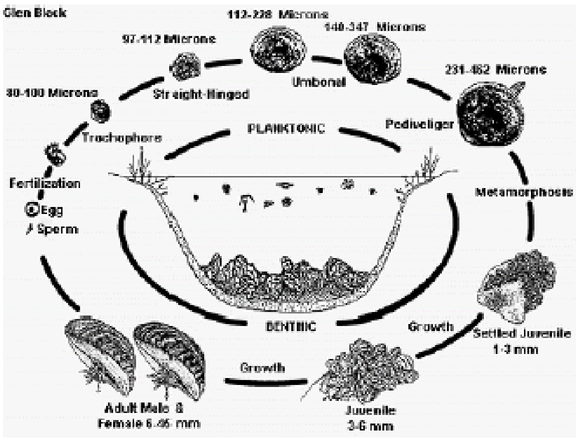

Figure 3. Zebra mussel life cycle showing larval, juvenile, and adult stages.

(Source:http://el.erdc.usace.army.mil/zebra/zmis/zmishelp4/life_cycle.html)

The quagga mussel has a similar life cycle. Quagga mussels were found in Lake Michigan in 1997, and initially were confused with zebra mussels. Quagga mussels appear to differ in their habitat preferences. Quagga mussels' colonies thrive from warm, shallow water to colder, depths >100 m. Young quagga mussels have been observed in water as cool as 8º C. Pligin (unpublished data) found that zebra mussels were most abundant in water depths of 4–12 m, but were rare in water depth >16 m in two Russian reservoirs. The zebra mussel is rarely found at depths exceeding 50 m in the Great Lakes (Ohio Sea Grant 1994).

The general life cycle characteristics of these species are of interest in this study because environmental factors that define habitat quality and influence the successful completion of the cycle can be used to forecast the vulnerability of non-infested surface waters to invasion and establishment of zebra and quagga mussel populations. Habitat factors determined to be important in the successful invasion and establishment of these mussels have been used as one component of an integrated risk-based decision support model developed for this project.

Individual Mussel Growth

Individual dreissenid mussel growth varies among watersheds in relation to habitat quality and food availability. A daily specific growth rate for the zebra mussel has been estimated as 0.00531 g/g/day and respiration rate as 0.00133 g/g/day (Fanslow et al. 1995). Zebra mussels have been observed to grow as much as 350 mg per year (Smirnova and Vinogradov 1990). Throughout Europe, zebra mussels can reach maximum shell lengths of 35–40 mm, and maximum growth rates can reach 0.5 mm/day and 15–20 mm/year (Mackie et al. 1989). Zebra mussels in the Great Lakes appear to be of similar size, (~40 mm maximum length), but grow faster (25 mm/year) than European zebra mussels (Ram and Walker 1993).

Rapid growth rate results in part from the filtration capacity of an individual zebra mussel. Zebra mussels consume bacteria, algae, zooplankton, and organic detritus ranging from particles <0.001 mm in length to algal colonies >3.0 mm in length. However, these mussels feed preferentially on 0.001–0.05 mm particles. Mean filtration rates have been measured as 16.2 mL/mg/h (range 4.0−40.7). An adult mussel can filter as much as one liter of water per day (Ohio Sea Grant 1994).

Seston concentration importantly influences the filtration efficiency of *Dreissena.* Ingestion rate increases linearly with food concentration until an incipient limiting concentration is reached (Walz 1978, Sprung and Rose 1988). Walz (1978) estimated the incipient limiting level at ~2 mg C/L. Mussel filtering rate declines exponentially at seston concentrations that exceed the incipient limiting level (Sprung and Rose 1988). Filtration rates can also vary in relation to seston quality. For example, measured filtration rates decreased when chlorophyll *a* decreased from 7.4 µg/L to 2.2 µg/L (Fanslow et al. 1995).

The growth rates and biological processes of zebra mussels are strongly temperature dependent and can be described by the following equation (Jantz and Neumann 1998):

$R = R_0 * \theta^{(t - t_{ref})}$,

where, R = process rate at temperature t,

R_0 = process rate at optimal temperature t_{ref},

θ = temperature coefficient = 1.08 for growth and 1.04 for respiration,

t_{ref} = 22 °C.

Walz (1978) observed that the temperature dependence of individual *Dreissena* filtration rate (L/h) or ingestion rate (mg C/h) could be described by a bell-shaped curve, with an optimal value of 12.5° C.

This discussion of individual growth has focused on the zebra mussel. However, Baldwin et al. (2002) reported that quagga mussels grow less efficiently at lower temperature (e.g., 6° C), but can exceed zebra mussel growth rate by >two-fold under at warmer temperatures (23° C). Quagga mussels can grow to twice the size and body weight of zebra mussels (Mackie and Schloesser 1996).

Population Sizes

The combined rapid growth rates and high fecundities of invasive dreissenids contribute to their excessive abundance following introduction to previously uninfested surface waters. For example, MacIsaac (1994) reported zebra mussel biomass values as high as 1,500 g dry mass/m^2 on rock surfaces in western Lake Erie. Mussel densities up to 700,000 individuals/m^2 have been measured (Mackie and Schloesser 1996). Abundances >20,000 individuals/m^2 have been reported for Lake Pepin, UMRS Navigation Pool 4 (USACOE 2003). Clearly, ecological and economic consequences of excessively large populations of zebra mussels justify the development of effective mussel management and control alternatives.

Risk of Invasive Mussel Establishment

Ecological risk assessment has become a common framework and tool for assessing environmental impacts and management alternatives in relation to ecosystem management and restoration (Bartell et al. 1992). Risk assessment is also a comprehensive process that identifies the risks and relevant information, analyzes pertinent data and evaluates management alternatives [Committee on Environment and Natural Resources (CENR) 1999].

The establishment of an invasive species is operationally defined as the development of persistent populations in a newly colonized (invaded) system. Clearly, establishment requires (1) some mechanism(s) for introducing the novel organism to an uninfested system, and (2) the ability of newly introduced exotic organisms to grow, reproduce, and persist. The following sections examine mechanisms for invasive mussel introduction to uninfested waters and discuss the development of persistent populations following introduction.

Mechanisms for introduction

The life history and biology of the zebra and quagga mussels confer definite advantages in increasing the probability that these mussels will be introduced to previously uninfested surface waters within the St. Croix Basin. Larval zebra mussel veligers are microscopic in size, planktonic, and easily transported by flowing water. Infested upstream rivers, lakes, and reservoirs can serve as a continuing source of veligers to downstream systems that are physically connected by streams and rivers (Bobeldyk et al. 2005, Stoeckel et al. 2004). An important implication is that geographically distant systems might be more readily infested by connected upstream sources than unconnected surface waters located closer to the source.

In addition to veliger transport and distribution by natural flows among connected surface waters, commercial navigation and recreational boating (Johnson et al. 2001, Johnson and Carlton 1996) provide mechanisms to move adult mussels from infested to uninfested systems. Because of their ability to adhere to objects, adult zebra mussels can easily be transported from lake to lake on the hull of a recreational boat. Adults attached to the hull of a commercial vessel can be transported long distances throughout major river systems, including the Mississippi River (e.g., Figure 2). Importantly, zebra mussel adults can survive out of water for up to 10 days if they are in a shaded, humid area (New York Sea Grant Extension Fact Sheet 1994, Carlton 1993, Johnson 1997). Additional means of artificial transport of veligers (and adults) exist in the form of live wells on boats and bait buckets moved by fishermen from one system to another. Piers and boat docks infested with zebra mussels are sometimes sold, disassembled, and reconstructed in previously uninfested waters. These artificial means of transport are opportunities for the inadvertent transfer of veligers and adult mussels.

Data that describe and quantify these pathways for zebra mussel infestation of surface waters can be used to develop an inoculation component of a risk-based decision support model for invasive mussels in the St. Croix Basin.

Establishment

To establish a persistent population, mussels introduced to a previously uninfested system must be able to survive, grow, and reproduce. The physical chemical environment must be inhabitable and favorable for successful reproduction. Food must be available in sufficient quantity and quality. A set of important habitat factors that determine habitat quality for zebra mussels has been defined (Cohen and Weinstein 2001). This set includes total hardness, conductivity, pH, salinity, Secchi disc depth, water depth, water temperature, current velocity, and concentrations of calcium, potassium, and ammonia. Chlorophyll *a* concentration is also included as a surrogate for food availability. Water temperature includes both temperature requirements for growth and survival, as well as temperatures conducive to successful spawning.

Data that describe and quantify these important habitat variables can be used to estimate the likelihood that introduced mussels will become established in a newly invaded system. These parameters were used to develop a habitat suitability component of a risk-based decision support model for invasive mussels in the St. Croix Basin.

Consequences of Invasive Mussel Establishment

The demonstrated economical and ecological consequences of dreissenid mussel establishment justify the need for mussel management and control. The invasion and establishment of zebra and quagga mussels can dramatically change aquatic ecosystems (MacIsaac et al. 1991). The ecological and economical impacts of these two dreissenids include alteration of food webs, potential impairment of native mussels, changes in water quality, increased levels of contaminants in food chains, and damage to water intake pipes and other infrastructure (Kovalak et al. 1993, Lange and Wittmeyer 1996, Vanderploeg 2002).

Food web alterations

Zebra mussels can produce several direct and indirect ecological impacts on aquatic food webs (Figure 4). The invasion of zebra mussels can divert biomass production from pelagic to benthic food webs in lakes and rivers (MacIsaac 1994). Zebra mussels can also shift an aquatic ecosystem from a turbid and phytoplankton-dominated condition to clear and macrophyte-dominated condition. Zebra mussels can directly or indirectly impact planktivorous and piscivorous fish by altering food supply or habitat quality.

Schematic of observed (solid line) and potential (dotted line) impacts of zebra mussels in freshwater communities based on European and North American studies. Taxa benefiting from zebra mussel invasion are indicated with a (+) symbol on the arrowhead, those adversely affected by a (-) symbol. Strong interactions are denoted by thicker arrows (MacIsaac 1996).

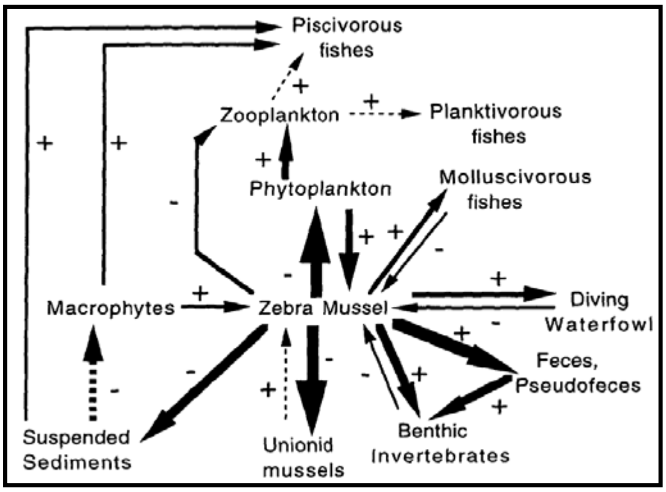

Figure 4. Ecological effects of zebra mussels (MacIsaac 1994).

One of commonly observed effect of zebra mussel establishment in lakes and rivers is greatly diminished phytoplankton biomass. Phytoplankton biomass (measured as chlorophyll *a* concentration) declined by as much as 60 percent in the western and west-central basins of Lake Erie between 1988 and 1991 following introduction of the zebra mussel (Leach 1993, Fahnenstiel et al. 1995). Similar impacts have been observed in smaller systems as well.

For example, phytoplankton biovolume declined by ~46 percent in a pond stocked with zebra mussels compared to a reference pond that did not have zebra mussels (Reeders et al. 1993). Additionally, the removal of seston by zebra mussel filter-feeding can increase water clarity and stimulate the growth of benthic algae and macrophytes (Leach 1993, Karatayev et al. 1997).

Zebra mussels utilize food resources similar to those required by native zooplankton and benthic invertebrates. Small rotifers in western Lake Erie decreased by ~75 percent in abundance following the establishment of large populations of zebra mussels (Leach 1993). *Diporeia* is a small, shrimp-like organism which lives in the sediments and feeds on algae that settle from the water column (Nalepa et al. 2006). Following the establishment of zebra mussels in the 1980's, the numbers of *Diporeia* in the Great Lakes have declined substantially. Based on the National Oceanic and Atmospheric Administration Great Lakes Environmental Research Laboratory (2006), between 1994 and 2000, *Diporeia* densities declined from a lakewide average of 5,200 to 1,800 individuals/m^2. In 2005, the average was only 300 individuals/m^2. Whitefish, alewife, bloater, smelt and sculpin directly depend on *Diporeia* as a food source. Therefore, declines in *Diporeia* might be linked to reduced numbers and condition of these fish species, as well as young lake trout. Declines in these fish might correspondingly reduce the abundance of sport fish such as adult salmon, trout and walleye. It is noteworthy that the most obvious system wide effects of zebra mussels have been in large lakes or relatively lake-like sections of run of river impoundments.

In contrast, the abundance and diversity of benthic invertebrates can increase in the presence of high densities of zebra mussels. The mussel shells greatly enhance the amount of habitat available to small crustaceans, snails, and other animals (Cohen and Weinstein 1998a). However, this habitat might likely be populated by oligochaetes and chironomids, which are often characteristic of degraded habitat. Populations of crayfish and other invertebrates, which prey on these worms, can in turn increase (Karatayev et al.1997). Another habitat factor of potential importance is likely to be the "pelagic" to benthic shift in organic carbon due to suspended organic material being bundle in mucus and deposited on the bottom due to filtration and either ingestion followed by defecation or ejection as pseudofeces.

Impacts on native mussels

Concerns regarding the potential impacts of invasive mussels on the continued viability of native unionid mussels further underscore the need for a risk-based decision support model. These concerns are especially important for threatened or endangered species such as winged mapleleaf mussel and the Higgins eye pearlymussel (Figure 5). The reasons for decreases in the native mussel populations appear related to declines in water quality and low oxygen levels (Nalepa et al. 1993). However, the impacts of water quality parameters on unionid mussels are not well understood. The observations reported by Nalepa et al. (1993) for the Great Lakes might not apply to the Upper Mississippi River System. Schloesser et al. (1998) found that high mortality of unionids can occur between 4 and 8 years after initial invasion by dreissenids. The difference in time to near-total mortality of unionids in different habitats could be attributed to differences in the time of invasion and successful settling of juvenile zebra mussels in different water bodies.

It has also been observed that impacts on unionids include attachment in sufficient numbers of zebra mussels to the shells of these native mussels that the infested mussels cannot travel or burrow. Attachment of approximately 100 zebra mussels to an individual unionid inhibits shell opening, feeding, respiring. In some instances, the native mussels cannot fully close their shells. Davis (M. Davis, Minnesota Department of Natural Resources, personal communication, 2007) reports that ~56 percent of the unionid mussels inhabiting Pool 5 in the UMRS have attached zebra mussels.

Winged mapleleaf (*Quadrula fragosa*)

Higgins eye (*Lampsilis higginsi*)

Figure 5. Illustrations of native unionids: winged mapleleaf mussel (*Quadrula fragosa*) and the Higgins eye pearlymussel (*Lampsilis higginsii*).

The winged mapleleaf mussel

The winged mapleleaf mussel (Figure 5) is a federally listed endangered species. Once found throughout many Midwestern rivers, only three known populations exist, one of which inhabits in a 10-mile stretch of the St. Croix National Scenic Riverway that borders Minnesota and Wisconsin. This species was found historically in riffles or on gravel bars in medium to large clear-water streams. These habitats have been largely lost to development of impoundments, channelization, soil erosion, and sediment accumulation from land-use practices [United States Fish and Wildlife Service (USFWS) 2001]. Land-use changes, river channel modifications, and pollution threaten the continued existence of the winged mapleleaf. Additional threats to the small, remaining populations include expanded agriculture, low water levels, and intense recreational boat traffic. These factors might also affect host fish, which are necessary for completion of the unionid mussel life cycle. In addition, it is possible that many of the remaining winged mapleleaf populations are sufficiently small that their long-term genetic viability is questionable (USFWS 1991).

Direct impacts of zebra mussels on the viability of winged mapleleaf populations remain to be documented. Given the ability of zebra mussels to attach and grow on the shells of native mussels, there appears to be a risk to winged mapleleafs posed by invasive dreissenids. Speculated impacts of excessive zebra mussel colonization on native mussels include (1) impairment of native mussel filter-feeding, (2) increased exposure to parasites and disease, and (3) diminished or potentially lethal water quality conditions [e.g., increased ammonia, decreased dissolved oxygen (DO)].

The Higgins eye pearlymussel

The Higgins eye pearlymussel (*Lampsilis higginsii*, Figure 5) is a freshwater mussel that is found only in the Mississippi River, the St. Croix River in Wisconsin, the Wisconsin River and the Rock River in Illinois (Thiel 1981). Higgins eye pearlymussels prefer larger, deeper rivers (USFWS 1997, Wilcox and Dietz 1995, Davis and Hart 1995). The Higgins eye occurs in low numbers throughout its range and was designated as an endangered species in 1976 (Cawley 1996). The 1993 flood and the infestation of zebra mussel might have posed additional threats to the continued existence of this species (Clarke and Loter 1995).

The reproduction and early life history of the Higgins eye mussel are poorly known. However, similar to other species of freshwater unionid mussels, the Higgins eye has a parasitic larval stage (glochidia) that requires a developmental period attached to the gills or fins of host fish (Gordon 2002; Heath 2001, 2002). Freshwater drum and sauger were thought to be the host species. However, more recent studies indicate that largemouth bass, smallmouth bass, yellow perch and walleye are also suitable host species (Burky 1983).

It is suggested that zebra mussel densities > $0.5/m^2$ could pose a threat to Higgins eye pearlymussel (USFWS 2004). Although zebra mussels are perhaps an important threat to Higgins eye pearlymussel, construction activities, environmental contaminants, and poor water quality also pose threats to the continued viability of this native mussel (USFWS 2004). Additional stressors to *L. higginsii* in the UMRS include impoundment, regulation of flows, dredging, placement of dredged materials, and availability of host fishes. Additionally, Miller and Payne (2007) report that at a secondary channel on the UMR near Prairie du Chien, zebra mussel densities increased steadily after they were first found in 1992. Measurable unionid mortality was noted in 1998, six years after zebra mussels were first collected. In 1999, zebra mussel densities were > $10{,}000/m^2$, and corresponding unionid density declined to less than $2/m^2$. No *L. higginsii* were collected. This mussel bed began to recover by 2005, when zebra mussel density declined to $\sim 250/m^2$. During 2005, *L. higginsii* density and relative abundance increased to near pre-infestation values. These data suggest that *L. higginsii* could be resilient to zebra mussels over the long term (Miller and Payne 2007).

Water quality impacts

The primary impacts of invasive dreissenid mussels on water quality include the depletion of suspended particulate matter from the water column, and perhaps more importantly the creation of zones of increased ammonia and depleted oxygen in layers of accumulated dead and decomposing dreissenid mussels. Though likely localized, these degradations of water quality can reduce habitat available for native mussels. The sheer impact of being covered by several feet of living and dead zebra mussels likely poses a greater risk to native mussels.

In addition, zebra mussels can contribute to the transfer and concentration of toxic contaminants in food chains, by accumulating chemicals in their tissues at levels up to 100,000 times the concentration in the surrounding water (de Kock and Bowner 1993). Waterfowl that consume contaminated zebra mussels have elevated concentrations of metals, organic pesticides and polychlorinated biphenyl compounds (e.g., Bemy et al. 2003). Species that feed on zebra mussels such as round gobies, freshwater drum and ducks such as scaup may be impacted by eating contaminated zebra mussels. Bioaccumulation may further increase the concentration of chemical contaminants in predator species (http://www.ofah.org).

Impacts on infrastructure

Biofouling is the greatest abiotic effect of zebra mussels in newly invaded lakes, reservoirs, streams, navigation channels and locks. In the Great Lakes, *Dreissena* fouling is generally limited to structures submerged below 1.2 m depth (Claudi and Mackie 1994). Permanent marine structures including pilings, bridges and docks are particularly vulnerable to fouling. Navigational markers and fishing buoys can sink as the result of accumulating zebra mussels (Martel 1993). Commercial trap nets and gill nets can also collect sufficient zebra mussels to render the equipment useless or difficult to retrieve. The hulls of boats and ships can become so infested that sailing efficiency is reduced. Zebra mussels colonize industrial, boat and domestic water intake pipes, and reduce or prevent water flow.

Water intake structures for municipal, industrial, and hydroelectric plants are highly vulnerable to fouling. Extraordinarily high zebra mussel densities can be achieved in water intakes because of the large number of potential mussels entrained in the intake current, constant replenishment of food resources, removal of mussel wastes, and the absence of predators. Power plant components that may become fouled include crib structures, trash bars, screen houses, steam condensers, heat exchangers, penstocks, service water systems and water level gauges (Kovalak et al. 1993, Claudi and Mackie 1994). Long and narrow pipelines are particularly vulnerable to fouling and subsequent severely impeded flow (Claudi and Mackie 1994). Kovalak et al. (1993) reported that mussel densities of 750,000 individuals /m^2 at Monroe Power Plant in western Lake Erie. These values far exceeded densities (<5000 individuals/m^2) on adjacent lake bed, and at a nearby (Fermi) nuclear power plant, which utilized ~30 times less water than the Monroe facility (Kovalak et al. 1993).

The economic costs of biofouling by zebra mussels have been substantial (Claudi and Mackie 1994). Earlier estimates were that zebra mussel prevention, control, and monitoring would cost facilities in the Great Lakes region a $2–5 billion dollars by the late 1990's (Office of Technology Assessment 1993). O'Neill (1996) surveyed costs associated with zebra mussel control, prevention, and research in the United States and Canada from 1989–1995. The 339 facilities, ranging from small businesses to large power plants, reported cumulative costs of just over $69 million for the six-year period, with 51 percent of the costs incurred by power plants, 31 percent by water treatment plants, 8 percent by industries, 7 percent by public agencies, and 1 percent by scenic river ways.

A 1995 study indicated that between 1988 and 1995 facilities spent over $69 million on zebra mussel monitoring and control (O'Neill 1996). A paper company spent $1.4 million to remove ~300 cubic meters of zebra mussels from its intake in Lake Michigan (USGS 1997). Ontario spent over $172 million for preventing zebra mussel infestations at 8 hydropower facilities, 86 municipal plants, and 67 industrial plants. In the Great Lakes, each small and large volume water user might annually spend $20,000 and $460,000, respectively for controlling zebra mussels (Indiana DNR 2006).

Risk to locks and dams

Zebra mussel densities continue to increase at locks and dams on the UMR (Mackie 1993). Based on the National Biological Service in Wisconsin, densities at the uppermost portions remain relatively low. Densities range from one per square meter at Lock and Dam 1 in St. Paul, Minnesota, to 11,432 per square meter at Lock and Dam 13 just north of Davenport, Iowa. Most of the Locks and Dams in between had densities greater than 1000/m2. The National Biological Service in Wisconsin states that these numbers have increased since 2005. According to USACE personnel (Tim Yager, currently at USFWS, Upper Mississippi Refuge) zebra mussels were observed to be sparsely attached to lock walls and hard surfaces throughout the lock chamber at Lock 6. Estimated mean density was 7.9 mussels per square meter along the lock floor; 19 per square meter along lock intake ports. One USACE report stated, "considering all surfaces available for attachment in the lock chamber, it can reasonably be estimated that 120,000 zebra mussels existed in the lock chamber are relatively low and currently causing no problems in terms of lock operation." It was also reported, "dewatering during cold weather periods is an effective method of killing zebra mussels within lock chambers." Lock and dam operators on the Mississippi River and water users throughout the region have incurred costs trying to control zebra mussels. However, no specific cost information was provided in the USACE report.

Implications for unionid refugia

Hunter and Bailey (1992) reported that some backwater habitats within the UMR system support diverse unionid faunas but do not support large numbers of zebra mussels. While not completely free of zebra mussels, such habitats might function as refugia for native unionids (Tucker 1994, Hebert et al. 1989, and Mackie 1991). However, winged mapleleaf and Higgins eye pearly mussels are typically found in large channels. While backwaters might not offer refuge from invasive dreissenids, marginal sandy habitats along channel borders characterized by low densities of unionids (and even lower densities of zebra mussels) might serve as refuge areas for winged mapleleaf and Higgins eye pearly mussels, especially when zebra mussels decline.

Existence of such refugia may allow recolonization of habitats where native unionids have been extirpated by zebra mussel colonization. Since zebra mussels can be expected to colonize most habitats in the UMR (Griffiths et al. 1991), identification of possible refugia allows resource managers the opportunity to protect unionid diversity in such backwater habitats. Such refugia might be needed only in a few areas when protection of unique unionid species is the primary objective (Schloesser and Kovalak 1991, Tucker et al. 1993).

Habitats Factors for Zebra Mussel Establishment

The following habitat factors were used as independent variables in the model that estimated the risk of zebra mussel establishment in lakes and rivers within the St. Croix Basin.

Calcium

Growth of mussels will occur at calcium concentrations of greater than 35 milligrams per liter (mg/L), but growth is inhibited at concentrations <4 mg/L (EPRI 1992). Twelve to fifteen mg/L appears as a minimum calcium concentration for reproduction and growth (Cohen and Weinstein 1998a). In laboratory studies, zebra mussels did not survive calcium levels below 15 mg/L (Vinogradov et al. 1993). In tests of rearing success, the number of deformed larvae decreased at >34 mg/L of calcium (McMahon 1996).

The regression for calcium showed negative growth below 8.5 mg/L and maximum growth at 32 mg/L, with the growth rate declining at higher calcium levels. No significant relationship was found between the number of veligers produced and any of the environmental variables, although veligers were only produced in waters with 20 mg/L or more of calcium and pH of at least 8.2 mg/L (Cohen and Weinstein 2001). Zebra mussel's calcium threshold is an indication for its potential distribution in North America.

Total Hardness

Total hardness is defined as the sum of calcium and magnesium concentrations expressed as calcium carbonate (Eaton et al. 1995). There are significant curvilinear relationships between juvenile growth rates and each of the buffer variables (calcium, alkalinity, and total hardness). As total hardness decreased below 25 mg/L, zebra mussels grow poorly; zebra mussels grow well when total hardness exceeds 90 mg/L (Cohen and Weinstein 2001).

Conductivity

Sorba and Williamson (1997) estimated that zebra mussel colonization is potentially determined by was calcium, total hardness, pH, temperature, DO, conductivity and turbidity. Conductivity <22 µS/cm will limit zebra mussel distribution and greater than 83µS/cm will be greatly favor zebra mussel colonization (Cohen and Weinstein 2001.

Dissolved Oxygen

Zebra mussels are among the least tolerant to low oxygen levels of all freshwater bivalves. DO concentrations less than 2–4 mg/L are lethal to zebra mussels and DO greater than 8 mg/L are favorable to zebra mussel growth (Cohen and Weinstein 2001). Oxygen depletion associated with respiration of zebra mussels has been documented in two large rivers, as evidence that DO is critical to zebra mussels’ growth and survival (Effler and Siegfried 1998, Effler et al. 1996, Caraco et al. 2000).

Chlorophyll *a*

Zebra and quagga mussels feed on natural seston. There are no significant differences in per capita clearance rates (CR), functional responses, or feeding behavior between zebra and quagga mussels. Per capita CR could range from 0.018 to 0.402 L/mussel/hr for zebra mussels and from 0.010 to 0.407 L/mussel/hr for quagga mussels.

Zebra mussel growth and body condition were weakly correlated with phytoplankton biomass, which can be represented by the concentration of chlorophyll *a* (Strayer and Malcom 2006). Filtration rate reduced when chlorophyll *a* decreased from 7.4 µg/L to 2.2 µg/L (Fanslow et al. 1995). Zebra mussels' mean filtration rate was 16.2 mL/mg/h (range 4.0 to 40.7 mL/mg/h) over a 2-year period observation. Lower rates could be attributed to higher concentrations of seston (chlorophyll, particulate organic carbon, and total suspended solids). Overall filtration rates were related to seston concentrations as described by a negative exponential function.

Salinity

Another possible limiting factor for zebra mussel establishment is salinity. Quagga mussels are usually found in fresh water in salinities up to 1 percent. These mussels can reproduce in salinities <2–3 PSU, but are killed by salinities >6 PSU (Setzler-Hamilton et al. 1997). Zebra mussel salinity tolerance limits depend not only on salinity levels, but also on the rate of change of salinity and on the composition of the salt. Mussels cannot survive the short-term fluctuations in salinity levels typical of estuaries and some coastal lagoons (Strayer and Smith 1993).

Potassium

Potassium levels of >100 mg/L are lethal to adult zebra mussels (Wildridge et al. 1998) and prevent settlement of veligers (immature mussels) at 50 mg/L (Claudi and Mackie 1994).

Ammonia

Ammonia concentrations were identified as potentially limiting to zebra mussels in lake water (Spada 2000). Zebra mussels reared under laboratory conditions are extremely sensitive to ammonia levels > 1 mgN/L (Nichols 1992). Ammonia is also toxic to zebra mussels at levels of about 2 mg/L (Wildridge et al. 1998).

Current Velocity

Mussel establishment also depends on water current velocity. Flow velocities exceeding 1.5 m/sec minimize mussel settlement in water intakes (Claudi and Mackie 1994). The water intake for the city of Windsor, Ontario, has a flow velocity ~2.5 m/sec and has not been fouled internally by zebra mussels (P. McQuarrie, personal communication). The mussels appear unable to attach where the water velocity exceeds 1.5 m/sec; attached mussels can be washed off at speeds exceeding 2 m/sec.

pH

Zebra mussels have distinct pH-tolerance limits. In the laboratory, a pH range of 7.3 to 9.4 is required for veliger development, and development success is greatest at pH ~8.5 (Sprung 1993). In the field, Ramcharan et al. (1992a, b) found that a pH of 7.3 was the lower limit of zebra mussel occurrence in 76 European lakes (Cohen and Weinstein 2001). Adult zebra mussels are more tolerant of lower pH than are larvae.

Secchi Disk Depth

Turbidity or water clarity, measured as Secchi disk transparency (cm), is inversely related to the concentration of suspended sediments, detritus and plankton (Preisendorfer 1986). As water clarity increases, zebra mussel growth can become limited by food availability. Subsequent zebra mussel population decreases have been reported (Strayer and Malcom 2006). Zebra mussels are not able to survive under the conditions of very high turbidity with corresponding Secchi disk depths <10 cm. Zebra mussels appear to grow rapidly under conditions where Secchi disk depth ranges between 40 cm and 200 cm (Cohen and Weinstein 2001).

Temperature

However, quagga mussels can be observed in water as cool as 8 ºC, zebra mussel most commonly occur at water temperature between 12.5−21.5 ºC. Adults are more adapted to water temperature between 14−28 ºC. Based on Walz (1978) there is no growth in zebra mussels at 4.5−5.5 °C. Juveniles and adults are able to grow over a range of temperatures (12–30 °C). Poorer growth of zebra mussels can result from water temperatures >28 °C. Zebra mussels do not survive at temperatures greater than 32 °C (Claudi and Mackie 1994, Ohio Sea Grant 1994). Populations have become abundant in the southern United States where temperatures often reach temperatures of 30 °C; however, massive die-offs have been observed at 31 °C (Cohen and Weinstein 1998b).

Spawning Temperature

Adults require a threshold temperature to initiate spawning. In North America, zebra mussels normally begin to spawn at 12 °C and above, though limited spawning has been reported at 10 °C in the Great Lakes and Europe (Nichols 1996). Spawning peaks at about 12–18 °C, which is also roughly the optimum temperature for larval development (Sprung 1993).

Water Depth

Based on Pligin's (unpublished data) observation in Kremenchug and Kakhovka Reservoirs from 1985 to 1992, zebra mussel and quagga mussel were observed in different water depths. It was found that mussels appeared most abundance in water depth of 4–12 m and rare in water depth of greater than 16 m, which may be related to a number of factors including lower DO, food availability, pH, etc. Shallow water limitations in zebra mussel habitat likely reflect aerial exposure under fluctuating water levels or impacts of waves and ice formation.

Habitat Suitability and Risk of Establishment

Habitat Factors

Fourteen physical-chemical factors that determine habitat suitability were identified for the dreissenid mussels. Functional relationships between values of these factors and habitat suitability were derived from the technical literature. The functional relationships are described by three basic formulations: (1) habitat suitability increases with increasing values of the environmental factor; (2) habitat suitability decreases with increasing values of the factor; and (3) habitat suitability is optimal for a defined range of the factor: factors too low or too high describe less than optimal habitat. Calcium is necessary for shell formation and Figure 6 illustrates the relationship between calcium concentration and habitat suitability for this environmental factor. The threshold value of 17 mg/L defines a concentration below which zebra mussel growth rate shifts from positive to negative. Concentrations >32 mg/L appear optimal to support zebra mussel growth. Other environmental factors included in the habitat analyses that have a similar functional relationship are total hardness, conductivity, DO, and chlorophyll *a*.

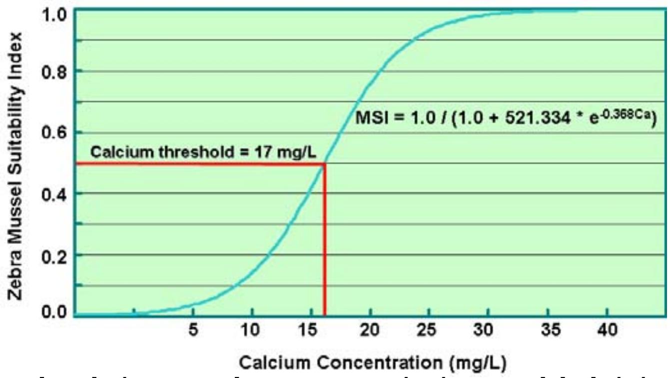

Figure 6. Relationship between calcium concentration and habitat suitability for zebra mussels.

In contrast to calcium, excessive concentrations of ammonia are toxic to zebra mussels (Figure 7). Habitat quality reduces from near optimal at concentrations <0.5 to 1.0 mg/L, where growth rate becomes negatively impacted. The functional form used to describe habitat suitability in relation to ammonia concentrations is also used for salinity, potassium, and water current velocity.

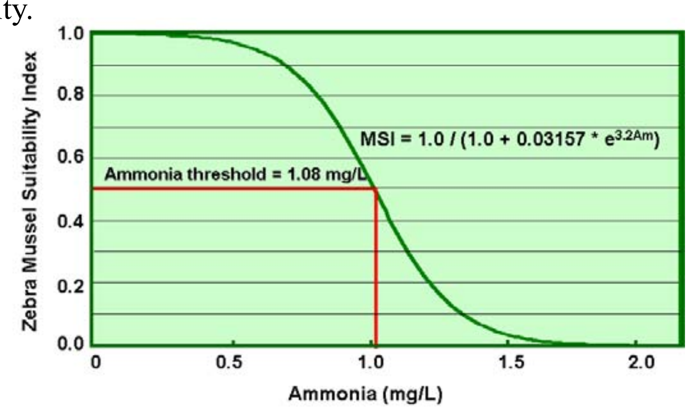

Figure 7. Relationship between ammonia concentration and habitat suitability for zebra mussels.

Figure 8 illustrates the relationship between water temperature and zebra mussel habitat suitability. Temperatures <14 or >28 °C correspond to negative growth rates for zebra mussels. Zebra mussels express optimal growth at a temperature of ~22 °C. Other habitat factors for the zebra mussel described by a critical range of values are pH, Secchi disc depth, spawning temperature, and water depth.

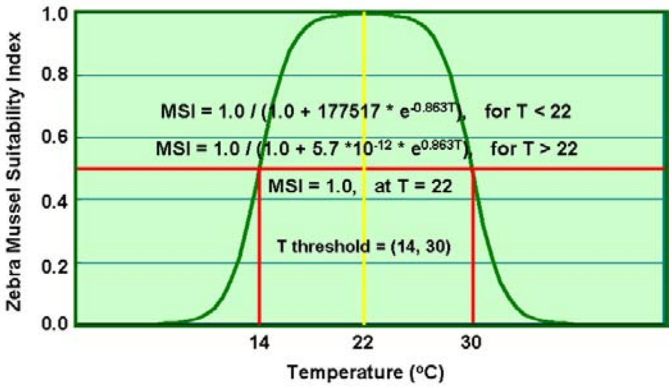

Figure 8. Relationship between water temperature and habitat suitability for zebra mussels.

The equations that describe the habitat suitability functions for all 14 parameters used in the zebra mussel model are presented in Table 1. These functions are used to translate data for specific water bodies in the St. Croix Basin to the Habitat Suitability Index (HSI) values used in the assessment of zebra mussel establishment.

Table 1. Zebra mussel habitat SI, threshold values of fourteen environmental variables.

Environmental Variables		Units	Equations	Threshold Values
(1)	Calcium	mg/L	$SI=1.0/(1.0 + 521.334 * e^{-0.368Ca})$	17 mg/L
(2)	Total Hardness	mg/L	$SI=1.0/(1.0 + 182.967 * e^{-0.0906TH})$	57.5 mg/L
(3)	Conductivity	μS/cm	$SI=1.0/(1.0 + 158.89 * e^{-0.097Co})$	62.5 μS/cm
(4)	DO	mg/L	$SI=1.0/(1.0 + 6858 * e^{-1.472DO})$	6 mg/L
(5)	Chlorophyll *a*	μg/L	$SI=1.0/(1.0 + 19.6754 * e^{-0.69954Ch})$	4.25 μg/L
(6)	Salinity	mg/L	$SI=1.0/(1.0 + 0.001 * e^{0.9585Sa})$	7 mg/L
(7)	Potassium	mg/L	$SI=1.0/(1.0 + 0.00069 * e^{0.1418K})$	50 mg/L
(8)	Ammonia	mg/L	$SI=1.0/(1.0 + 0.03157 * e^{3.2Am})$	1.08 mg/L
(9)	Current Velocity	m/s	$SI=1.0/(1.0 + 0.001 * e^{4.6045CV})$	1.5 m/s
(10)	pH		$SI=1.0/(1.0 + 3.121 *10^{39} * e^{-11.51pH})$ as $pH < 8.5$, $SI=1.0/(1.0 + 3.211 * 10^{-46} * e^{11.51pH})$ as $pH > 8.5$, $SI=1.0$ as $pH = 8.5$	(7.8, 9.2)
(11)	Secchi Disk Depth	Cm	$SI=1.0/(1.0 + 40.535 * e^{-0.07578SDD})$ as $T < 140$, $SI=1.0/(1.0 + 9.09 *10^{-9} * e^{0.07578SDD})$ as $T > 140$, $SI=1.0$ as $T = 140$	(75, 205)
(12)	Temperature	°C	$SI=1.0/(1.0 + 177517 * e^{-0.863T})$ as $T < 22$, $SI=1.0/(1.0 + 5.7 *10^{-12} * e^{0.863T})$ as $T > 22$, $SI=1.0$ as $T = 22$	(14, 30)
(13)	Temperature (Spawning)	°C	$SI=1.0/(1.0 + 120686.3 * e^{-1.90457T})$ as $T < 17$, $SI=1.0/(1.0 + 8.3 *10^{-12} * e^{1.90457T})$ as $T > 17$, $SI=1.0$ as $T = 17$	(12.5, 21.5)
(14)	Water Depth	M	$SI=1.0/(1.0 + 99 * e^{-1.6431WD})$ as $T < 7$, $SI=1.0/(1.0 + 6.227 * 10^{-9} * e^{1.7125WD})$ as $T > 7$, $SI = 1.0$ as $T = 7$	(3, 11)

Habitat Data

The previously described habitat factors were used to develop zebra mussel suitability index (SI) models for each factor, as well as composite HSI models. Data for each of these factors were obtained for surface waters located within the St. Croix River Basin, including Navigation Pools 2−4 on the UMR (Figure 9).

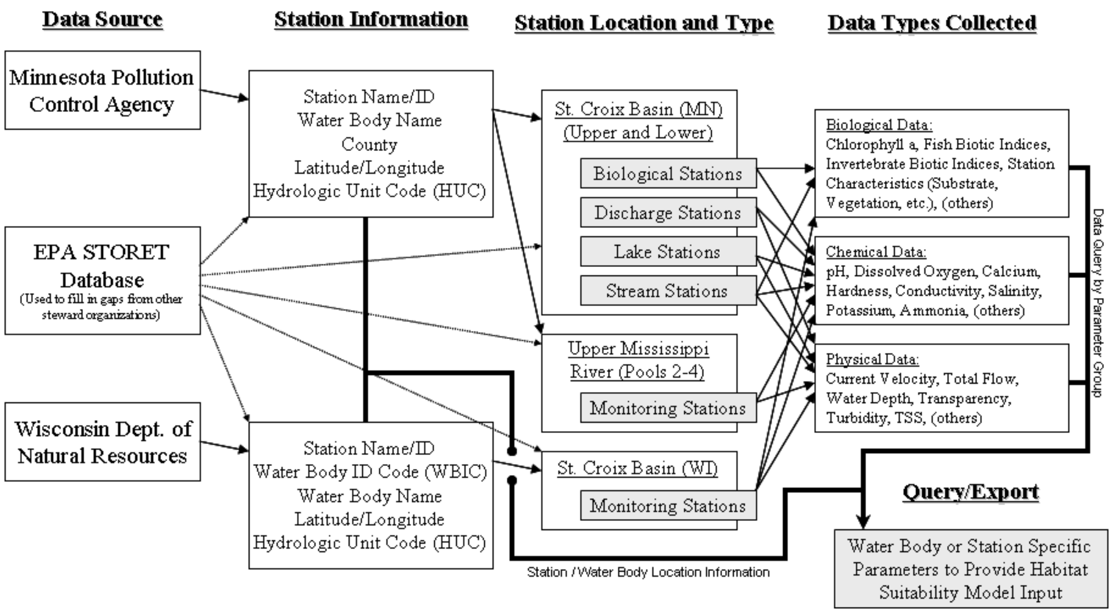

Figure 9. Schematic illustration of data collection and data management for the risk-based decision support system for zebra mussel establishment in the St. Croix River Basin.

The Minnesota Pollution Control Agency, EPA STORET, and Wisconsin Department of Natural Resources were the primary sources of habitat data. The data collated from each source included station information, location, and type (Figure 9). Monitoring stations were identified for the Wisconsin portion of the St. Croix Basin and Pools 2−4 on the UMR. Data identified for the upper and lower regions within the St. Croix Basin included biological, discharge, lake, and stream stations. Depending on the location, each station provided physical, chemical, and/or biological data relevant to the mussel habitat suitability model. Data collated from these sources were used to construct a database using Microsoft Access. This database was subsequently queried multiple times to obtain values of individual habitat factors for each sample location. Results of these queries were stored in spreadsheet format compatible with the data needs of the habitat model.

Evaluation of Data Completeness

Following compilation into the project data base, the data were evaluated to determine (1) temporal coverage, (2) number of parameters with useful data per sampling station, and (3) usefulness of the data without additional transformation. A ranking scheme was developed to evaluate each station/location record in relation to these three aspects of data completeness. Each station was assigned a value from 1−10 according to its temporal coverage. A value of 10 was assigned to stations that had monthly samples for the period 1996−2006. Other values were assigned as

- 8−9: sampled monthly for seven of the years 1996−2006 or multiple months for all ten years;
- 5−7: sampled monthly for five of the ten years, or multiple months for all ten years;
- 3−4: sampled monthly for two of last ten years, or multiple months for 3–4 of the ten years; and
- 1−2: sampled once within last 10 years, or sampled multiple times, but not within 1996−2006.

Stations were assigned values from 1−10 based on the number of habitat suitability model parameters (total of 13) for which data were available. Values were assigned as

- 10: each temporal sample has data for all 13 model parameters,
- 7−9: each sample has data for 10−13 of the model parameters,
- 6: each sample has data for 8−9 parameters,
- 5: each sample has data for 6−7 parameters,
- 4: each sample has data for 4−5 parameters,
- 3: each sample has data for 2−3 parameters,
- 2: each sample has data for one parameter, and
- 1: the data have none of the necessary model parameters.

Finally, each station was scored a value of 2 if the data could be used directly without additional transformation (e.g., unit conversion). A value of one was assigned if the data required some transformation prior to use by the habitat model.

Using the above scheme, the scores for temporal coverage (Tc), parameter completeness (Pc), and data conversion (Dc) were used to define a data completeness index (DCI) for each station according to

$$DCI = [(Tc + Pc + Dc)/22] \cdot 100$$

where, the minimum DCI = $(1+1+1)/22 \cdot 100 = 13.6$ and the maximum = 100.

Figure 10 illustrates the distribution of DCI values for 325 Minnesota and Wisconsin stations included in the analysis. The DCI values ranged from 13.6 to 77.3. The average value was 35.6; the median was 31.8. These results underscore the fact that the original data were not collected in anticipation of contributing to the assessment of habitat quality for zebra mussels.

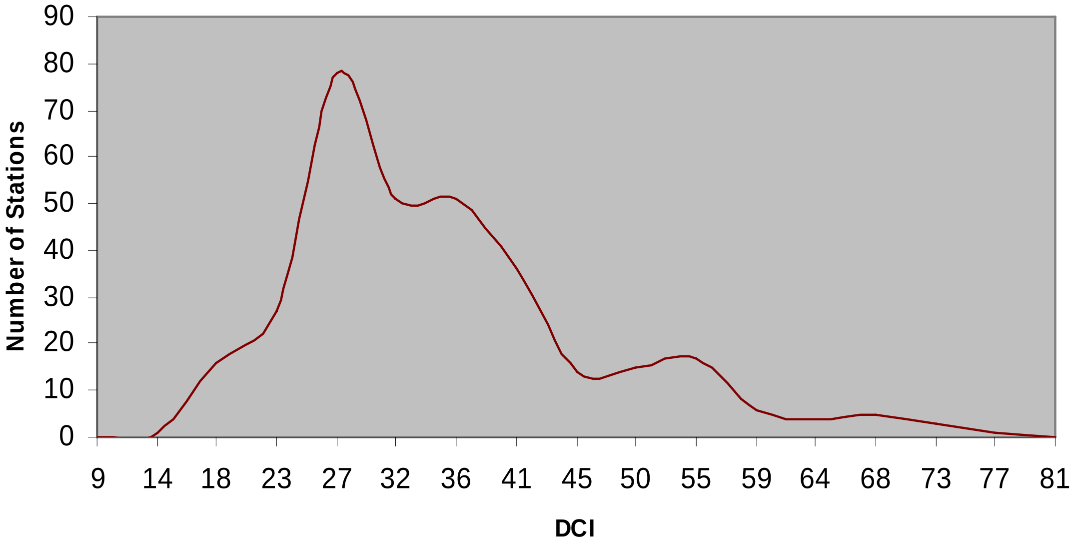

Figure 10. Distribution of DCI values for 325 Minnesota and Wisconsin sampling stations in the St. Croix Basin.

Table 2 summarizes these results for the separate sources of data used in developing the overall project data base. Perhaps the most useful data are for the Minnesota lakes and streams, followed by the Minnesota Rural Partners (MRP) 2–4, and Wisconsin Department of Natural Resources (WDNR) data. The data for Minnesota bio-stations and discharge monitoring stations were of the lowest quality used in this study. On average, none of the data sources achieves 50 percent of the possible data quality score. Thus, while it is possible to perform the model calculations (see preliminary results sections); the interpretation of the model results must be tempered with an understanding of the quality of the underlying data.

Table 2. DCI index values for different data groups used to develop the project data base.

Data Source	Station Type	Number of Stations	DCI		
			Average	Minimum	Maximum
Minnesota Pollution Control Agency (MPCA)	Biostations	59	27.1	18.2	36.4
	Discharges	23	27.7	27.3	31.8
	Streams	57	36.4	22.7	63.4
	Lakes	83	44.7	13.6	77.3
MRP	MRP 2–4	7	37.7	18.2	45.5
WDNR	WDNR	96	33.9	18.2	54.6

Sample data were converted into daily, monthly, and annual averages for use in correspondingly scaled versions of the zebra mussel habitat model. In many instances, daily values of these habitat data were linearly interpolated from weekly or bi-weekly samples. Appendix 1 lists examples of habitat data used in assessing the risk of zebra mussel establishment in surface waters of the St. Croix River Basin.

Correlations among Habitat Factors

The available habitat factor data were analyzed to determine if there were correlations among any of the parameters. Such correlations could be used to (1) reduce the dimensionality of the habitat suitability component of the risk-based decision support model, and (2) identify the most important data gaps to guide future data collection in the Basin. The following results focus on the water quality parameters and do not include chlorophyll *a*, water depth or current velocity. Analyses were performed separately for data collected in Minnesota (i.e., MPCA) (Table 3) and Wisconsin (i.e., WDNR) (Table 4).

Table 3. Correlations Among Habitat Factors in Minnesota Surface Waters in the St. Croix Basin.

	Calcium	**Chl *a***	**Dissolved Oxygen**	**Hardness**	**pH**	**Potassium**	**Secchi disc**	**Specific Conductivity**	**Water Temp**
Calcium	1.00								
Chl a	-0.24	1.00							
DO_2	0.18	0.11	1.00						
Hardness	0.89	-0.21	0.15	1.00					
pH	0.78	0.29	0.14	0.66	1.00				
Potassium	0.42	-0.29	0.12	0.38	0.19	1.00			
Secchi disc	0.29	-0.49	-0.05	0.47	-0.21	0.21	1.00		
Specific Conductivity	0.95	0.05	0.003	0.56	0.29	0.42	-0.004	1.00	
Water Temp	-0.10	0.18	-0.04	-0.10	0.32	-0.12	-0.27	-0.06	1.00

The results for the Minnesota surface water data show that calcium, hardness, and specific conductivity are highly (0.89 or greater) correlated in these systems. Values of pH are correlated with calcium (0.78) and hardness (0.66). Thus, including all of these factors in the assessment of habitat quality introduces some redundancy. It might prove possible to use these correlations to reduce the number of factors included in the assessment (e.g., Ramcharan et al. 1992a). However, selection of a single factor (e.g., calcium) is made difficult because not all water bodies in the data set have calcium data, or hardness and specific conductivity data. Correlations among the remaining water quality parameters are low. This degree of independence suggests that each of these parameters should be included in the habitat suitability calculations. Future data collection should include all of these parameters wherever possible. Given the sparse nature of these data and the comparatively simple calculation of habitat suitability, the model should continue to make use of as many factors as possible and use all the data available for any given water body. If future data collection provides even coverage for calcium, hardness, and specific conductivity data for the Minnesota water bodies, it might prove feasible to remove hardness and conductivity from the habitat calculations and simply use calcium concentration (i.e., Ramcharan et al. 1992a).

Table 4. Correlations Among Habitat Factors in Wisconsin Surface Waters in the St. Croix Basin.

	Calcium	Chl *a*	Dissolved Oxygen	Hardness	pH	Ammonia	Secchi disc	Specific Conductivity	Water Temp
Calcium	1.00								
Chl *a*	0.18	1.00							
Diss. O_2	0.35	-0.07	1.00						
Hardness	0.99	0.90	-	1.00					
pH	0.37	0.15	0.38	0.47	1.00				
Ammonia	-0.32	0.09	-0.56	-0.34	0.11	1.00			
Secchi disc	-0.06	-0.54	0.18		0.11	-0.50	1.00		
Specific Conductivity	0.94	0.11	0.12	0.99	0.53	0.19	-0.04	1.00	
Water Temp	-	0.07	-0.07	-	-0.12	-0.37	-0.38	-0.25	1.00

Analysis of the water quality data available for Wisconsin surface waters (Table 4) within the St. Croix Basin reinforced the results obtained for the Minnesota water bodies. Calcium, hardness, and specific conductivity are highly (>0.90) correlated among these water bodies. Correlations of pH with calcium and hardness are lower than in the Minnesota data (Table 3). Otherwise, correlations among the remaining factors in the Wisconsin water bodies are similarly low as in the Minnesota data. Again, future data collections might permit the omission of hardness and conductivity data and simply focus on calcium. Otherwise, the data suggest that all parameters should be included in the model, where data permit.

Risk-based Invasive Mussel Establishment Model

The central purpose of this project was to develop a methodology for (1) assessing the likelihood that zebra or quagga mussels will establish populations in previously non-infested surface waters in the St. Croix Basin, and (2) evaluating the effectiveness of alternative technologies in controlling or slowing the rate of invasive mussel spread throughout the basin. Figure 11a illustrates a contributing factors or influence diagram that identifies the aspects of habitat, location, and inoculation that affect the likelihood of invasive mussel establishment. The model also identifies the consequences (ecological, infrastructure, and economics) of mussel establishment that would be used in evaluating the effectiveness of alternative methods for controlling the rate of dreissenid invasions throughout the St. Croix Basin.

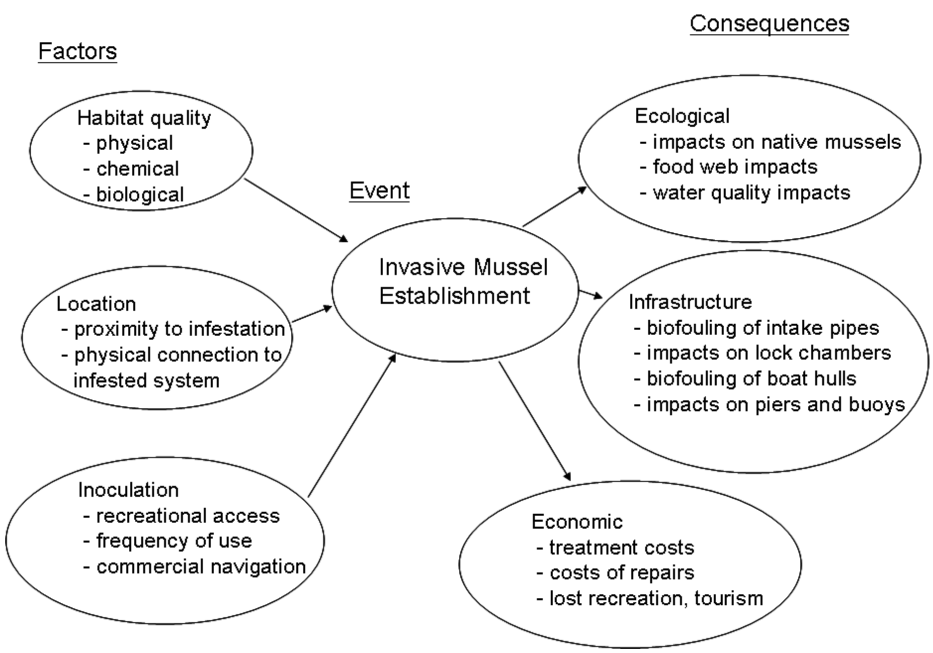

Figure 11a. Contributing factors diagram for assessing invasive mussel establishment in the St. Croix Basin.

Figure 11b presents a conceptual risk-based model derived from the influence diagram that has been designed to address the establishment of invasive mussels. As previously defined, establishment is the development of a persistent population of invasive mussels in a previously uninfested water body. According to the model, the risk of mussel establishment in a non-infested lake, stream, or river is a function of (1) habitat quality; (2) location (i.e., geographic or functional proximity to infested waters); and (3) inoculation, which addresses available modes of mussel introduction and rates of introduction. Fundamental to implementation of the model is a database developed for the St. Croix Basin and UMR Pools 2−4 to support the risk assessment.

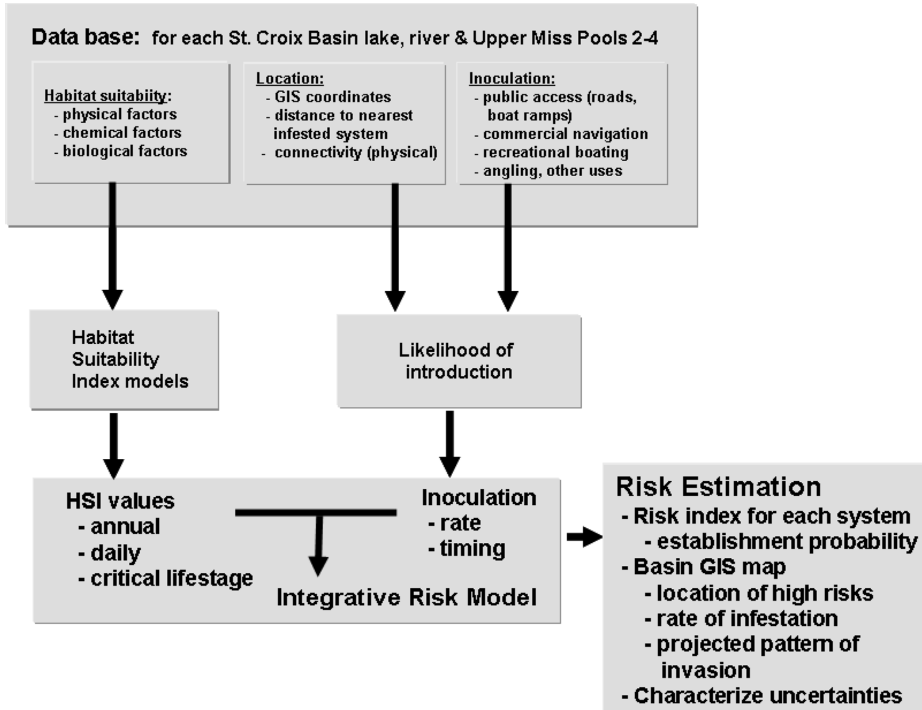

Figure 11b. Risk-based decision model to assess invasive mussel establishment in the St. Croix Basin.

Habitat

Overall habitat suitability for the invasive mussels is determined using the previously defined SI functions for the 14 physical-chemical parameters deemed important in terms of dreissenid biology and ecology. Using data obtained from the previously identified state and federal sources, SI values were calculated for selected surface waters in the basin. Importantly, the SI values can be calculated on an annual basis, daily basis, or for a critical period in the life history of these mussels. For an individual water body, the corresponding overall habitat HSI models can be calculated using (1) the minimum SI among all 14 individual habitat factors, or (2) the geometric mean value of these factors. The minimum value HSI reflects an underlying Liebig Law of the Minimum concept of habitat suitability (Odum 1971); wherein, the lowest value among the calculated habitat factors defines the overall HSI. If all but one of the factors have component values = 1.0, the HSI will nevertheless be determined by the score for the remaining factor, which could be zero.

The geometric mean HSI model connotes a more balanced, multivariate concept of habitat suitability. The geometric mean model also conveniently permits assigning weights or importance values to the individual habitat suitability factors. In the initial analyses reported in this study, both HSI models were used to describe potential dreissenid habitat. It was assumed that the results of these two alternative descriptions would at least bracket the actual zebra mussel habitat suitability for the water bodies of concern.

The habitat component of the risk-based decision support model has been implemented using water quality data collated for more than 70 individual water bodies in the St. Croix Basin.

Location

Location is the second component of the risk-based decision model. The location component refers to the distance from a non-infested water body to the closest infested system. The assumption is that the likelihood of introduction to a non-infested system is greater if the non-infested system is proximate to a continuing source of invasive mussels (e.g., Lake Pepin) (Stoeckel et al. 2004). Importantly, location can refer to geographical distance or functional distance–that is, if the non-infested water body is physically connected to a lake, stream, or river that harbors dreissenid mussels, the functional likelihood of invasive mussel introduction might be greater than that suggested by simple geographic distance. That is, if an upstream water body has an established population of invasive mussels, all the physically connected downstream water bodies are at increased risk of invasion and establishment. Geographic locations for water bodies of interest in the St. Croix Basin can be defined by their GIS coordinates and included in the project data base. Direct physical connections among water bodies in the Basin are more difficult to characterize and have not yet been included in the initial model implementation.

Inoculation

Inoculation refers to the modes and rate of mussel introduction to a non-infested water body. Introductions can result from recreational boating and commercial navigation, as well as angling (i.e., veligers in bait buckets). Convenient access via public boat launches and proximity to highways might increase the rate of invasive mussel introductions to non-infested surface waters within the St. Croix Basin. The inoculation component of the risk-based decision model had not yet been implemented. However, data describing public access and use rates (e.g., launch permits, fishing pressure) for non-infested water bodies might be obtained from state agencies and used to estimate inoculation, even if the estimates are more qualitative in nature. For example, the Wisconsin Department of Natural Resources conducted a boating survey in 1989–1990 to determine the relative rates of usage of individual water bodies on a county-by-county basis, including samples from adjacent counties in Illinois, Iowa, and Minnesota (Penaloza 1991). Padilla et al. (1996) used these results to forecast the spread of zebra mussels throughout selected Wisconsin lakes (mainly in southeastern Wisconsin) in relation to boater use and proximity to Lakes Michigan and Superior.

Methods borrowed from transportation theory have been adapted to forecast the spread of zebra mussels. The adaptations of transportation models emphasize the importance of recreational boating as a key mechanism for spread (Johnson and Carlton 1996). Bossenbroek et al. (2001) developed and evaluated a production-constrained gravity model that predicts colonization of inland lakes by zebra mussels in Wisconsin and Michigan as a function of recreational boating use, lake surface area, and habitat quality (i.e., pH, Ca concentration). Schneider et al. (1998) developed a production-attraction-constrained gravity model that similarly used recreational boating traffic to assess the spread of zebra mussels in Illinois surface waters. To implement the inoculation component of the risk-based decision model, a gravity model might be developed to characterize the role of recreational boating in introducing zebra mussels into uninfested surface waters within the Basin.

Using methods borrowed from machine learning, Drake and Bossenbroek (2004) applied a genetic algorithm for rule-set production (GARP) to predict the potential distribution of zebra mussels throughout the United States. Their resulting models were based on subsets of ten gross physical, climatic, and hydrologic factors for which data were available for the entire United States. Their modeling approach might be adapted to finer scale resolution within the St. Croix Basin by developing a corresponding data set, including the habitat factors already compiled. This approach would provide an alternative to the calculation of HSI values, or at least an independent evaluation of the vulnerability of Basin surface waters to zebra mussel establishment. However, application of the GARP methods to the Basin might be made difficult by the low number of infested systems. Expansion of the methods to include known infested surface waters outside the Basin might provide the necessary data set for training the GARP methods.

Integration

The integration of the three model components will be used to characterize risks of invasive mussel establishment for surface waters within the St. Croix Basin. The method for integrating results from the three model components into a comprehensive risk of establishment remains to be formalized. The potential methods for this integration range from a simple ranking scheme using the combined (i.e., summed) results of the individual model component scores to more complex analyses based on decision trees (e.g., Payne and Miller 2004), genetic algorithms (e.g., Drake and Bossenbroek 2004), or neural networks. The mathematical rigor of the selected integration method will be largely influenced by the kind of information produced by the location and inoculation components of the model.

Risk of establishment can be assessed in relation to habitat quality estimated for individual water bodies. In addition, spatial-temporal patterns of reported and estimated infestation can be used to infer rates and directions of dreissenid spread throughout the Basin. The overall model construct will permit the incorporation of uncertainty associated with the habitat, functional location, and inoculation components of the model. Corresponding sensitivity analyses can be used to (1) assess establishment risk in more probabilistic terms consistent with the definition of ecological risk (Bartell et al. 1992), and (2) determine the value of new data in reducing risk and improving model results used to evaluate management actions.

Preliminary Results for Habitat Suitability

The following section presents selected results from the preliminary assessment of dreissenid mussel habitat suitability in the St. Croix Basin. These results are intended to (1) indicate the potential for zebra and quagga mussels to establish populations in currently non-infested basin water bodies, and (2) highlight important issues associated with the overall modeling approach. Figure 12 presents the results for selected water bodies in the basin obtained using the minimum of the SI factors to define overall habitat quality. These results suggest that Cedar Lake in St. Croix County provides high quality dreissenid habitat and this lake might be a good candidate for invasive mussel establishment if mussels are introduced to this system.

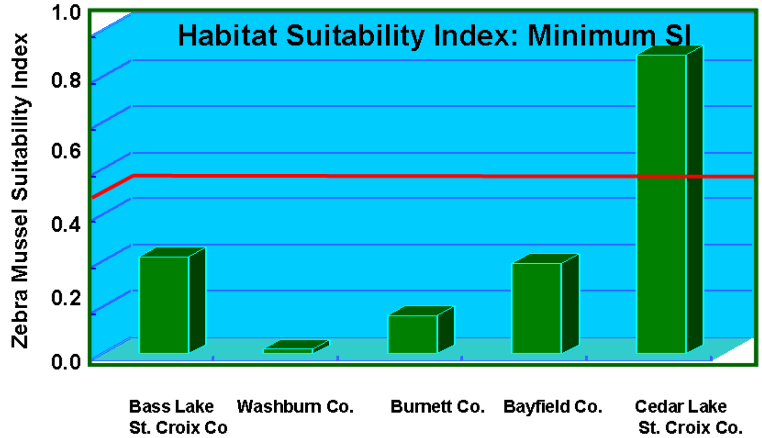

Figure 12. Habitat suitability for selected water bodies in the St. Croix Basin based on the minimum value of the individual component SI values.

As previously mentioned, the geometric mean value of all individual SI values produces a higher overall value of the HSI. Figure 13 presents the results using the geometric mean HSI approach for the same water bodies in Figure 12. According to these results, all five-example water bodies exhibit HSI values > 0.7 and suggest reasonable habitats for invasive dreissenids. Importantly, the use of the geometric mean values can even change the rank order of habitat quality among these systems. For example, compare the sample lakes from Burnett and Bayfield counties in Figures 12 and 13.

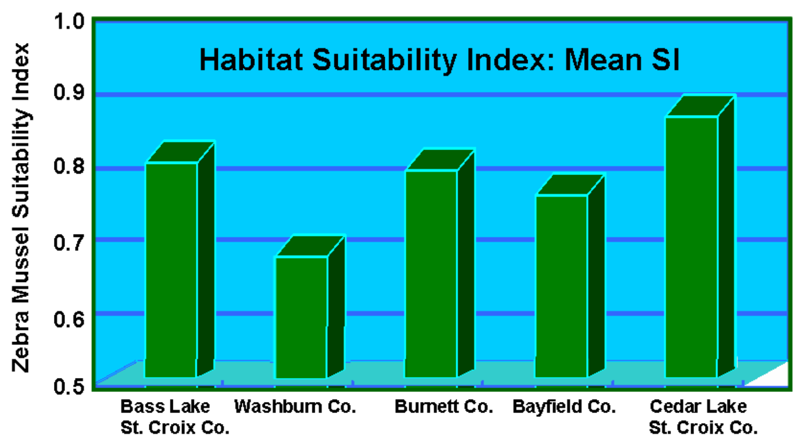

Figure 13. Habitat suitability for selected water bodies in the St. Croix Basin based on the geometric mean value of the individual component SI values.

The habitat SI was calculated for 70 lakes in the St. Croix Basin using the minimum SI value (Figure 14). The model results indicate that ~75 percent of the lakes exceed an HSI value of 0.5. Nineteen of the systems exceed an HSI value of 0.8. I do not think the geometric mean approach is as realistic as the limiting factor (minimum SI) approach.

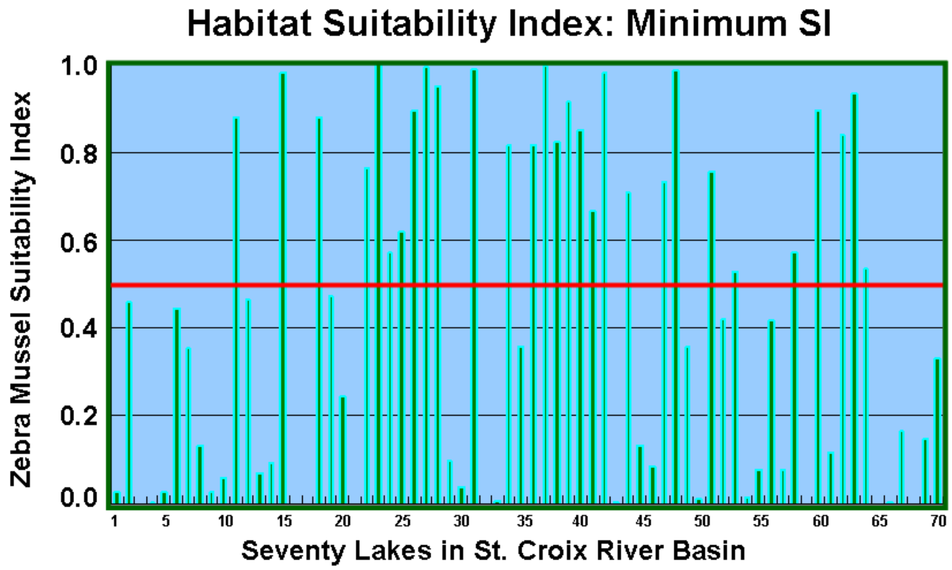

Figure 14. HSI values for 70 selected lakes in the St. Croix Basin calculated using the minimum SI value.

In comparison, all but four of the analyzed water bodies exceed the 0.5 HSI value when calculated using the geometric mean of the individual SI values (Figure 15). Sixty-six percent (46/70) of the analyzed systems have HSI values that exceed 0.8.

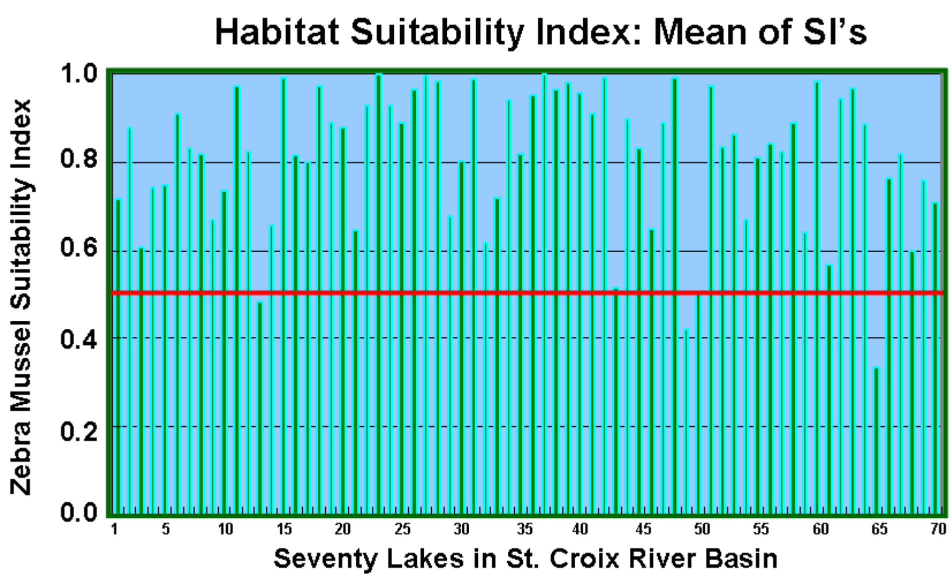

Figure 15. HSI values for 70 selected lakes in the St. Croix Basin calculated using the geometric mean of the SI values.

The preliminary results demonstrate the feasibility of assessing habitat quality for invasive dreissenids in the St. Croix Basin. The results also underscore the implications concerning model structure and estimates of mussel habitat suitability. The minimum SI represents a conservative (pessimistic) characterization of habitat quality; this model structure might underestimate the number of systems potentially at risk to mussel establishment. In contrast, the HSI values calculated using the geometric mean of the individual SI values might overestimate mussel habitat quality and correspondingly, exaggerate the number of water bodies amenable to dreissenid establishment. These initial results emphasize the need to decide on a model structure to include in the risk-based decision model. One alternative might be to use both HSI modeling approaches in an attempt to "bracket" the actual habitat quality characteristic of the water body of interest. Regardless of model structure, the underlying hypothesis is that zebra mussel HSI values are positively correlated with the likelihood of the invasive mussels achieving a self-sustaining population.

The habitat modeling approach permits a more detailed analysis of the individual habitat factors that determine overall dreissenid habitat suitability (Figure 16). For example, the results calculated for the individual SI show that DO may be a factor that would reduce the establishment of mussels in Bass Lake, St. Croix County. The results also illustrate the potential difference between the minimum and geometric mean calculations of the HSI. Using the minimum SI, a HSI of ~0.2 would result; the result using the geometric mean value would be substantially greater, given that the rest of the SI values exceed 0.6.

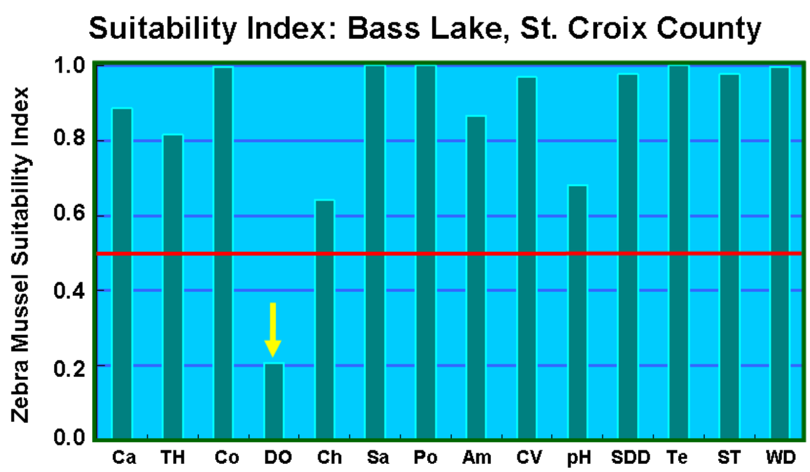

Figure 16. Annual average values of individual SI values for Bass Lake, St. Croix County. Ca =calcium, TH =total hardness, Co=conductivity, DO=dissolved oxygen, Ch =chlorophyll *a*, Sa =salinity, Po =potassium, Am=ammonia, CV=current velocity, pH, SDD=Secchi disc depth, Te=temperature, ST=spawning temperature, WD=water depth.

Figure 17 shows the corresponding results for Cedar Lake, St. Croix County. The individual SI values calculated for this water body suggest that Cedar Lake would be a good candidate for dreissenid mussel establishment. In this instance, either the minimum SI or geometric mean of all SI values would similarly identify this lake as a high quality habitat for mussel establishment.

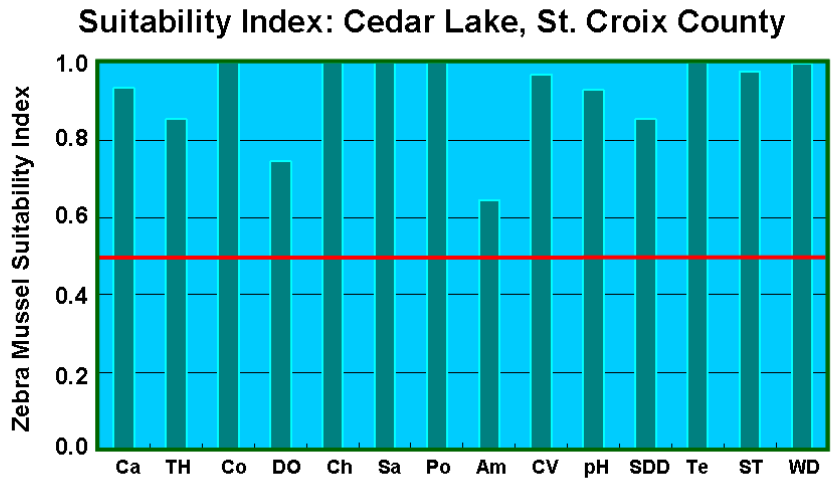

Figure 17. Annual average values of individual SI values for Cedar Lake, St. Croix County.

The previous results for Cedar Lake were based on annual average values of the 14 water quality parameters that define zebra mussel habitat suitability in the decision model. Figure 18 illustrates daily SI values based on DO concentrations interpolated from the existing data. The daily time scale permits evaluation of mussel habitat during critical time periods that could importantly influence establishment success in previously non-infested water bodies. In this example, the over winter DO concentrations in Bass Lake in St. Croix County produced SI values <0.2 for the first six months of the year. Mussels introduced during this time might have less chance of establishing a persistent population than mussels introduced during the later summer and fall, when DO concentrations appear more favorable. Appendix 2 lists results for other surface waters in the St Croix River Basin.

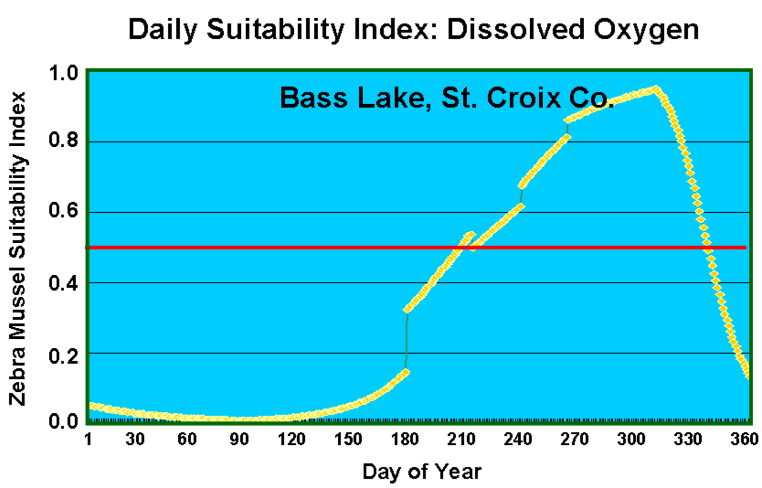

Figure 18. Daily SI values based on interpolated values of DO for Bass Lake, St. Croix County.

Similar analyses were performed using interpolated daily values of chlorophyll *a* for Bass Lake. This parameter correlates food availability with the invasive mussel SI. As with DO, the latter summer time period suggests ample food supply that might facilitate establishment of zebra mussels introduced to Bass Lake during this period. The important point is that the structure of the risk-based decision model permits this kind of detailed analysis of individual habitat factors; either on an annual average or daily time scale, depending on data availability.

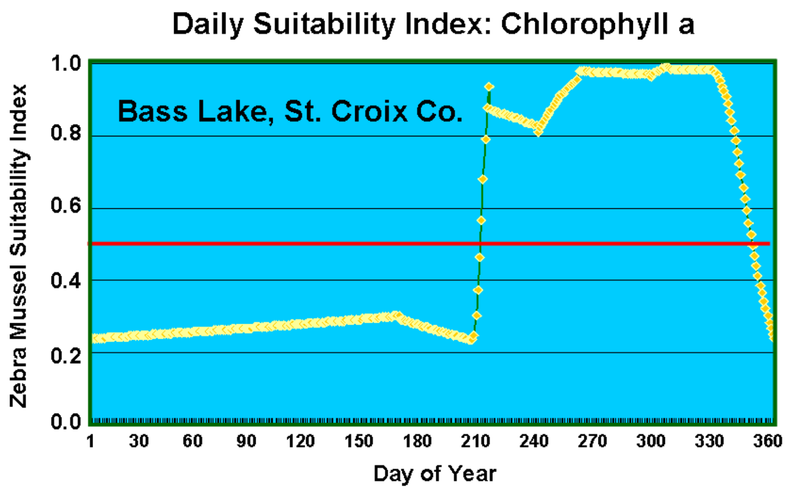

Figure 19. Daily SI values based on interpolated values of chlorophyll *a* for Bass Lake, St. Croix County.

Daily values of the overall HSI for invasive mussels can be correspondingly calculated from the daily SI values (Figures 20 and 21). These HSI estimates can be used to develop a more comprehensive description of the fine scale habitat quality for selected water bodies within the basin. As previously observed for the annual average results, the daily HSI values determined by the minimum SI (Figure 20) or the geometric mean SI (Figure 21) produce different characterizations of invasive mussel habitat quality. Figure 20 illustrates the day-to-day fluctuations in habitat suitability that result from different factors, each with their own daily fluctuations in SI, controlling the overall HSI. The geometric mean approach to calculation of HSI reduces these daily fluctuations and produces daily values that exhibit the same seasonal pattern, but with much less variability (Figure 21). Regardless of the magnitudes and fluctuations, both methods identify the seasons with the highest (and lowest) values of habitat quality. Information available at this refined temporal scale might help identify effective mussel management actions or define time periods for application to achieve the greatest reduction in risk of mussel establishment.

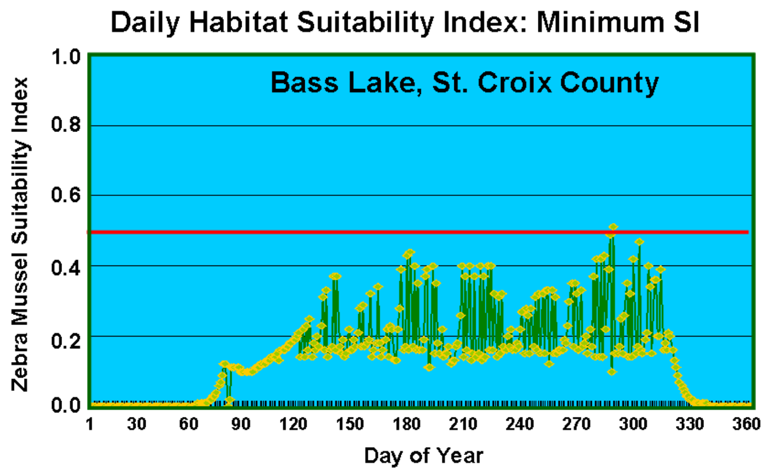

Figure 20. Daily HSI values calculated using the minimum of SI values for Bass Lake, St. Croix County.

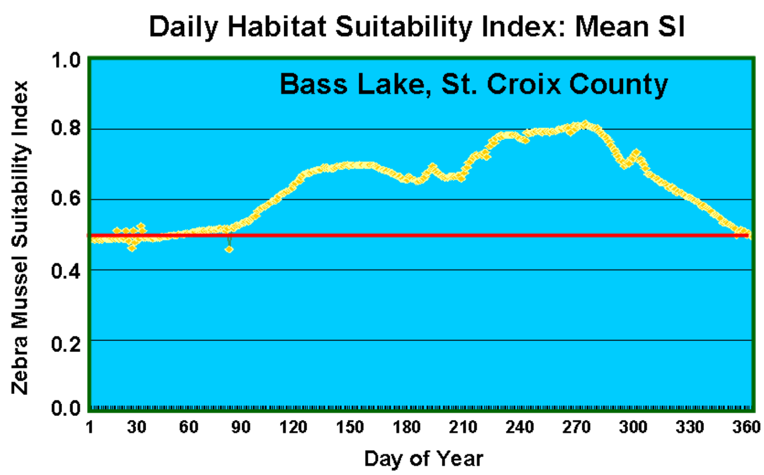

Figure 21. Daily HSI values calculated using the geometric mean of SI values for Bass Lake, St. Croix County.

Risk of Establishment

The above-described approaches were used to assess the risk of zebra mussel establishment in 70 selected surface waters primarily within the St. Croix Basin. Other Wisconsin water bodies were also assessed for potential infestations from Lake Michigan sources. Annual average HSI values were calculated from existing data using both the minimum SI and geometric mean SI for each water body. In addition, the results from the HSI calculations were used to classify the systems as low, medium, or high in relation to habitat suitability and risk of zebra mussel establishment. As an initial hypothesis concerning the integration of both HSI approaches, an initial classification was produced according to the following prescription:

Low Risk: Minimum HSI < 0.5 and Mean HSI < 0.8

Medium Risk: Minimum HSI < 0.5 and Mean HSI > 0.8

High Risk: Minimum HSI > 0.5 and Mean HSI > 0.8

Table 5 lists the results. According to this classification, 24 lakes were assessed as low risk lakes, 17 as medium risk lakes, and 29 lakes were determined to pose high risks of zebra mussel establishment.

Table 5. HSI Values and Risk of Dreissenid Establishment in 70 Selected Lakes in Wisconsin and Minnesota.

S#	St. Croix Basin Lake	HIS Minimum SI	HIS Geometric Mean	Risk Level
1	**Barker**	0.028	0.717	Low
2	**Bass**	0.459	0.879	Medium
3	**Big Carnelian**	0.001	0.606	Low
4	**Big Marine**	0.004	0.742	Low
5	**Birch**	0.028	0.746	Low
6	**Bone**	0.445	0.907	Medium
7	**Carol**	0.355	0.829	Medium
8	**Chisago**	0.135	0.82	Medium
9	**Cloverdale**	0.026	0.669	Low
10	**Comfort**	0.059	0.737	Low
11	**Coon**	0.877	0.97	High
12	**Downs**	0.466	0.821	Medium
13	**East Boot**	0.069	0.483	Low
14	**Edith**	0.094	0.656	Low
15	**Elwell**	0.98	0.991	High
16	**Fawn**	0	0.816	Medium
17	**Fish**	0.001	0.8	Medium
18	**Forest**	0.876	0.971	High
19	**Goose**	0.472	0.888	Medium
20	**Green**	0.242	0.878	Medium

Table 5.HSI Values and Risk of Dreissenid Establishment in 70 Selected Lakes in Wisconsin and Minnesota. (Continued.)

S#	St. Croix Basin Lake	HIS Minimum SI	HIS Geometric Mean	Risk Level
21	**Halfbreed**	0	0.645	Low
22	**Hay**	0.762	0.93	High
23	**Horseshoe**	1	1	High
24	**Island**	0.572	0.928	High
25	**Jellums**	0.618	0.89	High
26	**Klawitter Pond**	0.893	0.964	High
27	**Kroon**	0.992	0.996	High
28	**Legion Pond**	0.949	0.983	High
29	**Lily**	0.1	0.675	Low
30	**Linwood**	0.04	0.804	Medium
31	**Little**	0.987	0.987	High
32	**Little Carnelian**	0	0.616	Low
33	**Little Comfort**	0.009	0.72	Low
34	**Long**	0.814	0.942	High
35	**Loon**	0.359	0.82	Medium
36	**Louise**	0.814	0.951	High
37	**Mandall**	0.997	0.998	High
38	**Martin**	0.823	0.964	High
39	**Mays**	0.912	0.978	High
40	**McDonald**	0.849	0.958	High
41	**McKusick**	0.664	0.91	High
42	**Mergen's Pond**	0.979	0.992	High
43	**Moody**	0.005	0.514	Low
44	**Mud**	0.71	0.897	High
45	**North Center**	0.135	0.832	Medium
46	**North Center Pond**	0.088	0.65	Low
47	**North Twin**	0.73	0.889	High
48	**O'Connors**	0.984	0.991	High
49	**Pioneer**	0.358	0.42	Low
50	**Rabour**	0.01	0.504	Low
51	**Rush**	0.755	0.973	High
52	**S. School Section**	0.421	0.833	Medium
53	**Sand**	0.527	0.861	High
54	**School**	0.016	0.668	Low
55	**Shields**	0.077	0.812	Medium
56	**Silver**	0.416	0.842	Medium
57	**South Lindstrom**	0.08	0.821	Medium
58	**South Twin**	0.571	0.89	High
59	**Square**	0	0.641	Low

Table 5. HSI Values and Risk of Dreissenid Establishment in 70 Selected Lakes in Wisconsin and Minnesota. (Continued.)

S#	St. Croix Basin Lake	HIS Minimum SI	HIS Geometric Mean	Risk Level
60	**St. Croix**	0.895	0.983	High
61	**Staples**	0.119	0.565	Low
62	**Sunfish**	0.839	0.945	High
63	**Sunnybrook**	0.932	0.967	High
64	**Sunrise**	0.535	0.884	High
65	**Tamarack**	0	0.335	Low
66	**Terrapin**	0.003	0.763	Low
67	**Turtle**	0.166	0.82	Medium
68	**Twin**	0	0.6	Low
69	**Typo**	0.149	0.761	Low
70	**Wallmark**	0.331	0.708	Low

Spatial Characterization of Risk

The results for selected water bodies presented in Table 5 were mapped to develop a preliminary spatial characterization of risk for invasive mussel establishment in the St. Croix Basin (Figures 22 and 23). Additional Wisconsin and Minnesota water bodies were also analyzed and plotted. These lakes were selected in part to examine the model results for surface waters outside the St. Croix Basin and with the recognition that additional pressures for invasion come from infested water bodies outside the Basin (e.g., progressive invasions of Wisconsin lakes from recreational boaters using the Lake Michigan). The results indicate that lakes at similar levels of risk tend to cluster on the landscape. The spatial results, especially for Minnesota lakes within the Basin, illustrate the likely importance of the UMR as the original source of infestation for the St. Croix Basin. The northern Wisconsin low-risk lakes likely reflect the comparatively lower pH and lower total hardness attributes of water bodies in this region. Importantly, use of the HSI scores for these systems along with their location, and the location of currently infested systems might assist in forecasting the direction and rate of mussel establishment throughout the Basin. These kinds of forecasts could be used to develop effective management control strategies aimed at minimizing the rate of spread and establishment of dreissenids in the Basin.

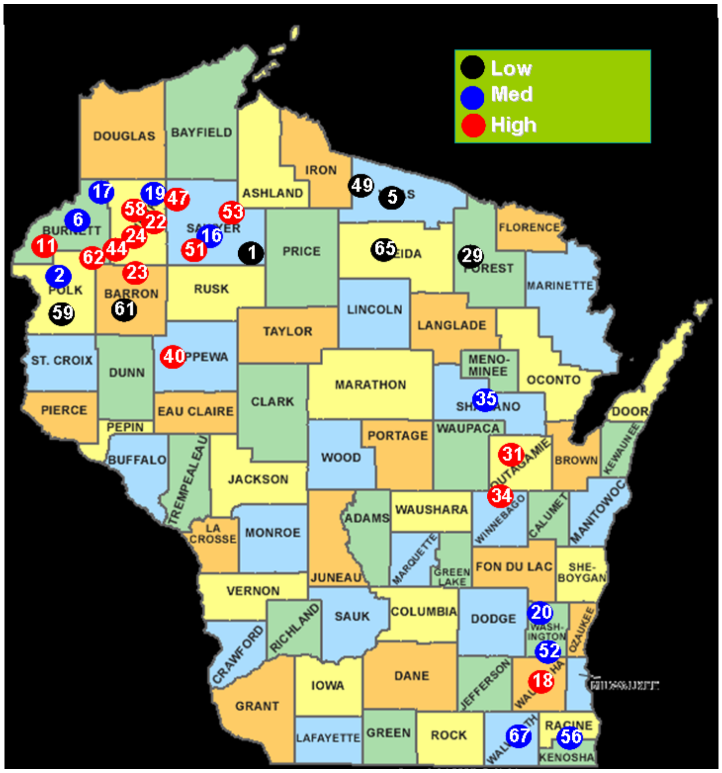

Figure 22. Location of selected lakes in Wisconsin characterized by different degrees of risk for invasive mussel establishment.

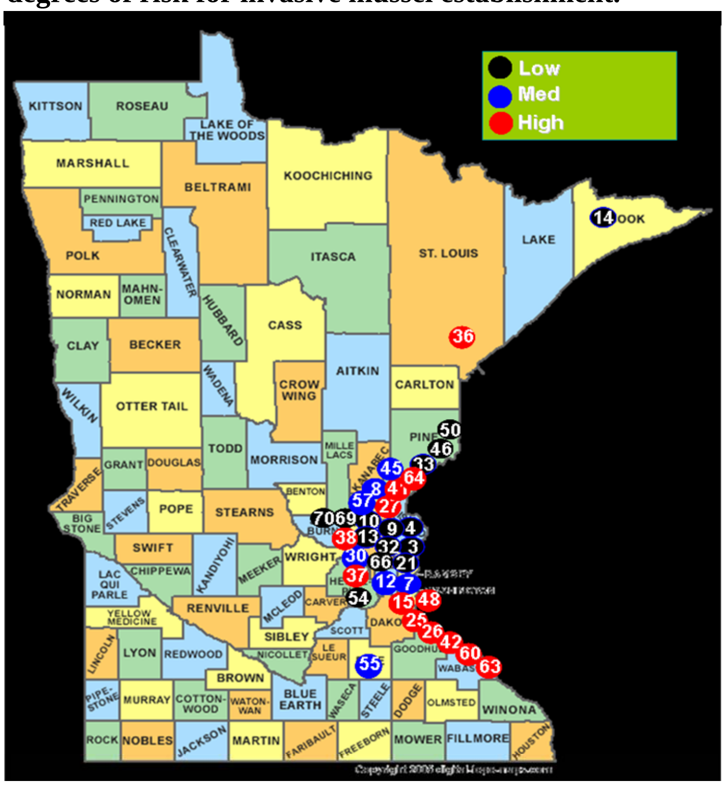

Figure 23. Location of selected lakes in Minnesota characterized by different degrees of risk for invasive mussel establishment.

Risk-Reduction and Invasive Mussel Management

The risk-based decision model also provides the conceptual basis for evaluating alternative approaches for controlling or managing the rate of establishment and spread of the zebra and quagga mussels throughout the St. Croix Basin (Figure 24). The previous discussion focused on use of the risk-based decision model for estimating a baseline risk, R_B, of dreissenid mussel invasion and establishment for selected water bodies in the basin. The risk assessments involved development of values of habitat parameters (H) for the systems included in the analysis. The location (L) and inoculation (I) parameters remain to be made operational components of the model, although the locations of the analyzed systems have been addressed.

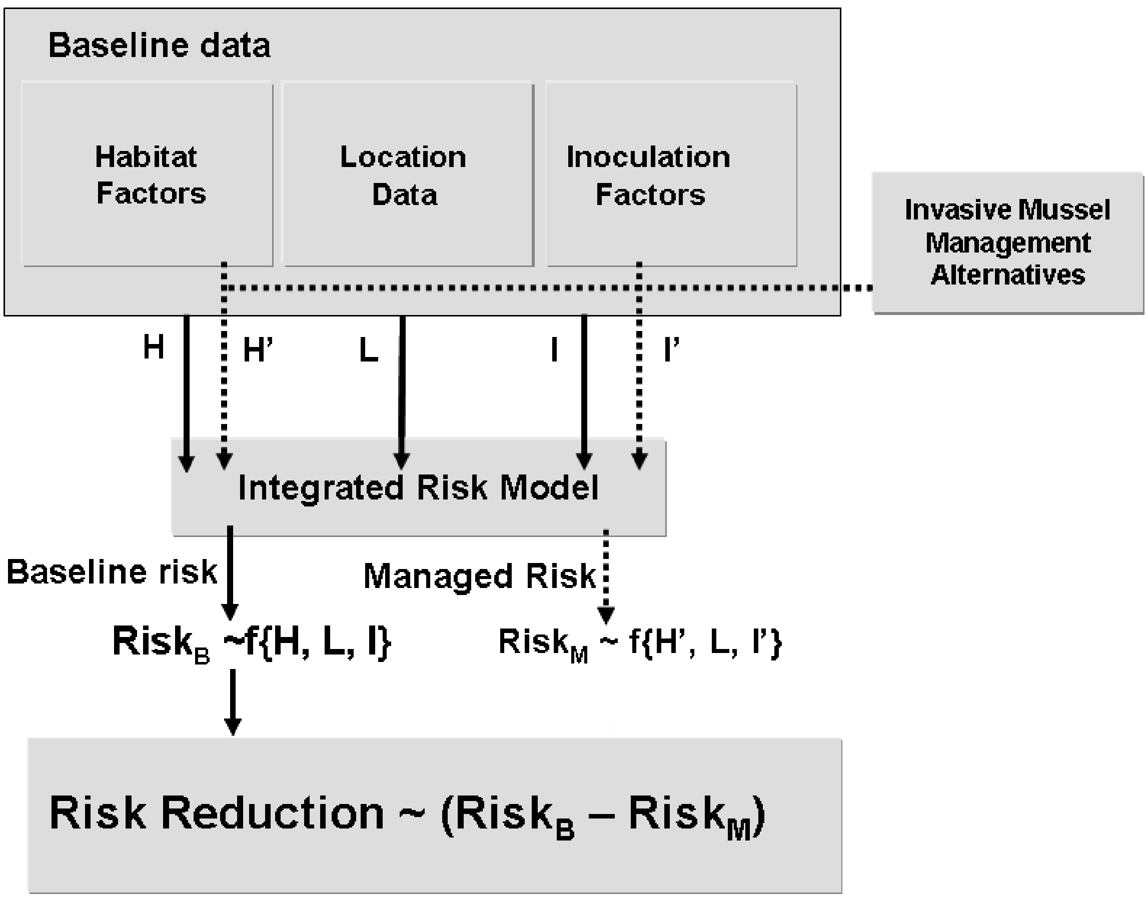

Figure 24. Conceptual model for risk reduction in relation to invasive mussel management.

Application of the decision model to evaluate the efficacy of a proposed management alternative essentially involves estimating the risk of mussel establishment subsequent to implementing the management action, that is, managed risk, R_M. The reduction in risk (i.e., R_B–R_M) can be estimated for feasible management alternatives to identify the most promising management action for individual water bodies within the basin (Figure 24).

The structure of the decision model provides some insight concerning its application in evaluating alternative mussel management actions. Clearly, the physical aspect of the L component of the model will remain unchanged in assessing baseline or managed risk. However, the functional aspect of location (e.g., a stream or tributary connecting a non-infested and infested water body) might be managed using a biocide or engineering structure. Additionally, it seems not practical, economical, or perhaps technically feasible to reduce habitat suitability for invasive mussels. None of the 14 water quality parameters appear manageable at the system level. Even if it were feasible, degradation of habitat to discourage dreissenid establishment might similarly impair native mussels that are also affected by these water quality parameters. Nevertheless, there might be circumstances where proposed modification of selected water quality factors is included as a management alternative. In the model, this would correspond to a modification of the vector of water quality factors (H'). Finally, it appears most likely that invasive mussel management would focus on reducing the likelihood and/or rate of dreissenid introductions to non-infested water bodies within the basin. These management actions would be represented in the model as reductions in the inoculation component (e.g., reduced public access, enforced recreational boat hull cleaning) of the model, which would manifest as a modified vector of inoculation parameters, (I'). Using the decision model with the baseline and modified parameter vectors would then produce an estimate of baseline and managed risk of mussel establishment for each assessed water body and management action. The management action selected among alternatives based on effectiveness could then enter the incremental cost analysis component of the USACOE planning process.

Application to Native Mussel Risk Assessment and Management

A framework for assessing risks posed to endangered unionid mussels should (1) evaluate the likelihood that invasive mussels will become established in water bodies currently or potentially inhabited by the native mussels, and (2) characterize the negative impacts of invasive mussels, once established, on the biology and ecology of the native mussels. Given an assessment that invasive mussels are highly likely to establish persistent populations in an endangered mussel habitat, risk managers might desire a framework that also assisted in identifying and evaluating the effectiveness of alternative invasive mussel control strategies. The following sections outline such a risk assessment and risk management framework for invasive mussels in the St. Croix Basin.

The proposed framework first addresses the chance that zebra or quagga mussels might become established in surface waters inhabited by endangered unionid mussels. The risk-based decision support model used to characterize the likelihood of zebra mussel establishment can be modified to address the potential implications for endangered unionid species, such as the winged mapleleaf and Higgins eye mussels. This modification can take two forms (1) the likelihood of zebra mussel establishment can be estimated for specific water bodies known to be inhabited by the endangered unionids, and (2) habitat suitability for both invasive and the endangered unionid mussels can be characterized for water bodies where winged mapleleaf or Higgins eye might occur. Clearly, the first modification is simply a site-specific subset of the second modification, wherein one of the model location parameters specifies whether the water body (i.e., river segment) is inhabited by winged mapleleaf or Higgins eye. This approach permits convenient modification to include other unionid mussels that might be of future concern.

As an initial step towards implementing the combined habitat evaluation for invasive and native unionid mussels, several preliminary suitability index functions have been derived using limited information available for the Higgins eye mussel (Figures 25–29). As a fundamentally large river mussel, Higgins eye mussels are capable of inhabiting surface waters with higher current velocities than zebra mussels (Figure 25). Zebra mussels are more characteristic of lentic environments. However, habitats with a velocity of ~1 m/s appear very inhabitable by both species.

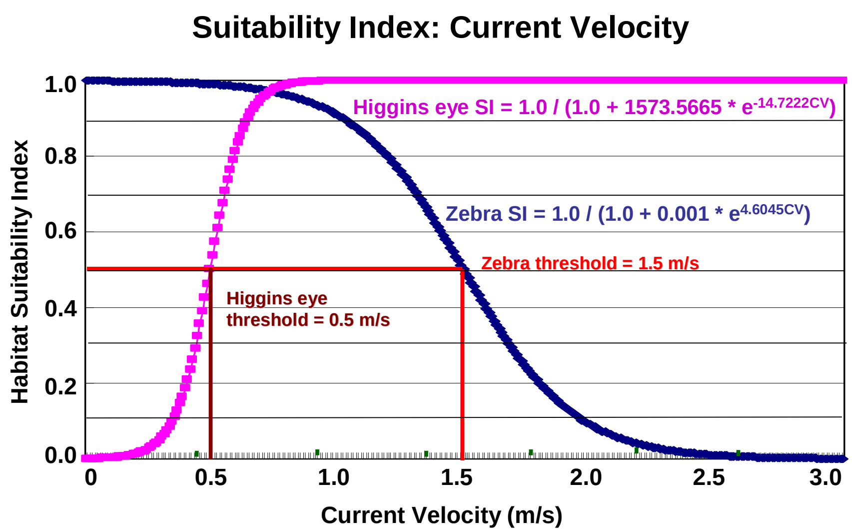

Figure 25. Habitat suitability as a function of water current velocity for zebra and Higgins eye mussels.

Higgins eye mussels also appear to prefer deeper water than zebra mussels (Figure 26).

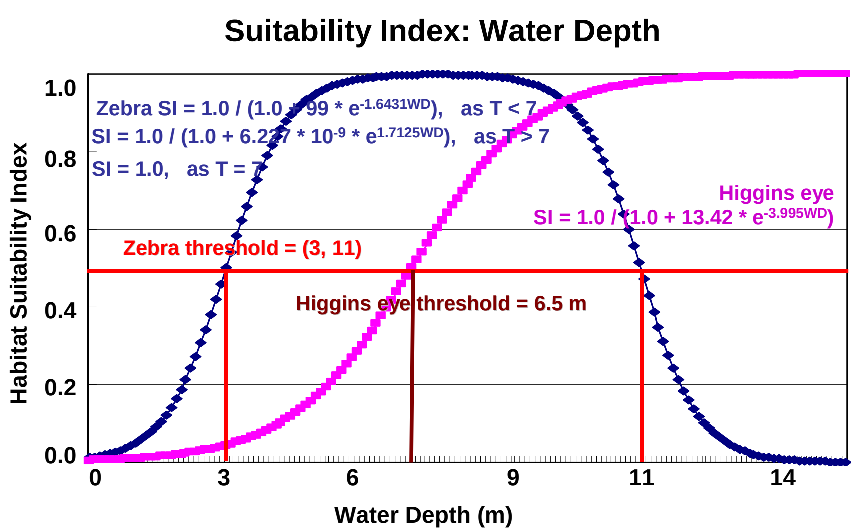

Figure 26. Habitat suitability as a function of water depth for zebra and Higgins eye mussels.

Zebra mussels appear to tolerate somewhat lower concentrations of DO than Higgins eye mussels (Figure 27).

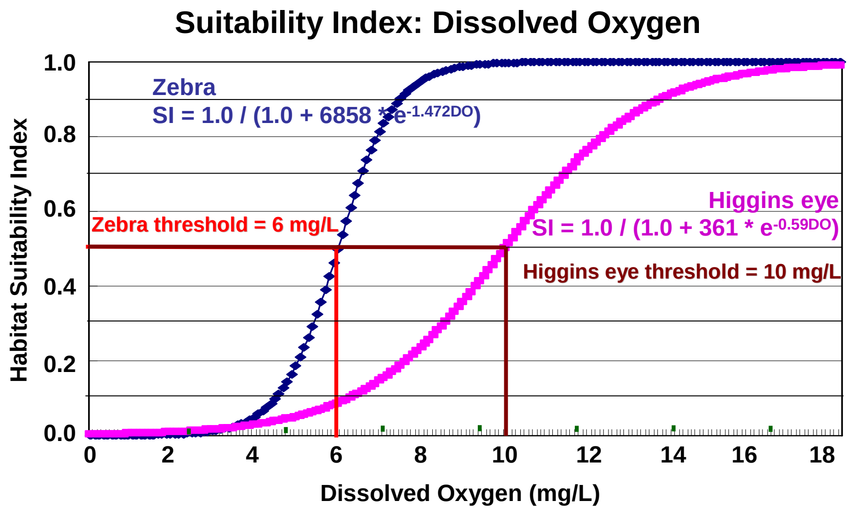

Figure 27. Habitat suitability as a function of DO for zebra and Higgins eye mussels.

Figure 28 suggests that zebra mussels are able to take advantage of waters characterized by lower concentrations of dissolved calcium than Higgins eye mussels.

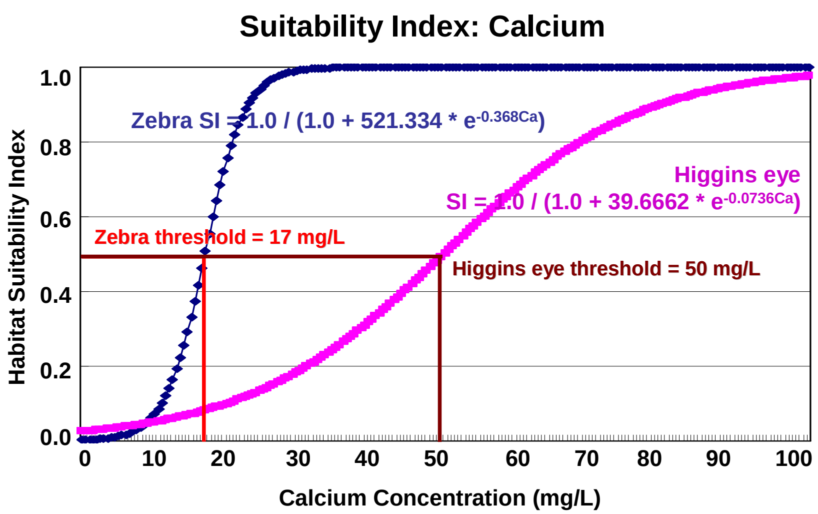

Figure 28. Habitat suitability as a function of calcium concentration for zebra and Higgins eye mussels.

Zebra mussels are also slightly more tolerant to concentrations of ammonia than are Higgins eye mussels (Figure 29).

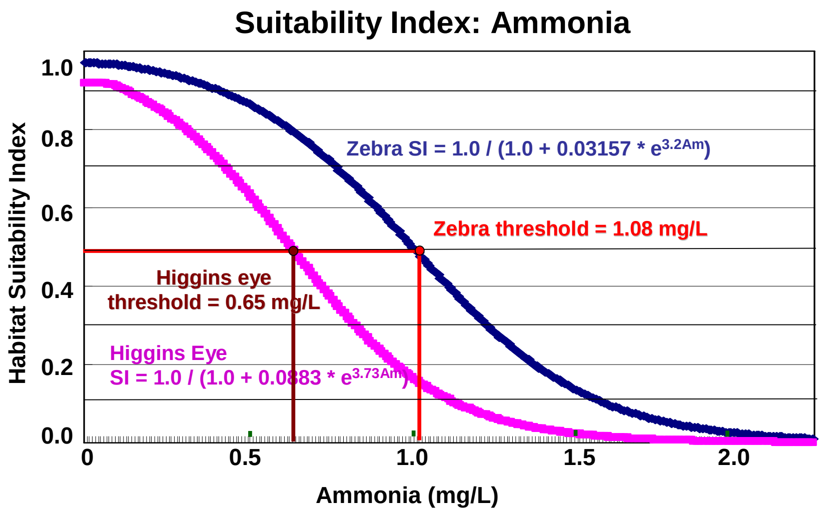

Figure 29. Habitat suitability as a function of ammonia concentration for zebra and Higgins eye mussels.

Less information is available for developing corresponding habitat suitability functions for the winged mapleleaf mussel.
It is assumed that the potential negative impacts (risks) of invasive mussels on endangered unionids are positively correlated with the population size of the established invasive mussel. Therefore, the proposed framework for native mussel risk assessment and management will include the capability to translate invasive habitat quality into estimates of population growth rate and population sizes for the colonized water body. Empirical relationships or mechanistic models (e.g., Bartell et al. 1999, Kennedy and Ackakaya 1999) will be included in the framework to characterize the effects of estimated abundance of established invasive mussels on (1) food availability for native unionid mussels, (2) physical impairment (e.g., reduced ability to open and close, reduced mobility) of native mussels directly colonized by invasive mussels, and (3) reduced habitat quality (e.g., increased ammonia concentrations, decreased DO) on native mussel growth and survival.

The risk assessment and management framework will include the capability to evaluate the effectiveness of selected invasive mussel control strategies, including a no-action alternative. Zebra mussel control technologies have previously emphasized physical or chemical treatment of structures that become fouled by colonizing mussels (e.g., ERDC 1994). Biological control of zebra mussels in circulating water infrastructure has also been explored (e.g., Molloy 2002, 1998). As stated earlier, it is not likely that zebra mussels can be controlled effectively or economically by manipulating water quality factors at the scales of entire lakes and extensive river segments. However, the specific large river habitat preferences of the winged mapleleaf and Higgins eye mussels might afford an opportunity to manage or control the rate of spread or subsequent impacts of invasive dreissenids in more localized habitats critical to the endangered unionids in the St. Croix Basin. Possible management and control strategies include (1) restriction of recreational boating on the St. Croix River (e.g., beyond Arcola sandbar north of Stillwater, Minnesota); (2) inspection, cleaning, or quarantine of commercial and recreational vessels; (3) construction of hot water station (e.g., using heated effluent from power plant near Lock 3) to kill mussels attached to vessel hulls; and (4) application of toxic and non-toxic paints or coatings to boat hulls (USACOE 2003). Other technologies (e.g., ultraviolet light, pulses of electricity, physical barriers) have been considered, but these more exotic methods seem infeasible for implementation in larger river systems, even at more local scales defined by winged mapleleaf or Higgins eye habitat (USACOE 2003). Nevertheless, the risk assessment and management framework will permit evaluation of the effectiveness, cost, and uncertainty associated with use of existing technologies in controlling or managing the infestation of endangered mussel habitats by zebra and quagga mussels. The framework will include the capability to perform sensitivity and uncertainty analyses in characterizing the likely effectiveness of management alternatives; these analyses will importantly identify the key data needed to improve these characterizations and better inform the management decision making process.

Risk-based Invasive Mussel Decision Model

Figure 30 illustrates a framework to assess the probability of invasive mussel establishment in selected surface waters of the St. Croix Basin. The model begins with a non-infested surface water (e.g., lake, river segment) and estimates the probability of invasive mussel establishment. The model importantly evaluates the effectiveness of alternative management actions aimed at controlling mussel establishment, assesses the consequences of new infestations, and addresses mitigation of potential impacts on endangered native unionid mussels. Upon implementation, the decision model will represent an operational extension of the conceptual models for invasive mussel establishment and evaluation of management alternatives.

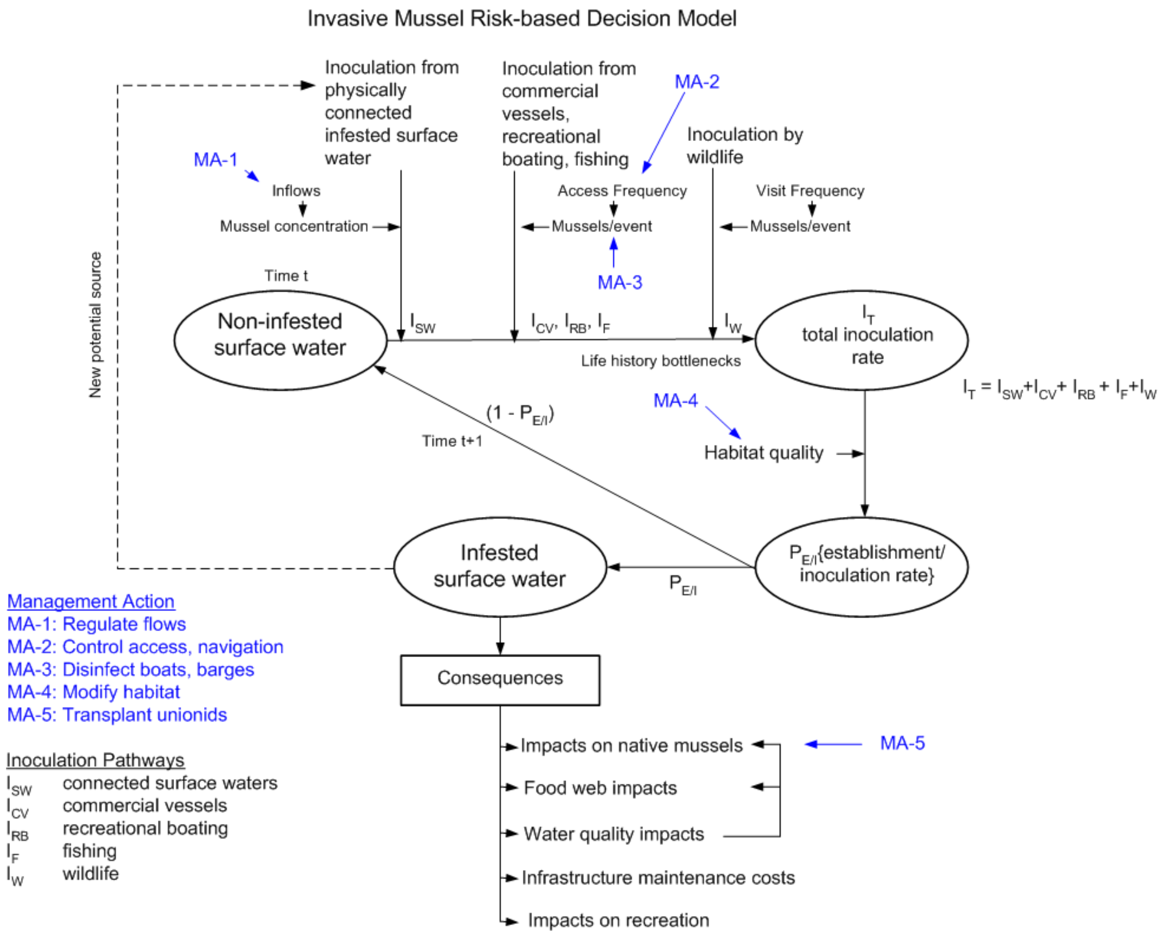

Figure 30. Multi-criteria risk-based decision model to (1) assess the probability of infestation by invasive mussels, (2) characterize ecological and economic consequences of mussel establishment, and (3) evaluate alternative management actions directed at controlling mussel spread.

Multi-Criteria Decision Approach

The overall approach is a consistent in concept with the multi-criteria decision analysis encouraged by Payne and Miller (2004). The model attempts to be comprehensive by including the many recognized events, processes, and pathways that convey invasive mussels to non-infested surface waters. Each of these mechanisms of inoculation can be associated with management actions and decision criteria aimed at reducing the likelihood of introducing invasive mussels.

The proposed model, however, (i.e., Figure 30) differs in technique. Instead of a using fixed, static decision-tree calculus, the risk-based decision model is a dynamic, event-driven simulation model. Patterned after the Lock and Dam 3 Outdraft Decision Model (Bartell and Nair 2000), the invasive mussel simulation model provides the structural flexibility to easily incorporate new (or remove unnecessary) components, represent time-varying aspects of invasive mussel establishment (e.g., mussel life history and biology/ecology, habitat quality, commercial navigation and recreational boating intensity, fishing, etc.), and explicitly characterize and propagate uncertainties associated with model assumptions and parameter estimates. The model performs numerical sensitivity and uncertainty analyses using Monte Carlo methods as part of a standard execution of the model. The results of these analyses identify the key contributors to uncertain estimates of establishment and define the value of new information that can be obtained to reduce uncertainties, improve model performance, and increase the usefulness of model results in management and decision-making.

Model Description

The model begins with a selected non-infested surface water body in the St.Croix Basin at some chosen time (t). The model then steps through time (e.g., one week), estimates the rate of inoculation by recognized pathways of importance, and determines an overall or total inoculation rate (I_T) during the time period. Inoculation refers to the input of adult, juvenile, or veligers into a previously non-infested system.

The total inoculation rate is estimated as the sum of five separate rates defined by alternative mechanisms for introducing invasive mussels into non-infested systems:
I_{SW}: Introduction of invasive mussels via inflows from a physically connected upstream infested system. This is modeled as a time-varying product of flow rate and the concentration of different life stages of invasive mussels.

I_{CV}: Introduction of invasive mussels by commercial navigation. This pathways refers primarily to Pools 2-4 on the Upper Mississippi River and any commercial traffic on major tributaries. This influx is the product of vessels per time step (i.e., access frequency) in the river reach of interest and the number of mussels potentially released from each vessel (i.e., barges and tow boat).

I_{RB}: Introduction of invasive mussels by recreational boating, including fishing boats. This inoculation rate is modeled as the product of access frequency (i.e., recreational boats per time step) and the number of mussels potentially released from each boat hull.

I_F: Introduction of invasive mussels by recreational fishing. This inoculation rate is modeled as the product of access frequency (i.e., fishing trips per time step) and the number of mussels potentially released by each fisherman (e.g., contaminated fishing gear, bait buckets).

I_W: Introduction of invasive mussels by wildlife (e.g., waterfowl, wading birds, small mammals) that visit the non-infested water body. This inoculation rate is modeled as the product of the frequency of visits per time step and the number of mussels potentially introduced by each organism. This pathway is probably of minimal importance (i.e., $I_W \sim 0$) and is least likely to be managed. However, this pathway was included for completeness.

Then, for each modeled life stage (veliger, juvenile, adult) and time step,

$$I_T = I_{SW} + I_{CV} + I_{RB} + I_F + I_W$$

Habitat quality for invasive mussels contributes to estimation of $P_{E/I}$, the probability of mussel establishment (i.e., self-reproducing population) given the total inoculation rate. Unfavorable values of the physical-chemical factors that influence mussel survival, growth, and reproduction can produce sufficiently low estimates of $P_{E/I}$ that the system remains non-infested (i.e., lacks a self-reproducing population) with a probability of $(1 - P_{E/I})$. Newly infested surface waters can subsequently serve as upstream sources for infestation (indicated by the dashed line in Figure 30).

This model construct requires derivation of a functional relationship between numbers of introduced mussels and the likelihood of establishing a self-reproducing population. Assumptions of a threshold population size for establishment may be required.

Consequences

For newly infested surface waters, the model characterizes the ecological and economic consequences of infestation. The ecological consequences include impacts on water quality (e.g., increased ammonia concentrations, increased water transparency), alterations of aquatic food webs (e.g., depletion of plankton), and effects on native mussels, especially endangered unionid mussels. Impacts on mussels are modeled as direct physical attachment or smothering, potential food limitation, and ammonia toxicity. One ecological consequence includes the need for removal or transplanting of unionid mussels–this is also addressed as Management Action 5.

Knowledge of existing infrastructure in the newly infested system (e.g., water intake pipes, buoys, piers, lock chambers, etc.) permits estimation of repair or maintenance costs that result from mussel establishment.

Management Actions

A primary objective in developing the decision model is the evaluation of alternative management actions that can impede the spread of invasive mussels throughout the St. Croix Basin. Figure 30 identifies four (MA-1 – MA-4) of these possible management actions. An additional action (MA-5) pertains to mitigating the potential impacts of new infestations on native unionid mussels. An implicit MA-0, no management action, is understood, but not illustrated.

MA-1 refers to the opportunity to regulate flows between physically connected infested and non-infested surface waters. For example, flows might be reduced, if possible, during periods when veligers or juveniles are present in large numbers.

MA-2 identifies actions that can be taken to schedule commercial navigation, restrict access by recreational boats, and control fishing to reduce the introduction of invasive mussels to non-infested surface waters.

MA-3 includes actions taken to reduce contamination of commercial barges and recreational boats by invasive mussels. Such actions could take the form of vessel quarantine and heating, drying or other treatments to disinfect boat hulls. Restrictions on the use of bait buckets and insistence on cleaning fishing gear could reduce inadvertent introduction by fisherman.

MA-4 addresses any action that might be undertaken to reduce habitat quality for invasive mussels introduced into non-infested surface waters. While potentially effective, the logistic difficulties in implementing habitat changes (e.g., reduced calcium) at a system-level scale suggest that this management action will seldom prove feasible. Nevertheless, MA-4 is included for those unusual opportunities where such manipulations might prove doable.

MA-5 does not attempt to reduce rates of inoculation as in MA-1 through MA-4. Rather, MA-5 addresses the feasibility of mitigating impacts on native mussels by removing and relocating unionids to conditions that are safe from invasive mussel infestations.

The model permits the specification of these management actions as input variables that modify flows, limit access, reduce mussel contamination of vessels, or redefine invasive habitat suitability factors for the system of interest. The associated reductions in inoculation rates are then translated into reduced values of $P_{E/I}$ and reduced values of ecological and economic consequences. Of course, MA-5 can be specified independently of any MA-1 through MA-4 alternatives.

Uncertainties

The formulation of the risk-based decision model (i.e., Figure 30) and estimation of necessary parameters implies uncertainties in model implementation and in the evaluation of management alternatives. Where possible, the input parameters are defined as statistical distributions to characterize uncertainty. Monte Carlo methods are used to propagate these uncertain parameter values through the model and characterize uncertainties on model results.

This approach also provides the capability for numerical sensitivity and uncertainty analyses as part of the overall decision model. The results of these analyses can be used to identify key sources of uncertainty as they influence the selection among management alternatives. Additional data can be collected that will reduce uncertainties and increase the usefulness of model results in management and decision-making.

Habitat Uncertainty

There are several sources of uncertainty associated with estimating habitat quality for establishment of invasive dreissenid mussels. First, the exact nature of the functions for the individual suitability indices were derived for systems other than surface waters in the St. Croix Basin. It is not known if these habitat suitability functions apply directly to the surface waters of interest in the Basin. However, it is possible to address this source of uncertainty through field observations or experiments performed using dreissenid mussels in selected lakes and stream segments within the Basin. Modifications to the existing suitability functions can be made as necessary based on the results of these kinds of studies.

Second, individual mussels might vary in their response to values of the physical-chemical factors included in the habitat characterization. This source of uncertainty can also be addressed through continued monitoring and particularly through experiments under controlled conditions using dreisseid mussels collected in Basin surface waters. In addition, to offset the variability of response, certain habitat factors may be weighted more or less than other factors to illustrate the impact that habitat factor has on the growth or spread of dreisseid mussels.

Third, the quality of the data used to estimate habitat suitability for the 14 factors varies widely throughout the Basin. The number of individual factors monitored for any individual system rarely includes all 14. The frequency and intensity of sampling varies among the factors within any single water body. Sampling frequency and intensity also varies substantially among the lakes, rivers, and streams of interest in the Basin. A standardized approach for monitoring dreissenid habitat quality can be put in place to reduce this source of uncertainty in habitat characterization.

Location Uncertainty

As suggested, a primary source of uncertainty in characterizing location concerns an incomplete description of the physical connections among surface waters in the Basin. This pertains particularly to smaller lakes that might be connected by streams that are dry during certain times of the year. Areal mapping of the Basin at selected (e.g., seasonal) times could provide the necessary data to reduce this source of uncertainty. Another source of uncertainty in describing location is the proximity to an infested system. The accuracy and completeness of current lists of infested surface waters within the Basin remain unknown. A periodic and standardized survey can be implemented to reduce this source of uncertainty.

Inoculation Uncertainty

Of the three model components, inoculation might be the most challenging to quantify within acceptable degrees of accuracy and precision. The shear number of Basin lakes, rivers, and streams potentially visited by recreational boaters and fishermen makes it logistically difficult and expensive to rigorously monitor access. Even if access can be adequately characterized, it remains difficult to know if an individual visit constitutes an inoculation event without corresponding inspection of boat hulls, bait buckets, fishing tackle, etc. It may prove difficult to reduce uncertainties associated with this component of the mussel establishment model. One possible solution would be to purposely bias the model towards overestimating inoculation (i.e., make the model conservative in this aspect). If such bias in estimation of establishment can be applied consistently, the subsequent risk-based decision model can still be used to evaluate the relative effectiveness of invasive mussel management alternatives.

Time Scales

The model time step is largely defined by the availability of data. For example, habitat quality data are available on a weekly basis for some surface waters in the St. Croix Basin. Other systems are sampled less frequently, although weekly values might be reasonably interpolated.
The temporal duration (e.g., number of time steps) can be defined by the model user. Clearly, the longer the simulated period, the greater is the chance for invasive mussel establishment. A 50-year planning horizon is commonly used by the Corps in its evaluation of management alternatives. Thus, a convenient temporal scale for the model would be weekly time steps for 50 years. The simulation could terminate after any time step if the non-infested water body was deemed infested, although the simulation might continue in order to accrue the full consequences for the entire planning period for purposes of comparing planning alternatives.

Spatial Extent

The model illustrated in Figure 30 applies to a single selected surface water body in the Basin. The model can be executed for as many non-infested systems as desired, depending on the availability of data. As a result of applying the model to many systems, a landscape pattern of infestation can emerge. The spread of the invasive mussels throughout the Basin will be influenced as physically connected systems become infested. The decision model is compatible with GIS summarization and presentation of mussel infestation across the landscape.

Evaluation of Invasive Mussel Management Alternatives

The main purpose of the risk-based decision model (Figure 30) is to estimate the risk of invasive mussel establishment for a non-infested surface water body in the St. Croix Basin. An equally important application of the decision model lies in characterizing the reductions in risk afforded by the proposed alternative management actions, including the no action alternative. As discussed previously, the existing data do not yet permit the actual implementation of the decision model.

As a prelude to the completion and application of an operational model, aspects of the proposed management actions can be evaluated, at least qualitatively, in relation to reducing risks posed by invasive mussels in the Basin (Table 6). These management actions were assessed in terms of their potential effectiveness in (1) controlling the rate of invasive mussel dispersal; (2) reducing the population sizes or levels of infestation; (3) minimizing ecological impacts on native mussels, especially endangered unionid mussels; (4) reducing economic costs associated with invasive mussel establishment; and (5) reducing overall risk of continued establishment throughout the Basin. Table 6 summarizes the results of this qualitative evaluation.

Table 6. Evaluation of invasive mussel management alternatives in the St. Croix Basin.

Management alternative	Dispersal	Infestation levels	Ecological impacts	Economic costs	Risk reduction
MA-0 No action.	Currently observed patterns and rates of dispersal throughout the Basin are anticipated to continue.	Currently measured intensities of infestation are anticipated in surface waters distributed throughout the Basin.	As invasive mussels increase in distribution and abundance, impacts on native mussels will likely increase.	Damages resulting from invasive mussel establishment will continue to increase; costs may reflect observed cycles in invasive mussel abundance.	None, by definition.
MA-1 Flow regulation	Dispersal throughout physically connected systems can be reduced; not effective for isolated surface waters.	Can reduce the levels of infestation in downstream surface waters that are physically connected to infested upstream systems.	Not likely to reduce impacts on native unionid mussels which inhabit main channels where flows are not likely to be regulated.	Damages might be reduced for those downstream surface waters physically connected to infested upstream systems.	Degree of risk reduction determined by relative numbers of physically connected surface in waters in the St. Croix Basin; risks not likely reduced in isolated surface waters.
MA-2 Control access	Particularly effective for reducing rates of dispersal to isolated surface waters; less effective for physically connected systems.	Not directly relevant for controlling levels of infestation; affects infestation mainly through reducing inoculation and dispersal.	Might reduce impacts on native mussels where access to main channels in larger rivers is regulated; not relevant for smaller systems not inhabited by endangered unionids.	Damages will be reduced in relation to reductions in levels of dispersal and infestation; costs determined in part by the value of infrastructure in isolated surface waters.	Might produce the greatest reductions in risk if the Basin is dominated by noninfested isolated surface waters.

Table 6. Evaluation of invasive mussel management alternatives in the St. Croix Basin. (Continued.)

Management alternative	Dispersal	Infestation levels	Ecological impacts	Economic costs	Risk reduction
MA-3 Vessel treatment	Particularly effective for reducing rates of dispersal to isolated surface waters (i.e., treatment of recreational boats); perhaps somewhat less effective for physically connected systems, including larger rivers (i.e., treatment of commercial vessels and barges).	Not directly relevant for controlling levels of infestation; affects infestation mainly through reducing inoculation.	Relevant if commercial navigation is a significant source of infestations in larger channels inhabited by native unionid mussels.	Can provide the greatest reduction in economic damages if the primary vectors for inoculation are commercial navigation and recreational boating.	Can provide the greatest reduction in risk if the primary vectors for inoculation are commercial navigation and recreational boating; maybe less feasible for commercial vessels and barges due to costs of treatment.
MA-4 Habitat modification	Alteration of potential invasive mussel habitat can reduce currently observed rates of dispersal throughout the Basin.	Alteration of potential invasive mussel habitat can reduce currently measured levels of infestation throughout the Basin; perhaps most effective means of reducing levels of infestation.	Not likely to be effective in larger river channels inhabited by endangered native mussels.	Damages might be reduced, but only for those few surface waters where it proves feasible to alter habitat quality in sufficient magnitude and extent.	The great logistic difficulties in modifying habitat factors in entire water bodies suggest that this management action is not practical; thus, minimal reductions in risk are anticipated.
MA-5 Mussel relocation	N/A	N/A	Relocation of endangered unionid mussels can help reduce or offset the increasing impacts of invasive mussels on selected species, e.g., winged mapleleaf, Higgins eye pearly mussel.	N/A	Only relevant to reducing risks of local dreissenid infestations on endangered unionid mussels.

Despite the limitations of this qualitative evaluation, some insights emerge concerning the relative effectiveness of the proposed management alternatives. Clearly, if no action is taken (MA-0), invasive mussels will continue to disperse and establish in surface waters throughout the Basin. Economic damages will continue to increase, although observed cyclic changes in invasive mussel abundance can affect the overall future costs associated with unmanaged invasive mussels. Relocation of endangered unionid mussels (MA-5) bears no direct impact on reducing risks associated with continued establishment of invasive mussels. However, risks to endangered unionids might be reduced through relocation to habitats that are of marginal quality to invasive dreissenids. MA-1 is relevant only for physically connected surface waters that are downstream from upstream infested surface waters. This action might be particularly effective for smaller systems, where flows could be effectively regulated during periods critical in the life history of the invasive dreissenids. Regulation of flows for larger systems (e.g., Navigation Pools 2-4, larger rivers) might be more difficult because flows are also managed for other reasons (e.g., navigation, flood protection). Habitat alteration (MA-4) might prove to be very effective in controlling dreissenid mussel establishment and level of infestation. However, the logistical challenges associated with modifying habitat factors at sufficient scale and magnitude will likely limit the application of this approach. Of primary interest are management actions that control access (MA-2) to non-infested surface waters or that require the treatment (or quarantine) of vessels (MA-3) that are moved from one water body to another. Both actions focus mainly on reducing the rate of inoculation of invasive mussels throughout the Basin. Vessel treatment can be particularly effective for recreational boats; this action might be too costly for commercial vessels and barges, however. It appears likely that the combination of controlling access and treating boat hulls will prove the most effective in reducing the rate of spread and establishment throughout the Basin, especially for isolated surface waters.

The preceding evaluation was qualitative. Given sufficient data, the decision model would be used to quantitatively evaluate the effectiveness of the relevant management alternatives in reducing the risk of invasive mussel establishment for each of the surface waters within the St. Croix Basin, including Pools 2-4 on the Upper Mississippi River. Consistent with the Corps planning process, each evaluation would focus on the incremental reduction in risk compared to the no action alternative. Risk reductions and their associated costs could then enter into an incremental cost analysis to identify the most cost-effective management action for each surface water body in the Basin subject to infestation.

Summary

A risk-based decision model was designed to address the likelihood and rate of dreissenid mussel spread throughout surface waters in the St. Croix River Basin, including Navigation Pools 2–4 in the UMR. The model comprises three components that determine mussel invasion and establishment in a non-infested water body: (1) habitat quality, (2) location in relation to infested sites, and (3) inoculation rate. In this initial phase of the project, only the habitat suitability and location components of the risk-based decision model were implemented. Water quality data were collated from existing state (i.e., Minnesota, Wisconsin) and federal (i.e., USEPA) databases. These collated data were further analyzed to provide the necessary input values to the habitat suitability model. Location parameters were obtained to define the position of selected water bodies within a GIS description of the St. Croix Basin.

The results of the initial implementation of the risk-based decision model demonstrate the feasibility of assessing habitat suitability for invasive mussels in the St. Croix Basin. Using existing water quality data and newly derived HSI functions, the decision model was able to classify 77 surface waters within the basin as low, medium, or high risk for dreissenid mussel establishment. More detailed analyses indicated which of the 14 habitat factors might reduce the likelihood of mussel establishment in selected lakes.

The decision model provides a conceptual template for evaluating the effectiveness of alternative management actions aimed at reducing the spread and establishment of the zebra and quagga mussel throughout the basin. Vectors of model parameter values associated with habitat (H), location (L), and inoculation (I) define the baseline risk, R_B, of mussel establishment in an individual water body. A selected management action will alter the values of some of these parameters and produce correspondingly modified parameter vectors for habitat (H') and inoculation (I'). It is assumed that the location parameters will remain constant. Using the modified parameter vectors, the model will estimate the risk of invasion and establishment in relation to the management action, R_M. The effectiveness of the management action can be defined in terms of risk reduction, R_B–R_M. Risk reductions can be estimated for several alternative management actions to identify the most effective management actions for specific water bodies within the basin.

The risk-based decision model for zebra mussel establishment served as a key component in the development of an operational framework (Figure 30) for assessing and managing the potential impacts of invasive mussels on populations of endangered unionid mussels as well as for evaluating the ecological, infrastructure, and economic consequences of mussel establishment.

Recommendations

The results of this initial study identified several key actions required to refine and fully implement the risk-based decision model for dreissenids in the St. Croix Basin. These recommendations focus on the water quality data underlying the habitat suitability calculations and the yet to be implemented inoculation component of the model. The primary data issues concern (1) continued searches for additional data, (2) data quality, and (3) data base development/management. In developing the initial database of water quality parameters, it became increasingly evident that the number of parameters, number of samples, and frequency/duration of sample collection varied substantially from water body to water body. Different sources of data (e.g., states, federal) were characterized by different sampling methods, sample frequency, reported units, duration of sampling, and number of water bodies for which data have been collected. These variations were addressed as much as possible in developing the St. Croix Basin database used in the initial assessment of the 77 surface waters. However, complete standardization was not possible. For example, risk characterization of a lake might be based on values for all 14 water quality parameters, while other lakes might only have data for three or four of these parameters. Additionally, data might be available at weekly intervals for several years for some systems, while others were sampled sporadically for a much shorter duration. It was possible to describe the data limitations and infer how these limitations can influence the resulting risk estimates. However, the challenge remains to construct a standardized water quality database of required quality and sufficiency using primary data sources that were not specifically designed to assess mussel establishment and spread.

In further developing the database, decisions concerning the temporal scale of the assessment must be made. For example, the preliminary results demonstrated the differences in risk estimates based on annual average versus interpolated daily values of selected water quality parameters (e.g., DO) for individual basin lakes. Different temporal scales of analysis can be justified: (1) annual averages provide a general description of the suitability of a water body for mussel establishment, and (2) daily values might be necessary to assess habitat quality for critical periods in the zebra mussel life cycle (e.g., mussel reproduction, veliger stage, veliger attachment to substrate). Monthly values might be appropriate for some factors used in the risk assessment, for example, overwintering temperature and DO. Decisions regarding the appropriate scaling of the assessment are needed to guide refinement of the St. Croix Basin database.

Analysis of correlations among the water quality data suggest that alkalinity, hardness, and calcium concentrations are highly correlated and introduce redundancy to the assessment of habitat quality. It is recommended that calcium be used as the key habitat factor an that alkalinity or hardness can be used to estimate calcium concentration for surface waters lacking calcium data. Otherwise, the weak nature of the correlations among the remaining habitat factors indicate that data should be obtained for as many of these factors as possible for individual water bodies included in the Basin.

The inoculation component of the risk-based decision model is essential. Available management actions will likely focus on the factors that influence inoculation (e.g., angling, recreational boating, commercial navigation). Note that it might prove possible to reduce habitat quality for invasive mussels in some instances, yet it appears unlikely that system-level modification of water quality parameters will prove pragmatic as a standard invasive mussel management action. Several tasks are underway to make the inoculation component of the decision model operational:

- The results of the 1989–1990 Wisconsin Department of Natural Resources boating survey (i.e., Penaloza 1991) have been obtained in raw format (Ed Nelson, Wisconsin DNR, Madison, personal communication). These data are being evaluated by E2 to determine the intensity of water body use within the Wisconsin counties that are included in the St. Croix Basin.

- Searches are underway to identify recreational boating data for Minnesota water bodies included in the St. Croix Basin.

- The National Park Service is being contacted as a potential source of recreational use data for the Upper St. Croix Scenic River.
- The gravity modeling of recreational boating (e.g., Bossenbroek et al. 2001) used to estimate spread of zebra mussels at larger scales is being evaluated to determine if the approach can be applied at a finer scale of resolution within the Basin.

The final recommendation concerns the implementation of the Comprehensive Aquatic Systems Model (CASM) for selected water bodies classified as high-risk locations for dreissenid mussel inoculation and establishment. The CASM is a bioenergetics-based model that describes the daily production dynamics of populations of producers and consumers in specified aquatic food webs in relation to varying environmental conditions (e.g., light, temperature, nutrients, and sediment loading). This model, developed by Bartell, has been applied previously to assess ecological risks posed by physical, chemical, and biological stressors in a variety of aquatic ecosystems (Naito et al. 2002, 2003; Bartell et al. 1999, 2000; Miyamoto et al. 1998; DeAngelis et al. 1989). The CASM can be used to examine the implications of varying environmental conditions, trophic interactions, and competitive interactions on the likelihood of dreissenid establishment for different rates of inoculation. The CASM can also be used to forecast the impacts of successful dreissenid mussel establishment on subsequent production dynamics and alterations in food web structure. Finally, the model can be used to examine the effectiveness of management actions that: (1) alter water quality parameters included in the model, (2) reduce rates of inoculation, or combine both approaches to invasive mussel management.

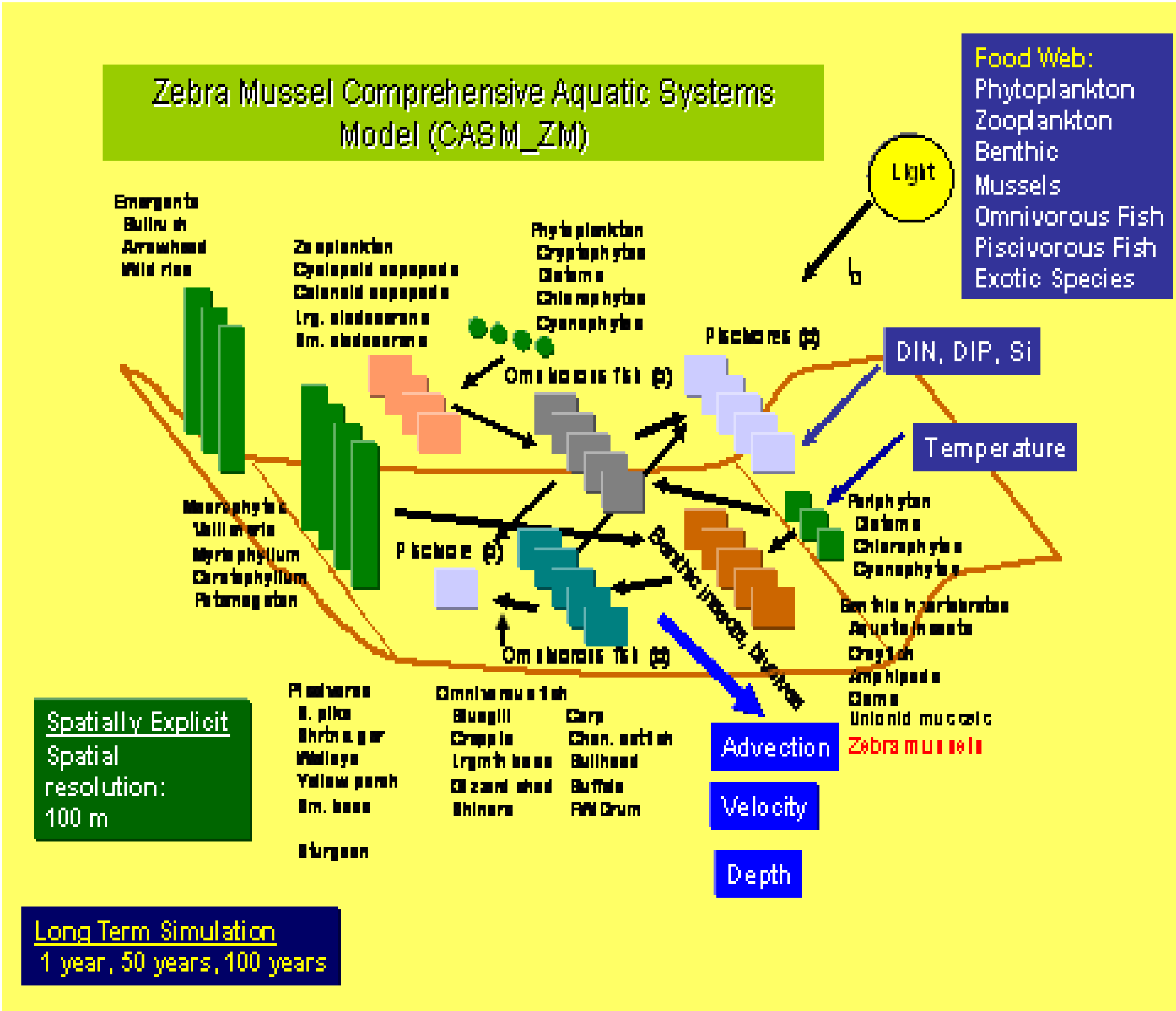

Figure 31. CASM adapted to assess zebra mussel establishment and subsequent impacts on system structure and function.

References

Ackerman, J.D.; Sim, B.; Nichols, S.J.; and Claudi, R. 1994. A review of the early life history of zebra mussels (*Dreisssena polymorpha*): Comparisons with marine bivalves. Canadian Journal Zoology 72:1169–1179.

Baldwin, B.S.; Mayer, M.S.; Dayton, J.; Pau, N.; Mendilla, J.; Sullivan, M.; Moore, A.; Ma, A.; and Mills, E.L. 2002. Comparative growth and feeding in zebra and quagga mussels (*Dreissena polymorpha* and *Dreissena bugensis*): implications for North American lakes. Canadian Journal of Fisheries and Aquatic Sciences 59:680–694.

Bartell, S.M.; Campbell, K.R.; Lovelock, C.M.; Nair, S.K.; and Shaw, J.L. 2000. Characterizing aquatic ecological risks from pesticides using a diquat dibromide case study III. Ecological process models. Environmental Toxicology and Chemistry 19:1441–1453.

Bartell, S.M.; Lefebvre, G.; Kaminski, G.; Carreau, M.; and Campbell, K.R. 1999. An ecosystem model for assessing ecological risks in Quebec rivers, lakes, and reservoirs. Ecological Modelling 124:43–67.

Bartell, S.M., R.H. Gardner, and R.V. O'Neill. 1992. Ecological Risk Estimation. Lewis Publishers, Inc., Chelsea, Michigan. 233 p.

Bemy, P.J.; Veniat, A.; and Mazallon, M. 2003. Bioaccumulation of lead, cadmium, and lindane in zebra mussels (*Dreissena polymorpha*) and associated risk for bioconcentration in tufted duck (*Aythia fuligula*). Bulletin of Environmental Contamination and Toxicology 71:90-97.

Bobeldyk, A.M.; Bossenbroek, J.M.; Evans-White, M.A.; Lodge, D.M.; and Lamberti, G.A. 2005. Secondary spread of zebra mussels (*Dreissena polymorpha*) in coupled lake-stream systems. EcoScience 12:339−346.

Borcherding, J. 1992. Morphometric changes in relation to the annual reproductive cycle. in *Dreissena Polymorpha*—a prerequisite for biomonitoring studies with zebra mussels. pp. 87−99. In D. Neumann and H.A. Jenner (eds.), The zebra mussel *Dreissena Polymorpha:* ecology, biological monitoring and first applications in the water quality management. VCH Publishers, Deerfield Beach, Flordia. pp. 262.

Bossenbroek, J.M.; Kraft, C.E.; and Nekola, J.C. 2001. Prediction of long-distance dispersal using gravity models: zebra mussel invasion of inland lakes. Ecological Applications 11:1778−1788.

Burky, A.J. 1983. Physiological ecology of freshwater bivalves. . In W.D. Russell-Hunter, (ed.), The Mollusca. Vol. 6, Ecology, 281−327. Academic Press, New York, New York.
Caraco, N.F.; Cole, J.J.; Findlay, S.E.G.; Fischer, D.T.; Lampman, G.G.; Pace M.L.; Strayer,

D.L. 2000. Dissolved oxygen declines in the Hudson River associated with the invasion of the Zebra Mussel (*Dreissena polymorpha*). Environmental Science and Technology 34:1204–1210.

Carlton, J.T. 1993. Dispersal mechanisms of the zebra mussel *Deissena*. In: *Zebra Mussels:Biology, Impacts, and Control*, Nalepa, T.F. and D.W. Schloesser (eds.). Lewis Publishers, Boca Raton, Flordia. pp 677–698.

Cawley, E.T. 1996. A compendium of reports of mussel studies containing *Lampsilis higginsii* from the period 1980–1996. Report for the Higgins Eye Recovery Team – Fish and Wildlife Service. Environmental Research Center–Loras College, Dubuque, Iowa. pp. 84.

Clarke, A.H., Jr.; and Loter, J.C. 1995. The 1994 mussel monitoring program in the east branch of the Mississippi River at Prairie du Chien, Wisconsin. A report for DeWitt, Porter, and Co. for Didion, Inc.

Claudi, R.; and Mackie, G.L. 1994. Practical manual for zebra mussel monitoring and control. Journal of the North American Benthological Society 13(3):411−412.

Cohen, A.N.; and Weinstein, A. 2001. Zebra Mussel's Calcium Threshold and Implications for its Potential Distribution in North America. San Francisco Estuary Institute, San Francisco, California. pp. 47. http://www.dfg.ca.gov/quaggamussel/docs/2001-Zebramusselcalcium.pdf.

Cohen, A.N.; and Weinstein, A. 1998a. *The Potential Distribution and Abundance of Zebra Mussels in California*. San Francisco Estuary Institute, Richmond California.

Cohen, A.N.; and Weinstein, A. 1998b. *Methods and Data for Analysis of Potential Distribution and Abundance of Zebra Mussels in California*. San Francisco Estuary Institute, Richmond California.

Committee on Environment and Natural Resources. 1999. *Ecological Risk Assessment in the Federal Government*. CENR/5-99/001, National Science and Technology Council, Washington, DC. pp. 219. http://oaspub.epa.gov/eims/eimscomm.getfile?p_download_id=36384.

Davis, M.; and Hart, R. 1995. Mussel habitat in the Richmond Island/Lock and Dam 6 Tailwater area of Pool 7, Mississippi River and its importance for recovery of the federally endangered mussel, *Lampsilis higginsii*. Ecological Services Section, Minnesota Department of Natural Resources. pp. 34.

DeAngelis, D.L.; Bartell, S.M.; and Brenkert, A.L. 1989. Effects of nutrient recycling and food-chain length on resilience. American Naturalist 134(5):778–805.

de Kock, W.; and Bowner, C. 1993. Bioaccumulation, biological effects, and food chain transfer of contaminants in the zebra mussel (*Dreissena polymorpha*). In T. Nalepa and D. Schloesser (eds.), Zebra mussels: biology, impacts, and control, Lewis Publishers, Ann Arbor, Michigan.

Drake, J.M.; and Bossenbroek, J.M. 2004. The potential distribution of zebra mussels in the United States. BioScience 54:931−941.

Eaton, A.D.; Clesceri, L.S.; and Greenberg, A.E. (eds.). 1995. Standard Methods for the Examination of Water and Wastewater. 19th Edition. American Public Health Association, American Water Works Association and Water Environment Federation, Washington, DC. pp. 1,541.

Effler, S.W.; Brooks, C.M.; Whitehead, K.A.; Wagner, B.A.; Doerr, S.M.; Perkins, M.G.; Siegfried, C.A.; Walrath, L.; and Canale, R.P. 1996. Impact of zebra mussel invasion on river water quality. Water Environmental Research 68:205.

Effler, S.W.; and Siegfried, C. 1998. Tributary water quality feedback from the spread of zebra mussels: Oswego River, New York. Journal of Great Lakes Research 24(2):453–463.

Engineering Research and Development Center. 1994. Review of zebra mussel control methods. Zebra Mussel Research Technical Note ZMR-2-14. U.S Army Corps of Engineers, Vicksburg, Mississippi. pp. 12. http://www.sgnis.org/publicat/papers/zmr_2_14.pdf.

Electric Power Research Institute (EPRI). 1992. *Zebra Mussel Monitoring and Control Guide*. TR-101782. EPRI Institute, Palo Alto, California. pp 2.2−6.39.

Fahnenstiel, G.L.; Lang, G.A.; Nalepa, T.F.; and Johengen, T.H. 1995. Effects of zebra mussel (*Dreissena polymorpha)* colonization on water quality parameters in Saginaw Bay, Lake Huron. Journal of Great Lakes Research 21:435–448.

Fanslow, D.L.; Nalepa, T.F.; and Lan, G.A. 1995. Filtration Rates of the Zebra Mussel *(Dreissena polymorpha)* on Natural Seston from Saginaw Bay, Lake Huron. Journal of Great Lakes Research 21(4):489–500.

Gordon, R. 2002. *Lampsilis higginsii* recovery project. U.S. Fish and Wildlife Service, Genoa National Fish Hatchery, Genoa, Wisconsin. pp. 7.

Griffiths, R.W.; Schloesser, D.W.; Leach, J.H.; and Kovolak, W.P. 1991. Distribution and dispersal of the zebra mussel (*Dreissena polymorpha*) in the Great Lakes region. Canadian Journal of Fisheries and. Aquatic Science 48:1381–1388.

Haag, W.R.; and Garton, D.W. 1992. Synchronous spawning in a recently established populations of the zebra mussel (*Dreissena polymorpha*) in western Lake Erie, USA. Hydrobiologia 234:103−110.

Heath, D.J. 2002. Report on placement of juvenile *L. higginsii* into the lower Black River, Wisconsin, during 2001. Wisconsin Department of Natural Resources, La Crosse, Wisconsin. pp. 2.

Heath, D.J. 2001. Report on placement of *L. higginsii*, inoculated fish in cages, and freeranging fish, lower Wisconsin River, 2001. Wisconsin Department of Natural Resources, La Crosse, Wisconsin. pp. 4.

Hebert, P.D.N.; Manchester, B.W.; and Mackie, G.L. 1989. Ecological and genetic studies on Dreissena polymorpha (Pallas): a new mollusk in the Great Lakes. Canadian. Journal of Fisheries and Aquatic Science 46(9):1587–1591.

Hunter, R.D.; and Bailey, J.E. 1992. *Dreissena polymorpha* (zebra mussel): Colonization of soft substrata and some effects on unionid bivalves. The Nautilus 106:60–67.

Indiana Department of Natural Resources. 2006. http://www.in.gov/dnr/invasivespecies/.

Jantz, B.; and Neumann, D. 1998. Growth and reproductive cycle of the zebra mussel in the River Rhine as studied in a river bypass. Oecologia 114(2):213−225.

Johnson, L. 1997. Westward expansion of the zebra mussel: will the spread be slowed by dispersal and demographic limitations? Paper presented at American Fisheries Society, 127th Annual Meeting, Aug. 24−28, 1997, Monterey, California.

Johnson, L.E.; and Carlton, J.T. 1996. Post-establishment spread in large-scale invasions: dispersal mechanisms of the zebra mussel *Dreissena polymorpha*. Ecology 77:1686-1690.

Johnson, L.E.; Ricciardi, A.; and Carlton, J.T. 2001. Overland dispersal of aquatic invasive species: a risk assessment of transient recreational boating. Ecological Applications 11:1789-1799.

Karatayev, A.; Burlakova, L.; and Padilla, D. 1997. The effects of *Dreissena polymorpha* (Pallas) invasion on aquatic communities in Eastern Europe. Journal of Shellfish Research 16:187−203.

Kennedy, R.H., and Açkakaya, H.R. 1999. Development of an age-structured metapopulation model of zebra mussels. In *Zebra Mussel Technical Notes Collection*(ZMR-3-21), U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi. pp. 6. http://el.erdc.usace.army.mil/elpubs/pdf/zmr3-21.pdf.

Kjos, C., O. Byers, P. Miller, J. Borovansky and U.S. Seals (eds.). 1998. Population and Habitat Viability Assessment Workshop for the Winged Mapleleaf (*Quadrula fragosa*): Final Report. CBSG, Apple Valley, MN.

Kovalak, W.P.; Longton, G.D.; and Smithee, R.D. 1993. Infestation of power plant water systems by the zebra mussel (*Dreissena polymorpha* Pallas). In Nalepa and Schloesser (eds), *Zebra mussels: Biology, Impacts, and Control.* Lewis Publishers, CRC Press, Inc., Boca Raton, Florida. pp. 359–380.

Lange, C.L.; and Wittmeyer, S. 1966. The contribution of zebra mussel (Dreissena spp.) faeces and pseudofaeces production to taste and odor episodes in the Niagara River and Lake Erie. Accepted for presentation at the Sixth International Zebra Mussel and Other Nuisance Organisms Conference, Dearborn, Michigan.

Leach, J.H. 1993. Impacts of the zebra mussel, *Dreissena polymorpha,* on water quality and spawning reefs in western Lake Erie. In Nalepa and Schloesser (eds), *Zebra mussels: Biology, Impacts, and Control.* Lewis Publishers, CRC Press, Inc., Boca Raton, Florida. pp.381–398.

MacIsaac, H.J.; Sprules, W.G.; Johannsson, O.E.; and Leach, J.H. 1992. Filtering impacts of larval and sessile zebra mussels *(Dreissena polymorpha)* in western Lake Erie. Oecologia 92:30–39.

MacIsaac, H.J.; Sprules, W.G.; Johannsson, O.E.; and Leach, J.H. 1991. Ingestion of small-bodied zooplankton by zebra mussels *(Dreissena polymorpha):* Can cannibalism on larvae influence population dynamics? Canadian Journal of Fisheries and Aquatic Science 48:2051–2060.

Mackie, G.L. 1993. Biology of the zebra mussel (*Dreissena polymorpha*) and observations of mussel colonization on unionid bivalves in Lake St. Clair of the Great Lakes. In: *Zebra Mussels: Biology, Impacts, and Control*, Nalepa, T.F. and D.W. Schloesser (eds.), Lewis Publishers, Boca Raton, Florida. pp. 153–166.

Mackie, G.L. 1991. Biology of exotic zebra mussel, Dreissena polymorpha, in relation to native bivalves and its potential impact on Lake St. Clair. Hydrobiologia 219(1):251−268**.**

Mackie, G.L.; Gibbons, W.N.; Muncaster, B.W.; and Gray, I.M. 1989. The zebra mussel, Dreissena polymorpha: a synthesis of European experiences and a preview for North America. ISBN:0-772905647-2, Great Lakes Section, Water Resources Branch, Ontario Ministry of the Environment, London and Ontario.

Mackie, G.L.; and Schloesser, D.W. 1996. Comparative biology of zebra mussels in Europe and North America: An overview. American Limnology.36:244–258.

Martel, A. 1993. Dispersal and recruitment of zebra mussel (*Dreissena polymorpha*) in a nearshore area in west central Lake Erie: the significance of postmetamorphic drifting. Canadian Journal of Fisheries and Aquatic Science 50:3–12.

McMahon, R. 1996. The physiological ecology of the zebra mussel, *Dreissena polymorpha*, in North America and Europe. American Zoologist 36:339–363.

McMahon, R.F.; Ussery, T.A.; Miller, A.C.; and Payne, B.S. 1993. Thermal tolerance in zebra mussels *(Dreissena polymorpha)* relative to rate of temperature increase and acclimation temperature. In J. L. Tsou and Y. G. Mussalli (eds.), Proceedings: Third international zebra mussel conference. EPRI TR-102077, Electric Power Research Institute, Palo Alto, California. pp. 4-97−4-118.

Miller, A.C.; and Payne, B.S. 2007. A Summary of Distribution and Abundance Data on the Endangered Higgins Eye Pearlymussel, *Lampsilis higginsii* (Lea, 1857) Environmental Laboratory, U.S. Army Engineer Research and Development Center, Vicksburg, MS 39180-6199

Miller, A.C.; Payne, B.S.; and McMahon, R.F. 1992. Zebra Mussel: Biology, Ecology, and Recommended Control Strategies, Technical Notes, ZMR-1-01, Zebra Mussel Research, U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, Mississippi. pp. 9. http://el.erdc.usace.army.mil/elpubs/pdf/zmr1-01.pdf.

Mills, E.L.; Chrisman, J.R.; Baldwin, B.; Owens, R.W.; O'Gorman, R.; Howell, T.; Roseman, E.F.; and Raths, M.K. 1999. Changes in the Dressenid Community in the Lower Great Lakes with Emphasis on Southern Lake Ontario. Journal of Great Lakes Research 25(1):187–197.

Miyamoto, K.; Naito, W.; Nakanishi, J.: and Bartell, S.M. 1998. Ecological risk analysis in Japanese lake: a case study using the CASM. In: Proceedings of the 1st International Workshop on Risk Evaluation and Management of Chemicals, Yokohama National University, Yokohama, Japan, January 22–23, 1998. pp. 111–118.

Molloy, D.P. 2002. Evaluation of biotic and treatment factors relating to bacterial control of zebra mussels. U.S. Department of Energy Technical Report 40751R04. pp. 6.

Naito, W.; Miyamoto, K; Nakanishi, J; Masunaga, S; Bartell, S.M. 2003. Evaluation of an ecosystem model in ecological risk assessment of chemicals. Chemosphere 53(4):363–75.

Naito, W.; Miyamoto, K; Nakanish, J; Masunaga, S; Bartell, S.M. 2002. Application of an ecosystem model for aquatic ecological risk assessment of chemicals for a Japanese lake. Water Resources 36(1):1–14.

Nalepa, T.F.; Fahnenstiel, G.L.; McCormick, M.J.; Cavaletto, J.F.; Fanslow, D.; Ford, M.; Gordon, W.M.; Goudy, G.; Johengen, T.; Jude, D.; Lang, G.A.; and Wojcik, J.A. 1993. Physical and chemical variables of Saginaw Bay, Lake Huron in 1991−93. NOAA Technical Memorandum ERL GLERL-91, Great Lakes Environmental Research Laboratory, Ann Arbor, Michigan. (NTIS# PB96-182357/XAB). pp. 78.
ftp://ftp.glerl.noaa.gov/publications/tech_reports/glerl-091/tm-091.pdf
New York Sea Grant Extension Fact Sheet 1994.
http://nis.gsmfc.org/nis_factsheet.php?toc_id=131.

Nichols, S.J. 1996. Variations in the reproductive cycle of *Dreissena polymorpha* in Europe, Russia, and North America. American Zoologist 36:311–325.

Nichols, S.J. 1992. Maintenance of the zebra mussel (Dreissena polymorpha) under laboratory conditions. In Zebra Mussels: Biology, Impacts, and Control. T.F. Nalepa and D. W. Schloesser (eds.). Lewis Publishers, Boca Raton, Florida. pp315−332.

Odum, E. P. 1971. Fundamentals of ecology. 3rd edition. W. B. Saunders Co., Philadelphia and London. pp. 544.

Office of Technology Assessment. 1993. Harmful non-indigenous species in the United States. Publication OTA-F-566, United States Congress, Washington, DC. pp. 409.

Ohio Sea Grant. 1995. Sea Grant Zebra Mussel Update: A 1995 Report of Ohio Sea Grant. http://sgnis.org/publicat/ohsg1995.htm.

Ohio Sea Grant. 1994. Zebra Mussels in North America: The invasion and its implications. Fact Sheet 045,OHSU-FS-045. Ohio Sea Grant College Program, the Ohio State University, Columbus, Ohio 43212-1194. pp. 4.

O'Neill, C.R., Jr. 1996. The Zebra Mussel: Impacts and Control. New York Sea Grant, Cornell University, Ithaca, New York. Cornell Cooperative Extension Information Bulletin No. 238.

Padilla, D.K.; Chotkowski, M.A.; and Bucahn L.A.J. 1996. Predicting the spread of zebra mussels (*Dreissena polymorpha*) to inland waters using boater movement patterns. Global Ecology and Biogeography Letters 5:353−359.

Payne, B.S.; and Miller, A.C. 2004. A probability tree applied to a common zebra mussel dispersal issue, ANSRP Technical Notes Collection (ERDC/TN ANSRP-04-1), U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi. pp. 7. http://el.erdc.usace.army.mil/elpubs/pdf/ansrp04-1.pdf.

Penaloza, L.J. 1991. Boating pressure on Wisconsin's lakes and rivers. Results of the 1989−1990 Wisconsin Recreational Boating Study, Phase I. Technical Bulletin No. 174. Wisconsin Department of Natural Resources, Madison, Wisconsin.

Preisendorfer, R.W. 1986. Secchi disk science: Visual optics of natural waters. Limnology Oceanography 31:909–926.

Ram., J.L.; and Walker, J.U. 1993. Effects of deionzed water on viability of the zebra mussel, *Dreissena polymorpha*. Comparative Biochemistry Physiology 105C(3):409–414.

Ramcharan, C.W.; Padilla, D.K.; and Dodson, S.I. 1992a. Models to predict potential occurrence and density of the zebra mussel, *Dreissena polymorpha*. Canadian Journal of Fisheries and Aquatic Science 49(12):2611–2620.

Ramcharan, C.W.; Padilla, D.K.; and Dodson, S.I. 1992b. A multivariate model for predicting population fluctuations of Dreissena polymorpha in North America lakes. Canadian Journal of Fisheries and Aquatic Sciences 49:150.

Reeders, H.H.; Bij de Vaate, A.; and Noordhuis, R. 1993. Potential of the zebra mussel *(Dreissena polymorpha)* for water quality management. In T.F. Nalepa and D.W. Schloesser (eds.), *Zebra mussels.* Biology, impacts, and control. Lewis Publishers, Boca Raton, Florida. pp. 439–451.

Schloesser, D.W.; and Kovalak, W. 1991. Infestation of unionids by *Dreissena polymorpha* in a power plant canal in Lake Erie. Journal of Shellfish Research 10(2):355–359.

Schloesser, D.W.; Kovalak, W.P.; Longton, G.D.; Ohnesorg, K.L.; and Smithee, R.D. 1998. Impact of Zebra and Quagga Mussels (Dreissena spp.) on Freshwater Unionids (Bivalvia: Unionidae) in the Detroit River of the Great Lakes. American Midland Naturalist 140:299–313.

Schneider, D.W.; Ellis, C.D.; and Cummings, K.S. 1998. A transportation model assessment of the risk to native mussel communities from zebra mussel spread. Conservation Biology 12:788−800.

Setzler-Hamilton, E.M.; Wright, D.A.; and Magee, J.A. 1997. Growth and spawning of laboratory-reared zebra mussels in lower mesohaline salinities. pp 141–154. In F.M. D'Itri (ed.), Zebra Mussels and Aquatic Nuisance Species. Lewis Publishers, CRC Press, Boca Raton, Florida. pp 638.

Smirnova, N.F.; and Vinogradov, G.A. 1990. Biology and ecology of *Dreissena polymorpha* from the European USSR. Presented at the workshop on Introduced Species in the Great Lakes: Ecology and Management. Saginaw, Michigan, September 26–28.

Sorba, E.A.; and Williamson, D.A. 1997. Zebra Mussel Colonization Potential in Manitoba, Canada. Water Quality Management Section, Manitoba Environment, Report No. 97–07.

Spada, M. E. 2000. Limitations on zebra mussels (*Dreissena polymorpha*) in a hypereutrophic system, Onondaga Lake, New York. M.S. Thesis. Department of Environmental and Forest Biology, State University of New York, College of Environmental Science and Forestry, Syracuse, New York. pp. 69.

Sprung, M. 1993. The other life: an account of present knowledge of the larval phase of Dreissena polymorpha. In: T.F. Nalepa, and D.W. Schloesser (eds.), Zebra Mussels: Biology, Impacts, and Control Lewis Publishers, Boca Raton, Florida. pp 39–53.

Sprung, M.; and Rose, U. 1988. Influence of food size and food quality on the feeding of the mussel *Dreissena polymorpha.* Oecologia 77:526–532.

Stoeckel, J.A.; Rehman, C.R.; Scneider, D.W. and Padilla, D.K. 2004. Retention and supply of zebra mussel larvae in a large river system: importance of an upstream lake. Freshwater Biology 49:919−903.

Strayer, D.L.; and Malcom, H.M. 2006. Long-term demography of a zebra mussel (Dreissena polymorpha) population. Freshwater Biology 51:117–130.

Strayer, D.L.; and Smith, L. 1993. Distribution of the zebra mussel in estuaries and brackish waters. In T.F. Nalepa and D.W. Schloesser, (eds.), Zebra mussels: biology, impacts, and control. Lewis Publishers, Boca Raton, Florida. pp. 715−727.

Thiel, P.A. 1981. A survey of unionid mussels in the upper Mississippi River (Pools 3−11). Wisconsin Department of Natural Resources Technical Bulletin No. 124. pp. 24.

Tucker, J.K. 1994. Colonization of unionid bivalves by the zebra mussel, *Dreissena polymorpha,* in pool 26 of the Mississippi River. Journal of Freshwater Ecology 9:129–134.

Tucker, J.K.; Theiling, C.H.; Blodgett, K.D.; and Theil, P.A. 1993. Initial occurrence of zebra mussels *(Dreissena polymorpha)* on freshwater mussels (Family Unionidae) in the upper Mississippi River system. Journal of Freshwater Ecology 8:245–251.

U.S. Army Corps of Engineers. 2003. Mississippi River between the Missouri River and Minneapolis, 9-foot channel project. Measures for managing zebra mussels. Reconnaissance report. USACE, St. Paul District, St. Paul, Minnesota. pp. 35. www.fws.gov/midwest/mussel/documents/recon_report_for_**zebra_mussel**_control_august_2003.pdf.

U.S. Fish and Wildlife Service. 2004. Higgins Eye Pearly Mussel (*Lampsilis higginsii*) Recovery Plan: First Revision. Ft. Snelling, Minnesota. USGS news release September 18, 1997. Zebra mussels are spreading rapidly, USGS report. pp 126.

U.S. Fish and Wildlife Service. 2001. Endangered and Threatened Wildlife and Plants; Establishment of Nonessential Experimental Population Status for 16 Freshwater Mussels and 1 Freshwater Snail (Anthony's Rivers snail) in the Free-Flowing Reach of the Tennessee River below the Wilson Dam, Colbert and Lauderdale Counties, AL. Federal Register 66(115):32250–32264.

U.S. Fish and Wildlife Service. 1997. Winged Mapleleaf Mussel (*Quadrula fragosa*) Recovery Plan, Region 3. Minneapolis, Minnesota. pp. 359.

U.S. Fish and Wildlife Service. 1991. Upper Mississippi River navigation studies, planning aid report. U.S. Fish and Wildlife Service, Rock Island, Illinois. pp 61.

U.S. Geological Survey. News release September 18, 1997. Zebra mussels are spreading rapidly, USGS report. http://www.usgs.gov/newsroom/article.asp?ID=881.

Vanderploeg, H. 2002. The Zebra Mussel Connection: Nuisance Algal Blooms, Lake Erie Anoxia, and other Water Quality Problems of the Great Lakes. National Oceanic and Atmospheric Administration, Great Lakes Environmental Research Laboratory. http://www.glerl.noaa.gov/pubs/brochures/mcystisflyer/mcystis.pdf.

Vinogradov, G.; Smirnova, S.; Sokolov, V.; and Bruznitsky, A. 1993. Influence of chemical composition of the water on the mollusk *Dreissena polymorpha*. In T.F. Nalepa and D.W. Schloesser, (eds.), Zebra mussels: biology, impacts, and control. Lewis Publishers, Ann Arbor, Michigan. pp. 810.

Walz, N. 1978. The energy balance of the freshwater mussel *Dreissena polymorpha* in the laboratory and in Lake Constance, Switzerland. Part 3. Growth under standard conditions. Archives of Hydrobiology Supplemental 55:121–141.

Wilcox, S.J.; and Dietz, T.H. 1995. Potassium transport in the freshwater bivalve *Dreissena polymorpha*. Journal of Experimental Biology. 198:861–868.

Wildridge, P.J.; Werner, R.G.; Doherty, F.G.; and Neuhauser, E.F. 1998. Acute effects of potassium on filtration rates of adult zebra mussels (*Dreissena polymorpha)*. Journal of Great Lakes Research 24(3):629−636.

Wu, Y., Bartell, S.M. and Nair, S.K. 2006. A Spatial Model for Restoration of the Upper-Mississippi River Ecosystems. SPIE Optics and Photonics. Proceedings of Remote Sensing and Modeling of Ecosystems for Sustainability III, Volume 6298: 147−162.