A Review of the Science and Management of Eurasian Watermilfoil: Recommendations for Future Action in New York State

Holly Menninger, PhD

Coordinator, New York Invasive Species Research Institute

Cornell University

August 8, 2011

Revised: November 11, 2011

Table of Contents

Executive Summary 3

Scope of This Report 6

Distribution & History of Eurasian Watermilfoil Invasion 6

- North America 6
- New York State 7

Biology and Ecology 8

Eurasian Watermilfoil Impacts 10

- Ecological impacts 10
- Economic Impacts 11

Management Approaches, Outcomes, and Limitations 13

- Physical management practices 13
- Mechanical control 15
- Chemical control 16
  - Contact herbicides 17
  - Systemic herbicides 17
  - Herbicide resistance 19
- Biological control 20
  - Euhrychiopsis lecontei 21
  - Acentria ephemerella 23
  - Cricotopus myriophylli 25
  - Grass Carp 25
  - Pathogens 27
- Management research needs 27

Spread Prevention 28

- Recreational boating as a transport vector 28
- Viability of transported fragments 30
- Efficacy of cleaning practices 31
- Spread prevention at the landscape scale: “Smart prevention” 31
- Offense or defense? 36

Moving Forward: Recommendations for New York State 36

Literature Cited 38

Executive Summary

Key findings and recommendations:

- EWM continues to infest and spread through the waters of New York State, despite significant expenditures for management and control.
- At best, physical, chemical, and mechanical management approaches reduce EWM over the short-term (up to a few years) and require a sustained and long-term commitment for maintenance control.
- Biological control holds promise for long-term EWM suppression, but research on the effectiveness of commercial weevil augmentation practices has rarely been evaluated and published in the peer-reviewed scientific literature.
- Research evaluating the outcomes of EWM management often lacks the scientific rigor to assess the effects of multiple interventions applied over time or to understand the long-term dynamics of EWM populations left unmanaged.
- Rather than continue the cycle of investment in ineffective maintenance control, New York State should prioritize resources towards the development of a landscape-level spread prevention strategy.

Overview of the Problem

Eurasian watermilfoil (*Myriophyllum spicatum* L., EWM) is one of the most widely distributed aquatic invasive species in the US. First detected in New York State in 1949, less than a decade after its initial collection in North America, EWM quickly spread throughout NY, most likely vectored by boat and trailer movement.

As of January 2011, EWM populations have been confirmed in 353 lakes and rivers in 54 New York State counties. Fifty-four of those infestations occur within the Adirondack Park, including Lake George where EWM was first detected in 1985. Despite significant and long-term management programs, the number of infested sites in NY has continued to grow, with 6 new EWM infestation locations detected in 2010.

Ecological and Economic Impacts

In general, aquatic plants provide important habitat structure for invertebrates and fish. However, in lakes and ponds where EWM dominates, the water bodies support lower abundances and fewer species of macroinvertebrates than waters with native plant-dominated assemblages. Moreover, dense EWM beds have been shown to negatively affect fish abundance, composition and diet. Additionally, EWM outcompetes native aquatic plants for light, space, and nutrients, resulting in reduced native plant diversity and abundance.

Eurasian watermilfoil has significantly affected recreational use of infested lakes and ponds by impeding boating, swimming, and angling as well as generally interfering with the aesthetic quality of infested water bodies. Evidence for the direct economic impacts of EWM is incomplete, but growing; for example, recent research in Wisconsin and Vermont indicates that EWM infestations may reduce lake front property values up to 16%. Control costs to manage or eradicate EWM are often reported for individual projects, but have not been aggregated and summarized for New York State over time.

Management Approaches, Outcomes and Limitations

A variety of management approaches have been employed to eradicate and manage Eurasian watermilfoil. Although the desired outcome is EWM eradication, most efforts result in just short-term (ranging from a few weeks to a few years) decreases in EWM biomass and percent cover. A significant and sustained investment of resources over the long term is required to achieve maintenance control.

Physical techniques – including hand harvesting, benthic barriers, and water drawdowns – have been used extensively to manage Eurasian watermilfoil infestations, particularly in lakes in the Adirondack Park. Physical management techniques can significantly reduce EWM biomass and cover, for as long as 3 years following treatment, but require a long-term commitment to maintenance management. Note that even with significant investments of resources and personnel over the long term (e.g., Upper Saranac Lake, Lake George), EWM has continued to grow back and spread to new areas in these lakes.

Options for chemical control of EWM have increased in recent years as aquatic herbicides have improved in terms of target plant specificity, reduced effective concentrations, and ease of application. Five aquatic herbicides registered in New York State are known to significantly reduce EWM: diquat, endothal, 2,4-D, fluridone and triclopyr. Fluridone (trade names, Sonar® and Avast!®) and triclopyr (trade name, Renovate ®) have been applied at the whole lake scale, and can cause subsequent declines in EWM biomass that may last up to a few years and may result in some recovery of native plant diversity and biomass. Concerns have emerged, however, regarding the potential for herbicide resistance (as has been documented in Florida with fluridone-resistant hydrilla); thus, the rotation of chemical products as part of an integrated pest management approach is strongly encouraged for water bodies with recurring EWM control.

Mechanical harvesting is frequently employed to clear EWM-infested areas around docks, landings, and swimming areas as well as create edge habitat for fish. It provides short-term reductions in EWM biomass (as well as reductions of any other submerged aquatic vegetation intermingled with it), on the order of weeks to months, but may actually exacerbate spread of EWM if cut plant fragments are not completely removed.

In contrast to the above methods, biological control offers the potential for long-term suppression of EWM; it will not, however, eliminate the plant entirely from a water body. To date, two indigenous insects (milfoil weevil, milfoil midge) and one naturalized insect (an aquatic moth) located within EWM's introduced range in North America have been identified and investigated for their potential for biological control. Research suggests that the native milfoil weevil, *Euhrychiopsis lecontei* (Dietz), offers the most promise as a control agent, yet efforts to augment natural weevil populations as a control strategy have produced mixed results and outcomes of this work are rarely reported in the peer-reviewed scientific literature.

Many of the lakes where EWM control actions have been implemented and studied over the past three decades have been subjected to multiple treatments, either at the plot or whole lake scale, over time. And rarely, if at all, do management studies pair treated lakes with untreated but infested control lakes where EWM populations can be monitored over time without intervention. Consequently, it’s difficult to tease apart any interactive, possibly synergistic, effects resulting from repeated EWM management actions or assess what might happen to EWM populations if left unmanaged over time. In fact, a report from Wisconsin indicates that EWM populations left unmangemd may actually decline over time. Recent evidence from studies of other well-established invasive plants in North America (e.g., garlic mustard, Asian stiltgrass) suggest that invasive species are capable of accumulating pathogens as well as herbivores (as observed with EWM) in their non-native habitat, and these acquired enemies might, given time and opportunity, reduce populations sufficiently to permit substantial growth of native vegetation. The question remains if there is willingness on part of the multiple stakeholders so invested in EWM management to take a more scientifically rigorous approach to management efforts that would require us to leave some infestations unmanaged over the long-term.

Shifting focus and resource to spread prevention

Generally, preventing the establishment and spread of invasive species is more effective and economical than eradicating or managing the impacts of an invasive species after it has already established. Yet, with respect to Eurasian watermilfoil, studies evaluating boaters’ launching activity and efficacy of cleaning practices suggest that despite significant outreach and education campaigns, there remains much room for improvement. Moreover, recent research concludes that for species where post-establishment eradication is unlikely, the most effective way to prevent new locations from being colonized is to reduce spread from locations where the invader is well-established (as opposed to spending resources to “eradicate” those large populations).

New York State has the opportunity to take a more strategic approach to the invasion of Eurasian watermilfoil in our waters, one that requires a shift in focus and investment from maintenance control to spread prevention at a landscape scale. This approach would incorporate knowledge of EWM-infested waters (aggregated in the GIS database, iMapInvasives), patterns of boater movement (which to date, has only been surveyed at a few locations), the ecology and life history of EWM, and an understanding of the ecological or human-ascribed values of un-infested water bodies to prioritize action and resource investment.

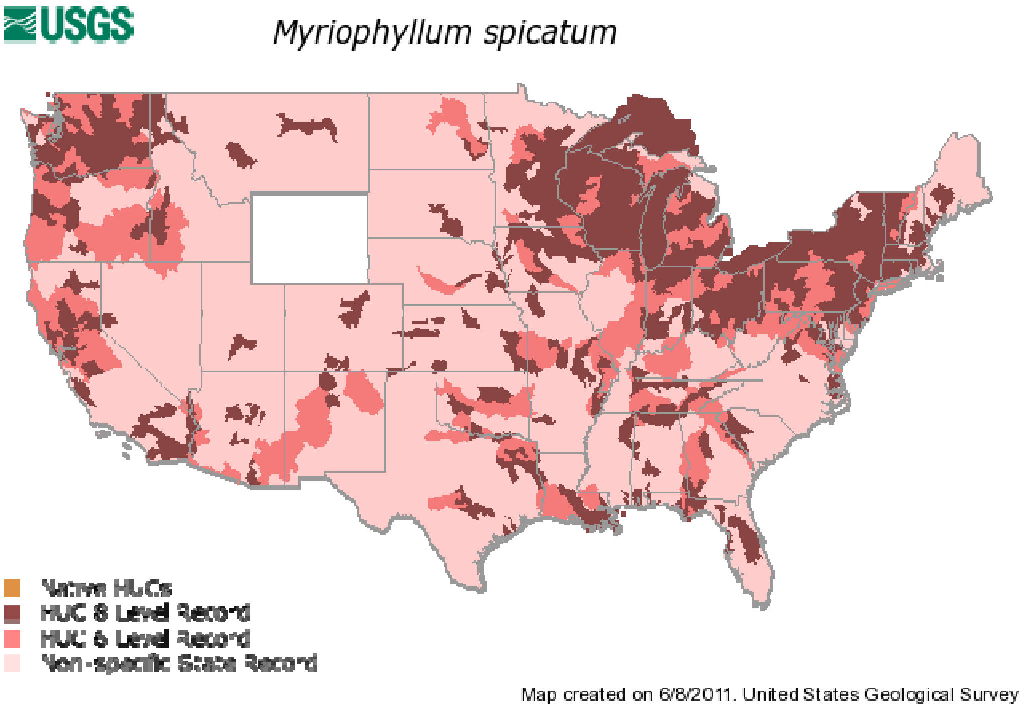
Scope of This Report

Eurasian watermilfoil (*Myriophyllum spicatum* L., EWM), an invasive submerged aquatic plant, has and continues to be the subject of intensive management efforts throughout New York State (NYS). In fulfillment of a request from the NYS DEC Office of Invasive Species Coordination, I review the current state of scientific knowledge regarding the plant's biology, ecology, and impacts. I examine the various management approaches that have been used to control EWM and discuss the outcomes and limitations of these different approaches. Finally, based on my synthesis of the scientific literature, I make recommendations for future research and management actions in NYS.

Distribution & History of Eurasian Watermilfoil Invasion

North America

Native to Europe, Asia and northern Africa, EWM was first collected in North America in 1942 from a pond in Washington, DC (Couch and Nelson 1985). Other specimens were collected from various, widespread locations throughout North America in the following decade, including Arizona, California and Ohio (Couch and Nelson 1985). Today, EWM is considered one of the most widely distributed aquatic invasive species (*Figure 1*), with records of EWM confirmed in 47 US states (Hawaii, Alaska, and Wyoming, being exceptions) as well as British Columbia, Ontario, and Quebec, Canada (Jacono and Richerson 2011, Kartesz 2011)



Map created on 6/8/2011. United States Geological Survey

Figure 1: Distribution of Eurasian watermilfoil in the continental US. Image credit: US Geological Survey

There remains some debate regarding the original source of the introduction of EWM to North America. Some report that the plant was introduced accidentally as shipping ballast in the Chesapeake Bay region in the late nineteenth century (Aiken et al. 1979). Others suggest that EWM was introduced intentionally and perhaps multiple times as an ornamental plant for aquatic gardens or as an aquarium plant (Couch and Nelson 1985). Despite this lack of agreement regarding the initial introduction of EWM, it is widely accepted that recreational boating has been the primary vector responsible for the subsequent spread of EWM to new water bodies (Smith and Barko 1990, Buchan and Padilla 2000, Johnson et al. 2001).

New York State

Eurasian watermilfoil was first observed in New York State in 1949. However, it is quite likely that EWM was introduced multiple times at various locations across the state. The spread of EWM through time tracks closely to New York's river and highway systems, suggesting a significant dispersal role played by boat and trailer transport (Boylen et al. 2006).

EWM was first detected in the Adirondack Park in 1979 in the Chateaugay Lakes (Kelting and Laxson 2010). It continues to be the most commonly observed aquatic invasive plant in the Park, now infesting 54 lakes and ponds (Smith, personal communication). EWM was first observed in Lake George on the southeastern edge of the Adirondack Mountains in 1985 (Madsen et al. 1988), and the lake has subsequently hosted a significant amount of EWM research over the last 25 years.

As of January 2011, populations of EWM have been confirmed in 353 lakes and rivers in 54 counties in New York State (*Figure 2*, Eichler 2010). The number of infested sites has continued to expand, with 6 new sites located in 2010. This may seem like a large number of infested water bodies, but New York State has over 7600 ponded bodies of water, including lakes, ponds, and reservoirs (NYS DEC 2010). Over 3000 of these water bodies are greater than 6.4 acres (minimum size for inclusion in gazetteer), and many are regularly included in state water quality inventories that include monitoring for aquatic plants.

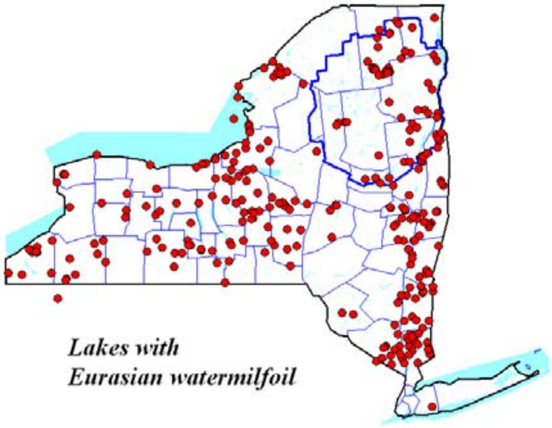


Figure 2. Distribution of EWM in New York State as of December 2010 (Image Credit: Larry Eichler)

Many organizations and programs regularly survey and report new occurrences of Eurasian watermilfoil throughout New York State. These include: NYS DEC Lake Classification and Inventory Survey; the NY Citizens Statewide Lake Assessment Program (CSLAP); other NYS DEC offices including the Bureau of Fisheries; Office of Parks, Recreation and Historic Preservation; NY Natural Heritage Program, local lake associations; private consultants; county planning departments and environmental management councils; soil and water conservation districts; university and botanical garden researchers; and volunteers with Partnerships for Regional Invasive Species Management. Databases of EWM occurrence throughout NYS have been maintained by the Darrin Fresh Water Institute Aquatic Plant Identification Program and the Bureau of Water Assessment and Management in the NYS DEC Division of Water (Eichler, personal communication). Recently, these data have been aggregated and uploaded to NY's all taxa, GIS-based invasive species database, iMapInvasives.org (Dean, personal communication).

Biology and Ecology

Eurasian watermilfoil is a submersed perennial with long branching stems (up to 7m long) that feature finely dissected, feathery leaves arranged in whorls (*Figure 3*, Aiken et al. 1979). Plants can be found growing in waters 1–10m deep (Aiken et al. 1979), but typically are most abundant at depths of 1–4m (Nichols and Shaw 1986). EWM obtains most of its nutrients from lake sediments via roots that arise from lower, buried portions of the stem (Carignan and Kalff 1980). Roots may also arise from upper stems that are preparing to propagate vegetatively by fragmentation (Smith and Barko 1990). The plants produce small spike-like, emergent flowers, only after the stems grow to the surface.

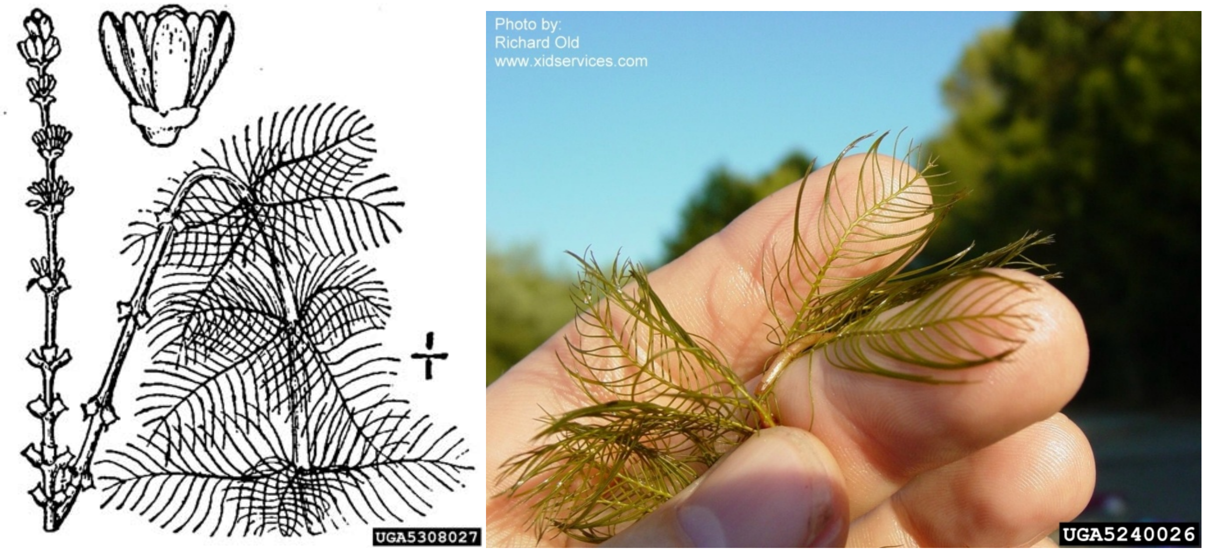


Figure 3. *Left*: Botanical illustration of *Myriophyllum spicatum* flower stalk and feathery leaf whorls (Image Credit: USDA PLANTS Database, USDA NRCS PLANTS Database, Bugwood.org); *Right*: Close-up of EWM leaves (Image Credit: Richard Old, XID Services, Inc., Bugwood.org)

Seeds are produced via sexual reproduction, but it is generally thought that Eurasian watermilfoil primarily reproduces and disperses by asexual, vegetative means (stolon and fragment production) rather than seed (Madsen and Smith 1997). The plant can expand locally via stolons, stems running horizontally from the parent plant across the upper few centimeters of sediment that then give rise to roots and vertical stems. Fragments may be created by the plant itself after flowering through a process called autofragmentation or by mechanical damage from human activities or water turbulence (Smith and Barko 1990). These fragments are then dispersed long-distance among water bodies by water currents, wind, waterfowl, and particularly the movement of boats and trailers (Buchan and Padilla 2000). Autofragments contain a high concentration of stored starches, an adaptation that may promote overwintering success (Madsen 1997). Fragment deposition can show strong seasonal trends, with highest fragmentation reported from Lake George, NY, in late September (Madsen et al. 1988). Also in Lake George, NY, Madsen and Smith (1997) report that 46% of EWM fragments that settled on the lake's sediment established as new plants.

Eurasian watermilfoil plants are evergreen, and shoots from the previous year can overwinter. Unlike other common invasive aquatic plants, EWM does not produce an overwintering storage structure but maintains carbohydrate stores for the next year's growing season in its overwintering roots and shoots (Grace and Wetzel 1978). New shoots are initiated in the autumn, but do not begin to grow until water temperatures warm in the spring (Smith and Barko 1990). Typically, rapid growth begins early in spring when water temperatures approach 15°C. When stems reach the water's surface, they branch extensively, forming an expansive horizontal canopy; often, leaves on the lower parts of the stem will drop off due to the intense shading that results (Lillie and Budd 1992).

Eurasian watermilfoil can tolerate a range of water conditions. It has been reported in waters ranging from highly eutrophic (Aiken et al. 1979) to oligotrophic; however, in low nutrient, oligotrophic lakes like Lake George, infestations of EWM tend to be most common at the mouths of tributary creeks where nutrients and sediments tend to accumulate (Eichler et al. 2001). EWM has a wide pH tolerance (5.4 - 11, Aiken et al. 1979) and can grow across a range of alkalinity (Madsen 1998) and water clarity conditions. Although it is capable of growing in soft water lakes (low alkalinity), recent studies indicate that EWM grows more frequently and abundantly in hard water lakes (Madsen 1998, Buchan and Padilla 2000, Roley and Newman 2008). Generally, the clarity of the water determines the rooting depth and growth form of the plant. In more turbid waters, plants grow at shallower depths and have more horizontal stems at the surface whereas in clearer waters, EWM roots more deeply and may not reach the surface over the course of the growing season (Smith and Barko 1990).

Using molecular genetics tools, Moody & Les (2002) confirmed the existence of three populations of a hybrid between EWM and its close relative, the native northern watermilfoil (*M. sibiricum*), in Wisconsin and Minnesota. These plants featured morphological traits intermediate to both species (particularly leaf-segment number), and anecdotal evidence suggested increased vigor in hybrid plants – e.g., growing in dense, monospecific stands and reproducing aggressively via vegetative propagation. Hybridity could also potentially affect management strategies, particularly biological and chemical control (see *Herbicide Resistance* section below). Northern watermilfoil has a known distribution throughout NY including the Hudson Valley, the Adirondacks, the Finger Lakes region, and western NY (Weldy and Werier 2011). Inter-specific hybrids have been confirmed in the lakes of five northern US states, spanning Michigan to Washington (Moody and Les 2007), but not yet New York State.

Eurasian Watermilfoil Impacts

Ecological impacts

Substantial overwintering biomass and an early start to spring growth gives EWM a competitive advantage over neighboring macrophytes. Dense canopies of EWM stems on the water's surface reduce light for other submerged plant species and the extensive root systems below reduce space and access to nutrients by other plants (*Figure 4*, Madsen et al. 1991). Consequently, significant declines in native vegetation have been observed as a result of EWM infestation including reductions in both native plant species abundance and richness (Madsen et al. 1991, Boylen et al. 1999).

Figure 4. A dense infestation of Eurasian watermilfoil (Image credit: Chris Evans, River to River CWMA, Bugwood.org)

Aquatic plants provide critical habitat structure for invertebrates and fish as well as important substrates on which algae and microbes grow (Carpenter and Lodge 1986). Yet, not all macrophytes provide equal quality habitat and structure. In general, submerged aquatic plants with dissected (complex) leaf architecture provide more surface area and protection from predators and, therefore, greater densities and biomass of macroinvertebrates (Cheruvelil et al. 2002). However, despite having finely dissected leaves, EWM tends to support fewer individuals and species of macroinvertebrates (Cheruvelil et al. 2002). Work by Wilson and Ricciardi (2009) further suggests that EWM supports *different* macroinvertebrate communities (in terms of both species richness and abundance) than structurally similar native milfoils. They hypothesize that subtle differences in plant chemical exudates, rather than plant structure (which is very similar) might be responsible. EWM exudes polyphenols in higher concentrations than other macrophytes. These chemicals appear to inhibit the growth of epiphytic algae (Gross et al. 1996) and may deter generalist herbivores from eating it (Marko et al. 2008).

By altering the abundance and composition of their macroinvertebrate prey as well as habitat structure, EWM may negatively affect fish abundance, composition and diet (Keast 1984, Lyons 1989, Dibble and Harrel 1997). Dense beds of EWM may lead to the overpopulation and stunted growth of forage fish, like bluegill (Engel 1995), as well as physically inhibit foraging behavior by predatory game fish such as largemouth bass (Valley and Bremigan 2002).

Dense canopies of invasive macrophytes like EWM have also been found to profoundly alter water quality, creating strong vertical gradients of pH and dissolved oxygen in the water column, which in turn affects the distribution of macroinvertebrates and fish (Carpenter and Lodge 1986, Frodge et al. 1990).

Box 1. The impact challenge

Management actions are enacted to halt or slow the negative impacts of an invasive species. Yet for many invasive species, we have incomplete and often only anecdotal information about impacts, particularly with respect to ecological communities (species assemblages) and ecosystem processes (Blossey 1999, Parker et al. 1999). Baseline data are infrequently collected prior to invasion, and most impact studies take place at small spatial and temporal scales, such that variability in an invader's impact over its entire range or over extended time scales is rarely quantified. In the studies discussed above, the impact of EWM has typically been observed in a single or limited number of water bodies and rarely includes pre-invasion baseline data or comparisons to uninfested water bodies. Moreover, given that EWM interferes with numerous human uses of water bodies (see *Economic Impacts* below), long-term studies that monitor the ecology of EWM-infested waters (without any or just minimal management intervention) are rare.

Long-term ecological studies in water bodies where EWM has been well-established may yield important insights that could shape future management actions and priorities. In a rare example because of its time coverage and the absence of management intervention, Carpenter (1980) reported that EWM populations declined dramatically after 10 years in Lake Wingra, WI. Carpenter could not attribute the decline to a specific factor, but hypothesized that the decline might be the result of naturally occurring interactions between several ecological factors, including nutrients, epiphytes, competitors, and parasites or pathogens. Evidence has similarly emerged from terrestrial ecosystems. A long-term monitoring study of garlic mustard infestations in woodlands has noted population declines in garlic mustard with concomitant recovery of the native groundlayer vegetation less than 10 years after initial invasion (Blossey, personal communication). Indeed, more long-term studies like these are needed to improve our understanding about how the impact of invasive species may change over time, and consequently how our management priorities ought to shift in light of those changes.

Economic Impacts

Eurasian watermilfoil significantly affects the recreational use of infested lakes and ponds by impeding boating, swimming and angling as well as generally interfering with the aesthetic quality of infested water bodies (Newroth 1985, Smith and Barko 1990). Yet, the specific economic costs of lost recreational opportunities directly attributed to EWM and other invasive plants have been difficult to quantify (Eiswerth et al. 2000, Eiswerth et al. 2005).

Recently, ecological economists have tried to estimate the costs of invasive milfoil infestations to property and land values. For example, Horsch and Lewis (2009) estimated that the presence of an EWM infestation in Wisconsin lakes resulted in an average 13% decline in property value. Halstead et al. (2003) suggested that the presence of another invasive milfoil, *Myriophyllum heterophyllum*, could result in 20-40% declines in the value of New Hampshire shoreline properties. Although they could not directly attribute property value decline to EWM alone, Zhang and Boyle (2010) reported that total aquatic plant coverage (of which EWM was a major component) significantly reduced lake front property values (from <1% to 16%) on five Vermont lakes. These studies, although somewhat limited in sample size, geographic scope, and specificity of cost estimates, mark a significant step forward towards the accounting of the real costs of invasive species and may provide a stronger basis for natural resource agencies and government decision-makers to prioritize EWM prevention efforts.

Although assigning costs to lost recreational opportunities and property values has been challenging, the costs to remove and manage EWM – known as control costs – are easier to quantify (Eiswerth et al. 2000). For example, Kelting and Laxson (2010) reported that labor costs for the hand harvesting of EWM in Upper Saranac Lake, NY, from 2004 – 2008 averaged more than \$350,000 per year (~\$725 ha^{-1}) during the intensive management period and approximately \$150,000 per year (~ \$300 ha^{-1}) during the maintenance period. Boylen et al. (2011) reported that over \$5 million was spent to control EWM in Lake George between 1985 – 2010. Between 2005 and 2008 NYS issued nearly \$1.3 million in "eradication grants" specifically for Eurasian watermilfoil management. Unfortunately, much of the data regarding EWM control costs is buried in unpublished project reports and lake management plans (*Table 1*), and has rarely been analyzed and discussed in a systematic way in the scientific literature (But see Kelting and Laxson 2010). Work is currently underway to comprehensively tally aquatic plant management costs in New York State (Mueller, personal communication).

Table 1. Cost comparison of methods for removing Eurasian watermilfoil in Black Lake, Hammond, NY. Information is replicated from a management plan prepared by Quantitative Environmental Analysis, LLC (2008) for the Black Lake Invasive Weeds Committee, and is discussed in Zhang and Boyle (2010).

Class	Method	Costs
Physical	Hand harvesting	\$400 - \$1,000 per acre
	Suction harvesting	\$20,000 - \$30,000 for equipment and \$1,000 - \$25,000 per acre for operations and disposal of harvested plants
	Benthic barrier	\$10,000 - \$20,000 per acre for professional installation
Mechanical	Rotovating	\$100,000 - \$200,000 for equipment & \$200 - \$300 per acre for operations; \$1,500 per acre to hire professional service
	Mechanical harvesting	\$100,000 - \$200,000 for equipment and \$200 - \$300 per acre for operations
	Dredging	\$1,000 - \$40,000 per acre depending on chemical nature of sediment and need for off-site disposal
Biological	Herbivorous insects	Stocking costs approximately \$1,000 per acre

	Grass carp	Stocking costs $50 - $100 per acre
Chemical	Aquatic herbicides	$200 - $400 per acre

Management Approaches, Outcomes, and Limitations

Given the potential ecological and economic impacts associated with its invasion (above), Eurasian watermilfoil has long been the target of numerous and costly management efforts in New York State. As discussed below, a number of approaches can be employed to manage EWM. Although often the desired outcome is eradication, most methods offer only limited relief (on the order of weeks to a few years).

Physical management practices

Physical techniques – including hand harvesting, benthic barriers and water drawdowns – have been used extensively to manage Eurasian watermilfoil infestations, particularly in lakes in the Adirondack Park. Physical management techniques can significantly reduce EWM populations, but require extensive amounts of time and money to achieve long-term control (Boylen et al. 1996).

Both hand and suction harvesting involve divers, aided by SCUBA equipment or some other breathing apparatus (e.g. hookah rig), that remove whole EWM plants (including stems, roots, and leaves) from the substrate. Divers either stuff harvested plants into mesh dive bags that they return to the surface or are aided by a hydraulic vacuum dredge hose attached to an engine and collection basket on a surface barge.

Hand and suction-harvesting have typically been employed at smaller spatial scales to manage EWM on a site-by-site basis within larger bodies of water, as has been the case at Lake George (Eichler et al. 1993, Boylen et al. 1996). Both techniques can result in significant reductions in EWM biomass and percent cover, with concomitant increases in native species in the year post-harvest (Eichler et al. 1993, Boylen et al. 1996). However, neither method results in the eradication of EWM, and regular maintenance management via additional hand harvesting is recommended at least every 2-3 years to prevent EWM dominance (Boylen et al. 1996). Moreover, despite intense physical management efforts since 1986, the number of locations in Lake George confirmed to have EWM has continued to increase, with 183 total sites confirmed as of 2010 (King and Laginhas 2010).

Harvesting by hand has minimal direct impacts on native aquatic plants as divers can selectively pick out individual plants (Boylen et al. 1996). Native plant communities may initially decline following suction harvest, however, especially if native macrophytes are co-mingled with EWM. Coontail (*Ceratophyllyum demersum* L.), for example, lacks roots and may be particularly susceptible to incidental harvest (Eichler et al. 1993). Despite these short-term impacts, native plant communities appear to recover and even increase in number of species the year following harvest (Boylen et al. 1996).

Generally suction harvesting is not recommended in lakes with soft, flocculent sediments that are easily disturbed or in shallow water areas that may limit access by divers and suction equipment (Eichler et al.

1993). Eichler et al (1993) suggest this technique works best where EWM grows in moderate to high density over a limited geographic area. Logistical issues related to the transportation and set-up of equipment make it impractical for widely scattered populations.

Kelting and Laxson (2010) recently reported the outcomes of an intensive, whole-lake hand harvesting effort in Upper Saranac Lake. Prior to the whole-lake approach initiated in 2004, localized control efforts (i.e., hand harvesting, benthic matting) were conducted 1999 – 2003; although localized reductions in EWM were achieved, EWM expanded to non-managed areas of the lake. Intensive management occurred over 3 years (2004- 2006) with dive teams circumnavigating and removing EWM from the entire littoral zone of the lake at least two times a season (June – August), with smaller teams conducting maintenance harvests 2007-2008. Following the intensive and maintenance management periods, EWM cover was reduced to less than 5% cover for more than 90% of the littoral area of the lake. Moreover, EWM stem densities at 13 monitoring sites declined from an average 1650 stems ha^{-1} (± 343 SE) in 2004 to 63 stems ha^{-1} (± 9.26 SE) in 2006, maintaining similarly low densities through the maintenance period. Kelting and Laxson (2010) noted, however, that EWM colonization did trend upwards during the maintenance period indicating that maintenance efforts might have to increase and certainly continue indefinitely to keep EWM in check.

Benthic barrier mats made out of solid plastic (PVC, e.g. Palco^{TM}) or fiberglass small-mesh screen (e.g. Aquascreen) are deployed and secured to lake bottoms as another physical aquatic plant control strategy. Solid barriers limit plant emergence indiscriminately, and result in plant decay within 30 days (Mayer 1978) and root mass decomposition within 60 days (Eichler et al. 1995). Although rare, some plants, including EWM, may survive underneath permeable screen barriers and can send shoots up through the mesh (Mayer 1978, Eichler et al. 1995). Generally, given the non-selectivity of the treatment, benthic barriers are deployed where EWM dominates the aquatic plant community (>50%) (Eichler et al. 1995).

Following removal of the benthic barriers installed at Lake George for 1-2 years, native plant species recolonized the habitat within 30 days and were present up to 2 years later. However, EWM also quickly recolonized, as it was present in 44% of formerly covered areas 30 days after barrier removal and increased to over 70% presence 90 days after removal, a figure that remained consistent one year later (Eichler et al. 1995, Boylen et al. 1996). Average percent cover of EWM increased over this period from about 3% 60 days after barrier removal to nearly 14% the following year (Eichler et al. 1995). Eichler et al. (1995) suggested that proximity of benthic barriers to other areas infested with EWM (source of fragment propagules) strongly affects rate of recolonization and should be a major consideration when choosing management techniques. Moreover, they concluded that benthic barriers are not a stand-alone management technique and require significant follow-up maintenance management actions (e.g. hand harvesting).

Winter drawdown of lake levels is another physical EWM management option for lakes with water levels controlled by dams or other structures (NYS DEC Division of Water 2005). Water levels must be drawn down at least three feet to expose plants and sediments to freezing and drying action. Drawdowns will disturb the entire aquatic plant community and may negatively impact benthic macroinvertebrates that cannot tolerate the stress of winter drawdown conditions (Harman et al. 2005). Moreover, large quantities of decaying plant biomass in the spring may limit water clarity (Adamec and Husak 2002). Additionally, if EWM grows in waters deeper than that affected by the drawdown, little control will result (Wagner and Falter 2002). Water drawdowns have rarely been evaluated in the scientific literature (but see Smith and Barko 1990, Adamec and Husak 2002, Wagner and Falter 2002). Limited reports in the gray literature indicate that it knocks back EWM populations for a few years and that native plant species, if present, recover (NYS DEC Division of Water 2005). Consequently, it is considered a short-term management practice.

Mechanical control

Mechanical harvesting has been used extensively to manage EWM infestations in the upper Midwest (Crowell et al. 1994, Wilson and Carpenter 1997) as well as New York State (NYS DEC Division of Water 2005). Harvesting is frequently employed to clear infested areas around docks, landings, and swimming areas as well as create edge habitat for fish (Wilson and Carpenter 1997, Unmuth et al. 1998). Conventional weed harvesters are typically outfitted with two upright cutting bars and a vertical bar that cut channels 6 – 10 feet wide and from 4 – 8 feet in depth (NYS DEC Division of Water 2005). Some mechanical harvesters can be fitted with a deep cutting bar that can be lowered as deep as 16 feet below the water's surface (Wilson and Carpenter 1997, Unmuth et al. 1998). Cut material is either transported directly to a conveyer belt and stored on the harvester for later disposal or floats to the surface and is raked up by a trailing machine. Unless targeting a monospecific stand of plants, mechanical harvesters are not selective in their removal of submerged aquatic vegetation. Plant fragments that are not removed may disperse and actually exacerbate spread of the EWM (Madsen et al. 1989).

Harvesting generally provides short-term reductions in EWM biomass, on the order of weeks to months. For example, in Lake Minnetonka, MN, Crowell et al. (1994) observed that a mid-growing season (July) harvest reduced average EWM biomass in plots for 6 weeks, compared to reference areas. Other studies have reported that the effects of EWM harvesting last only 3-4 weeks, with multiple harvests required over the growing season (Rawls 1975, Cooke et al. 1990). Kimbel and Carpenter (1981) suggest that despite the rapid re-growth of biomass and return to reference conditions 6 weeks after harvest, there may also be some beneficial longer term effects of harvesting; the year following a harvest in Lake Wingra, WI, EWM biomass was significantly lower in the cut plot. They suggest that harvesting, particularly later in the growing season, disrupts carbohydrate storage. Additionally, cuts made close to the sediment surface in deeper waters (> 3m), closer to the root crown at the base of the shoots, appear to have longer lasting (up to three years) negative impacts on EWM growth (Unmuth et al. 1998).

Few studies have evaluated the long-term impacts of mechanical harvesting (but see Bowman and Mantai 1993, Nichols and Lathrop 1994), and results have been equivocal. Chautauqua Lake, particularly the north basin, has been mechanically harvested since the 1950s (Nicholson 1981). Comparing data from aquatic plant species surveys conducted in 1937, 1972-1975, and 1991, Bowman and Mantai (1993) conclude that the aquatic plant community of Chautauqua Lake, although maintaining similar species richness across years, has experienced a major change in relative plant abundance; larger leaved species, including several *Potamogeton* spp., have declined dramatically in favor of species more resistant to mechanical cutting and herbicides (e.g., EWM, *Ceratophyllum demersum*, *Elodea*). Nichols and Lathrop (1994) compared the aquatic plant communities in portions of Lake Wingra, WI, that have been mechanically harvested since the late 1960s and others that remained unharvested. Although species richness was greater in 3 out of 4 unharvested areas compared to harvested areas, differences could not be directly attributed to a history of mechanical harvesting.

Chemical control

Scientific research over the last 60, and particularly the last 20, years has advanced our understanding and improved the application of chemical strategies to control invasive aquatic plants (Getsinger et al. 2008). The use of aquatic herbicides has evolved from broad-spectrum inorganic compounds in the 1900s (e.g. copper) to plant-specific enzyme inhibitors in the 1980s-90s (e.g. fluridone) to hormone-mimicking systemic herbicides selective for dicots (broadleaf plants) in the 2000s (e.g. triclopyr). Methods and chemical formulations have been developed to minimize non-target effects and reduce chemical use rates.

Herbicides are applied to submerged aquatic vegetation via the water column and can be greatly affected by water exchange processes that may dilute herbicide concentrations and move the herbicide away from the target plant (Getsinger et al. 2008). A major challenge faced in the application of any aquatic herbicide is maintaining contact times sufficient for achieving control of target plant. In light of this challenge and the strict regulatory frameworks established by the US EPA and state agencies like NYS DEC, most published scientific research on aquatic herbicides has investigated the concentration and exposure time necessary for effective control under laboratory and field settings.

Below I review the science pertaining to the five aquatic herbicides registered in New York State known to have a significant impact on EWM: Diquat, Endothal, 2,4-D, Fluridone and Triclopyr (*Table 2*).

Table 2. Attributes of aquatic herbicides registered for use in New York State (USACE Environmental Laboratory 2011).

Herbicide	Trade names	Activity	Maximum Water Concentration	Time To Plant Control
Endothall 7-oxabicyclo[2,2,1] heptane-2,3-dicarboxylic acid	*Aquathol K (liquid)* *Aquathol Super K (granular)* Hydrothol 191 (g & l)	Contact	5 ppm	7-14 days
Diquat 6,7-dihydrodipyrido (1,2-α:2',1'-c) pyrazinediium dibromide	Reward (liquid) Weedtrine-D (liquid)	Contact	0.37ppm	7 days
2,4-D 2,4-Dichlorophenoxyacetic acid	Aqua-Kleen (granular) Navigate (granular) DMA 4 IVM (liquid) Weedar 64 (liquid)	Systemic	(4 ppm)	2 weeks
Triclopyr [(3,5,6-trichloro-2-	Renovate 3 (liquid)	Systemic	2.5 ppm	3-5 weeks

pyridinyl)oxy]acetic acid				
Fluridone 1-methyl-3-phenyl-5-[3-(trifluoromethyl) phenyl]-4(1H)-pyridinone	Sonar AS (liquid) Sonar SRP (granular) Sonar Q (granular) Sonar PR (granular) Avast! (liquid)	Systemic	90 ppb in ponds, 150 ppb in lakes/reservoirs	30-90 days

Contact herbicides

Both diquat and endothall are contact herbicides, acting on plants' exposed tissues, but not affecting "underground" portions such as root crowns, roots or rhizomes. Both chemicals are broad-spectrum herbicides and will effectively knock back a wide range of aquatic plants quickly (Madsen et al. 2010). As a result both chemicals are frequently used to relieve aquatic plant nuisance issues in areas where water exchange can shorten chemical exposure time, including docks, marinas, swimming areas, and irrigation canals (Skogerboe et al. 2006, Madsen et al. 2010).

Registered for aquatic use by the US EPA in 1962 (Netherland et al. 2005), diquat acts quickly by disrupting photosynthesis in exposed tissue. However, because diquat is cationic (positively charged), molecules of the herbicide bind tightly to suspended particles in water; thus, turbid or muddy waters can significantly reduce its effectiveness by preventing sufficient contact time of the herbicide with the plant (Weber et al. 1965, Hofstra et al. 2001).

Endothall, a dipotassium salt registered by the US EPA in 1960, acts on plant cell membranes and may interfere with protein and lipid synthesis during metabolism (Senseman 2007). Studies of the effect of endothall on aquatic plant communities in both northern and southern latitudes suggest that plant response to the herbicide varies with species and concentration (Skogerboe and Getsinger 2001, 2002). Results from a concentration exposure time study by Skogerboe and Getsinger (2002) indicate that EWM could be killed effectively at low application rates of endothall ($\leq$ 1mg L^{-1}) without persistent effects on native aquatic plants like sago and Illinois pondweeds (*Potamogeton pectinatus* L., *Potamogeton illinoensis* Morong.) , American eelgrass (*Vallisneria americana* L.), or coontail (*Ceratophyllum demersum* L.). In a follow-up field study, Parsons et al. (2004) applied a low concentration (1.5 mg L^{-1}) of endothall to 8 ha around the edge of Kress Lake, WA, that was heavily infested with EWM. They monitored the aquatic plant community before and after (10 weeks, 1 year, 2 years, 3 years), and found that EWM significantly declined in biomass (99% reduction) and frequency (present in only 15% of samples compared to 85% pre-treatment) 10 weeks after treatment. Although EWM frequency increased to 39% of samples by the third year following treatment, biomass remained depressed (97% reduction from pre-treatment). Native plants including common elodea (*Elodea canadensis* Rich.), muskgrass (*Chara* sp. Vallaint.), and bladderwort (*Utricularia* sp. L.) all significantly increased in frequency and biomass post-treatment.

Systemic herbicides

Systemic herbicides are typically absorbed from the point of application by plant leaves or roots and are then translocated throughout the plant's tissues, acting on photosynthetic or metabolic pathways. Systemic herbicides tend to be slower acting than contact herbicides, but are generally considered more effective rendering plant control.

2,4-D is absorbed by treated leaf tissues and is then translocated throughout the plant. Mimicking the plant-growth hormone auxin, 2,4-D affects plant respiration, food reserves and cell division, ultimately causing plant death in 7-14 days (Christopher and Bird 1992, Senseman 2007). 2,4-D was one of the first aquatic herbicides to provide selective control for dicots, and has remained an important and popular aquatic plant management tool since the 1940s (Lembi 1996). Field applications of 2,4-D at both the small plot (Getsinger and Westerdahl 1984, Killgore 1984, Parsons et al. 2001) and whole lake scale (Crowell 1999) have indicated that 2,4-D significantly reduces EWM in the short-term (28 days to 6 weeks), but that typically EWM biomass rebounds over the longer term (2 months to 1 year). Thus, 2,4-D may be a useful tool for slowing spread of EWM, but will not eradicate it from a water body and requires maintenance or follow-up interventions.

Triclopyr, in its triethylamine salt formation (Renovate^{TM}3), was registered for aquatic use by the US EPA in 2002 and by New York State in 2007. It has a similar mode of action as 2,4-D, mimicking the plant growth hormone auxin. Plants absorb triclopyr through roots, shoots and leaf tissues, and it is then translocated through the plant, causing death within 7-14 days (Netherland and Getsinger 1992). Field experiments indicate that triclopyr is quickly degraded by microbes, rapidly dissipating from aquatic systems and posing little toxicological risk to non-target organisms, including fish and shellfish (Petty et al. 2003). Like other aquatic herbicides, the efficacy of triclopyr is dependent on the concentration and the amount of time EWM is exposed to the herbicide (Netherland and Getsinger 1992). The efficacy of triclopyr in controlling EWM as well as the recovery of native plant communities following treatment has been evaluated in both lake and regulated river systems at various concentrations (Getsinger et al. 1997). Following experimental applications in a Washington river and adjacent cove (at maximum label rate of 2.5 mg ae L^{-1}), Getsinger et al. (1997) reported that EWM biomass was reduced by 99% four weeks after treatment, with significant declines in EWM biomass (< 50% pre-treatment levels) still discernible two years post-treatment. Moreover, non-target native plant biomass and diversity increased dramatically following the removal of EWM, discernible up to two years after EWM treatment. Poovey et al. (2004) conducted experiments where they applied lower doses of triclopyr (0.5 – 1.5 mg ae L^{-1}) at small plot (2.5 acre) scales in two Minnesota lakes and evaluated chemical residues in water samples as well as EWM and native plant species. They reported that triclopyr dissipates rapidly at small plot scales, resulting in less effective control than reported by Getsinger et al (1997). Poovey et al. (2004) conclude that longer term EWM control could be better achieved by applying triclopyr in larger contiguous areas (5-10 acres), at higher rates (> 1.5 mg ae L^{-1}), and potentially by using sequential applications in the same area (over a 2-8 hour period) so long as total treatment dose did not exceed maximum label rate.

Box 2. Triclopyr: New York Case Study

In a 2-phase process beginning in 2009 and ending in 2010, the Town of Cazenovia applied triclopyr (trade name Renovate OTF) at a target rate of 1.5 – 2.0 mg ae L^{-1} to over 400 surface acres of Cazenovia Lake, a moderately productive hard water 1164-acre lake that supports a diverse and productive warm water fish community (Town of Cazenovia 2009). Evaluation of effects on EWM and the native aquatic plant community following Phase I (application to northern end of lake) indicated significant reduction of presence and biomass of EWM. In 2008 pre-treatment assessments, there were 281 locations with EWM present, and 60% of those points had medium or dense abundance of EWM. Following Phase I in 2009, EWM was detected in only 122 locations, and just 14% of those had medium or dense abundance in 2009 (Johnson et al. 2009). After Phase II (treatment of the southern half of the lake), EWM was found at only 88 locations with just 1% of those categorized as having dense or medium abundance (Johnson et al. 2010a). Native plant species were not significantly impacted by the triclopyr application, and in fact, a general increase in abundance and diversity was recorded after treatment as compared to pre-treatment. Continued monitoring will be necessary to assess the longer-term success of the triclopyr applications in Cazenovia Lake and what, if any, maintenance management actions may be needed.

Fluridone, a systemic herbicide approved for aquatic use by the US EPA in 1986, is absorbed from the water column by plant shoots and disrupts plant photosynthetic pathways by inhibiting a specific plant enzyme, phytoene desaturase. Fluridone has been used extensively to control EWM and hydrilla in lake and flowing water systems. When fluridone is applied at high concentrations, close to the maximum US EPA labeled dose rate (0.15 mg L^{-1}), it acts like a broad spectrum herbicide; however, in aquatic systems it is generally applied at much lower concentrations and tends to act more selectively. Unlike other approved aquatic herbicides, target plants must be exposed to fluridone for an extended time (> 60 days) to successfully kill the plant and prevent regrowth (Netherland et al. 1993). Large, whole-lake applications of fluridone were conducted throughout the late 1980s and early 1990s (Getsinger et al. 2002); during that time, however, concerns emerged regarding the non-target effects of the herbicide on native vegetation at herbicide concentrations $\geq$ 10 µg L^{-1}(Smith and Pullman 1997, Madsen et al. 2002). Research in the field and laboratory confirmed that EWM was sensitive to fluridone at very low concentrations (< 5 µg L^{-1}) provided sufficient exposure time (> 60 days). Consequently strategies were developed to increase selectivity to EWM by considering the timing of the treatment (to allow for mixing and thermocline effects) and aquatic plant species composition of the infested lake as some native plant species are more sensitive than others, e.g. elodea, coontail, naiad, and native watermilfoils (Smith and Pullman 1997, Getsinger et al. 2002). Today, whole lake treatment at low concentration of fluridone is a widely accepted practice (Getsinger et al. 2008).

Herbicide resistance

When pesticides, particularly those that act on a single biochemical site of action (e.g. a single enzyme), are used repeatedly in a growing season without any other control actions, the target pest may develop pesticide resistance. The pesticide exerts intense selection pressure on target pests, such that individuals in the population carrying a mutation for resistance will survive and successfully produce offspring. The proportion of resistant pests in the population will increase as the pesticide continues to be applied and the resistant pests reproduce.

Pesticide resistance in weeds, insects, and pathogens has been widely documented in terrestrial agricultural systems over the last century; in 2004, fluridone resistance was confirmed for the first time in an aquatic invasive plant, hydrilla (Michel et al. 2004). Fluridone-resistant hydrilla has now spread throughout Florida. In 2006, diquat-resistant dotted duckweed (*Landoltia punctata*) was also confirmed in Florida (Koschnick et al. 2006). To date, herbicide resistance has not been detected in EWM, but there remains concern about the potential for herbicide resistance in *M. spicatum* x *M. sibiricum* hybrids given previous documentation of it in terrestrial plant hybrids (Moody and Les 2007). Laboratory tests comparing the susceptibility of a milfoil hybrid (a Minnesota population) and its EWM parental species to triclopyr and 2,4-D found no indication of hybrid resistance (Poovey et al. 2007). However, mutations can happen randomly, and could therefore emerge in EWM and its hybrid at any time.

Thus, herbicide resistance in aquatic invasive plants remains a significant concern for scientists and natural resource managers. Recurring EWM control should, at the very least, involve the rotation of chemical products and ideally integrate multiple control techniques (e.g., integrated pest management).

Box 3. On timing of control treatments

It should be noted that the timing of EWM control applications with respect to EWM phenology must be considered in order to maximize treatment efficacy. Reviewing phenological studies of EWM carbohydrate storage from across the US, Madsen (1997) suggests that timing treatments with EWM carbohydrate storage low points may help limit re-growth post-treatment. In northern EWM populations, Madsen (1997) suggests that these low points occur May through July. If and how carbohydrate low points may shift with climate change remains an open question.

Biological control

Unlike physical, mechanical, and chemical methods that typically provide control over shorter time scales, biological control is employed for the long-term suppression of invasive plants; it will not, however, eliminate the target plant. Consequently, biological control may be better suited for use in lower priority sites or over large areas where other management options are not feasible or cost effective. High priority sites requiring rapid and effective control – e.g. boat channels, docks, swimming areas – should consider alternative approaches.

To date, efforts to locate and test the specificity of herbivores from the native distribution of Eurasian watermilfoil for a classical biological control program have been unsuccessful (Cock et al. 2008). However, two indigenous insects and one naturalized insect located within EWM's introduced range in North America have been identified and investigated for their potential for EWM biological control. Research to date suggests that the native milfoil weevil, *Euhrychiopsis lecontei* (Dietz), offers the most promise as a control agent (Newman 2004).

Euhrychiopsis lecontei

The milfoil weevil, *Euhrychiopsis lecontei* (Dietz) (Coleoptera: Curculionidae), is native to the northern US states and southern Canadian provinces. Both the weevil adults and larvae feed on EWM (*Figure 5*), preferring the invasive over its native host plant, northern watermilfoil (*Myriophyllum sibiricum*) (Creed and Sheldon 1993). Johnson et al. (1998) surveyed EWM herbivores at 35 sites in 26 lakes throughout 16 counties in the Finger Lakes-Lake Ontario Watershed region of New York. The milfoil weevil was common, occurring at 29 of 35 sites, in 24 of 26 lakes.

Adult milfoil weevils overwinter in dry leaf litter along a water body's shore and return to the water in the spring after ice out. Adults will feed on the top portions of the plants, and after gaining sufficient energy for gamete maturation, will mate. Females lay their eggs (2 per day) near the milfoil's growing tip (meristem) after water temperatures reach 10-15°C. After hatching, larvae feed on the meristem and bore down the stem, mining a length of approximately 15 cm (Mazzei et al. 1999). They bore a chamber further down the stem (0.5 – 1m from the plant's growing tip) to pupate. Developmental rates are temperature dependent, ranging from approximately 62 days at 15°C to 17 days at 31°C for the completion of all stages (from egg to adult). Consequently, stem damage rates also increase with increasing temperatures. In Minnesota, milfoil weevils complete 3-6 generations per summer (Mazzei et al. 1999), whereas in Vermont, 3 generations are more typical. The last summer generation allocates its energy to fat-body reserves and the development of flight muscles, rather than eggs, and these adults migrate to the shoreline in September – November.

Figure 5. Left: Adult milfoil weevils feeding on EWM. Right: Milfoil weevil eggs on EWM, with larva feeding in center (Image Credit: Robert Johnson)

Under controlled experimental conditions, feeding by the adult and larval weevils can significantly reduce EWM plant buoyancy, shoot biomass and height, and root and carbohydrate stores, suppressing its growth and potentially affecting EWM's ability to overwinter and grow the following spring (Newman et al. 1996). However, evidence of good EWM control in the field has been equivocal, and when control is observed, it is often localized in effect (Creed and Sheldon 1995, Newman et al. 1996, Newman and Biesboer 2000, Newman 2004).

Weevil density appears to be a significant factor determining the degree of EWM control. Weevil densities $\geq$ 0.5 per stem were associated with EWM decline and suppression in Vermont (Creed and Sheldon 1995), although Newman and Bisboer (2000) suggest a higher density, 1.5 weevils per stem, might be necessary to effect EWM control, as observed in a Minnesota lake. Predation by sunfish (Sutter and Newman 1997, Ward and Newman 2006), lack of suitable over-wintering habitat (Newman 2004), and impacts on host plants from mechanical harvesting or herbicide control may all reduce natural milfoil weevil population densities (Sheldon and O'Bryan 1996, Newman 2004, Newman and Inglis 2009).

Attempts have been made to augment natural weevil populations in lakes to achieve densities of 1-2 weevils per stem (Jester et al. 2000, Reeves et al. 2008). In fact EnviroScience, Inc. now offers milfoil weevils as a commercial control product (Milfoil Solution ®, formerly known as The Middfoil® process). Their proprietary augmentation process involves attaching clusters of milfoil containing eggs and larvae to EWM stems in an infested lake, with the expectation that newly hatched larvae will move over to the lake's resident EWM.

Formal evaluations of weevil augmentation are rarely published in the peer-reviewed, scientific literature. Most information evaluating the efficacy of augmentation remains bound up in unpublished annual reports or closely guarded by EnviroScience, Inc. Only one peer-reviewed paper considers the effectiveness of The Middfoil® augmentation process (Reeves et al. 2008). Using data from 30 augmented lakes in Michigan (n=29) and Wisconsin (n=1) collected over multiple years, Reeves et al. (2008) report no significant difference in EWM plant density at augmented versus control sites; moreover, their results indicate no significant relationship between final beetle density at augmented sites and proportion EWM plant density change. In Lake Bonaparte, NY, Johnson et al. (2008, 2010b) consistently found no significant increase in either milfoil weevil populations or EWM plant damage at sites augmented via the Middfoil® process (2002-2008) compared to controls over six years of monitoring (2003-2008). Taken together, these results suggest that augmenting milfoil weevil populations with the egg/larvae stage is an ineffective approach.

At previously untreated sites in Lake Bonaparte, NY, Johnson et al. (2008, 2010b), pursued an alternative augmentation strategy: releasing adult weevils. Comparing late summer weevil density (all stages) at adult-augmented and egg/larvae-augmented sites, Johnson et al. (2010b) reported significantly greater weevil density at adult-release sites. Moreover, weevil densities significantly increased the year after adult augmentation, while sites augmented with eggs/larvae exhibited no change. Johnson et al. (2010b) suggest that adult weevil augmentation will result in the production of multiple generations during a field season and will increase prospects for more successful biological control; eggs/larvae introduced in June/July (per standard EnviroScience Inc. protocol) are unable to produce more than 1 generation per season and may experience 20-70% survival as they develop to adult stages (Newman et al. 1997, Mazzei et al. 1999). More data and rigorous statistical analyses are needed to bolster these conclusions.

Some concern has been expressed regarding the susceptibility of in *M. spicatum* x *M. sibiricum* hybrids to herbivore attack, especially by *E. lecontei*. Moody & Les (2002) genetically confirmed that plants from a milfoil population in Lake Beulah, WI, exhibiting low susceptibility to the milfoil weevil (Jester et al. 2000), were indeed hybrids. Roley & Newman (2006) investigated milfoil hybrid resistance by measuring weevil performance on EWM, northern watermilfoil (NWM, *M. sibiricum*), and hybrids in a laboratory experiment. Weevils successfully developed on the hybrid, with survival rates intermediate to rates on EWM (high) and NWM (low); there was no difference in weevil development times or adult mass among the three species. They conclude that *M. spicatum* x *M. sibiricum* hybrids are not particularly resistant to weevil herbivory and that herbivore resistance is an unlikely mechanism for increased invasiveness in the hybrid.

Acentria ephemerella

The naturalized aquatic moth*, Acentria emphemerella* (Lepidoptera: Pyralidae), is commonly associated with EWM in New York and other northeastern states (Johnson et al. 1998)and has expanded its range to the upper Midwest (Johnson et al. 1998, Newman 2004). Native to Europe and Asia, *Acentria* was likely introduced to North America in the 1920s; it was first collected in Montreal, Quebec, in 1927. A survey of central New York lakes for EWM herbivores (1995-1997) indicated that *Acentria* commonly occurred throughout the region as it was present in 25 of 26 lakes (Johnson et al. 1998).

Studies of *Acentria* in its native European habitat have reported densities of 300 larvae m^{-2}, with ranges of 10 – 10,000 larvae m^{-2} (Gross et al. 2001, Gross et al. 2002). In Cayuga Lake, NY, Johnson et al. (1998) report *Acentria* densities during the summer at 27 – 100 larvae m^{-2}, but consider these to be a gross underestimate. Hibernating larvae surveyed in milfoil stems in nearby Seneca Lake in late fall were denser, approximately 500 m^{-2}.

Adult *Acentria* emerge throughout the summer, rather than in a synchronous fashion (Newman 2004). Male moths are small, winged, and grayish-white in color whereas females have reduced wings and are flightless (*Figure 6*, Buckingham and Ross 1981). Males mate with females sitting on the water's surface, and females lay single clusters of 115-400 eggs on submerged vegetation.

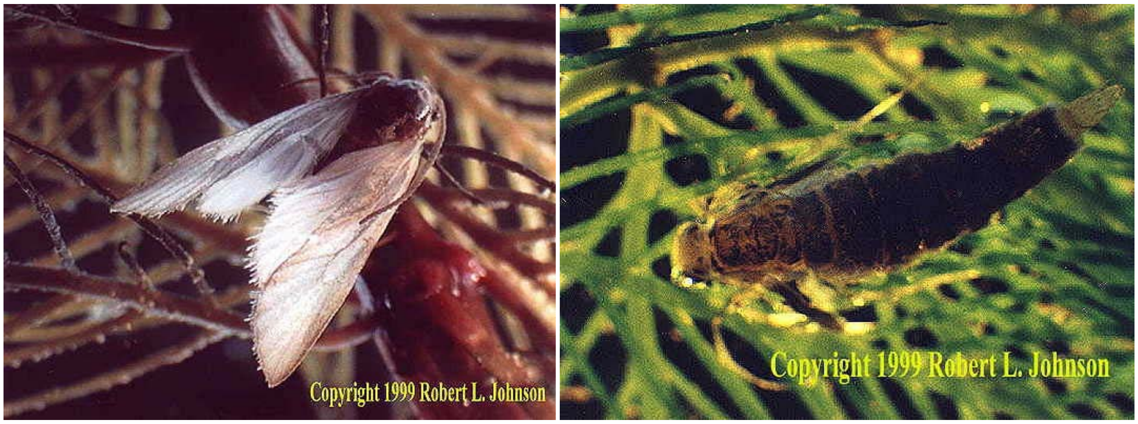


Figure 6. Left: Winged adult male *Acentria*. Right: Flightless adult female *Acentria*. (Image credits: Robert Johnson)

Moth larvae are generalist shredder herbivores and can successfully feed on a number of aquatic macrophytes, more than 20 species from 7 different plant families (Newman 2004), including EWM, *Ceratophyllum demersum*, *Elodea*, and *Potamogeton* spp. In choice tests, *Acentria* larvae preferentially choose EWM over *Elodea*; in the field, *Acentria* preferentially feed on EWM over a number of other macrophytes (Johnson et al. 1998). Larvae feed primarily on apical shoots and leaves, often constructing protective retreats from cut leaf material for protection from fish predation (*Figure 7*). Larvae reach the pupa stage in 5-7 weeks (Gross et al. 2001), pupating in a white cocoon firmly attached to a plant stem near the water's surface (Newman 2004). In the autumn, larvae in a range of instars (life stages) move to the lower stems of plants to hibernate (Newman 2004). In the spring, larvae begin actively feeding when temperatures rise above 10°C, provided adequate plant material is available. *Acentria* generally produce one generation per year, but may have 2-3 generations if environmental conditions are favorable (Gross et al. 2002).

Figure 7. Left: *Acentria* larvae of various ages and sizes. Right: Protective larval retreat in EWM apical shoot. (Image credits: Robert Johnson)

Acentria larvae feed primarily on apical meristems of EWM, the actively growing tip of the plant shoot that contains the greatest concentrated amounts of nutrients (Newman 2004). Herbivory by *Acentria* reduces EWM biomass and suppresses plant elongation and growth (Gross et al. 2001). Additionally, damage to apical shoots may interfere with allelopathy against epiphytes and phytoplankton, create wounds that facilitate pathogen invasion, and reduce autofragmentation (Gross et al. 2002).

Johnson et al. (1998) attribute the long-term suppression of EWM and the subsequent recovery of native macrophytes in Cayuga Lake, NY, to *Acentria.* This finding contrasts observations from Minnesota and Vermont where such lakewide declines and suppression in EWM have been linked to milfoil weevils, even though *Acentria* were present (Creed 1998, Newman and Biesboer 2000). Sampling five milfoil-infested lakes in central New York for *Acentria* and *Euhrychiopsis*, Johnson et al. (2000) report a negative correlation in densities of these two herbivores, indicating possible competition between the species. Survey work further indicates that *Acentria* may be denser in larger and deeper lakes with more diverse macrophyte communities (less EWM biomass) whereas *Euhrychiopsis* tends to be more abundant in smaller and shallower lakes where EWM beds dominate vegetation (Johnson et al. 2000). These observations are consistent with differences in the life histories of the herbivores (i.e., *Euhrychiopsis* needs onshore overwintering habitat that is not as common on large lakes) and their known diet breadth (i.e., *Acentria* is more of a generalist)(Johnson et al. 2000, Newman 2004).

Acentria can be successfully mass-reared in the lab and greenhouse, and their potential for biological control via augmentation is currently under investigation (Johnson, personal communication). Between 1999 and 2002, *Acentria* larvae and pupae were released in an augmentation experiment at Lincoln Pond, NY (Deming 2002). Augmentation neither increased total moth numbers nor caused a pond-wide decline in EWM. However, in one plot where *Acentria* pupae were released in late fall, significant declines in EWM biomass and total stem length (compared to controls) were observed the following summer. Deming (2002) hypothesizes that fish predation (particularly by sunfish) may be negatively affecting *Acentria* augmentation success in Lincoln Pond, and future research is warranted (Johnson, personal communication)

Cricotopus myriophylli

The milfoil midge, *Cricotopus myriophylli* (Diptera: Chironomidae), is native to North America and is widely distributed across the southern Canadian provinces and northern US states including New York, Wisconsin, and Minnesota. It is the least studied of the potential EWM insect biocontrol agents (Newman 2004).

Like *Acentria* larvae, *Cricotopus* larvae also establish on the apical shoots of Eurasian watermilfoil. They construct protective cases with silk and EWM leaf tissue and feed on the meristemic tissue (MacRae et al. 1990). Adults emerge throughout the summer, with mating, egg-laying and egg hatching occurring within a two week period following emergence (MacRae and Ring 1993). Typically producing only one generation per year, *Cricotopus* complete 4 larval instars before metamorphosis, and generally overwinter in the third instar within sealed cases of plant material (MacRae and Ring 1993).

In the late 1970s *Cricotopus* larvae were observed suppressing large beds of EWM in a lake system in British Columbia. Larval feeding inhibited EWM growth, preventing plant shoots from growing to the water's surface. Feeding trials suggested that one individual larva could crop a single meristem in 3-5 days (MacRae et al. 1990). Lab feeding experiments also indicated larvae are specific to *Myriophyllum* species (likely expanded from native *M. sibiricum* to EWM), as they exhibited little to no feeding activity on an array of native aquatic macrophytes (MacRae et al. 1990). *Cricotopus*, thus, has potential for EWM biological control, but generally has not been observed in the field in the high densities required for adequate control (Newman 2004).

Grass Carp

Grass carp, *Ctenopharyngodon idella*, a fish native to the temperate rivers of China and Russia, has been used extensively for aquatic plant management over the last 40 years in North America (*Figure 8*).

Concerns about the unintended escape and spread of reproducing fish to rivers as well as the potential negative impacts on native fisheries led to the development of a sterile triploid fish, produced by shocking eggs so that they keep an extra set of chromosomes (Cuda et al. 2008).

Figure 8. Triploid grass carp, *Ctenopharyngodon idella* (Image credit: NYS DEC)

The grass carp are considered “selective” generalist herbivores that feed on species in order of food preference and palatability. Over 50 genera of food items, primarily aquatic macrophytes, but also algae, invertebrates, and vertebrates, are consumed by grass carp. Preference studies indicate the carp preferentially consume hydrilla, pondweeds (*Potamogeton* sp.), and elodea, generally avoiding *Myriophyllum* species (Pine and Anderson 1991, Dibble and Kovalenko 2009). However, preferences can vary with carp age, sex, size, and population density as well as the species, abundance, and location of plants within a water body (Sutton and Vandiver 1998). Grass carp can live longer than 10 years and have persisted in ponds at least 2 years following the complete removal of the plants they were stocked to control (Kirk and Socha 2003).

Many concerns have emerged regarding the negative ecological impacts of triploid grass carp introduced for biological control (Bain 1993, Cuda et al. 2008). Grass carp can significantly reduce or eradicate desirable native plant species, reducing diversity and altering habitat structural complexity (Dibble and Kovalenko 2009). Changes in habitat structure may then negatively impact the abundance and diversity of macroinvertebrates, fish, and other aquatic invertebrates (Dibble and Kovalenko 2009). Moreover, the feeding and defecation of the grass carp can negatively affect water quality and clarity (Bain 1993, Cuda et al. 2008).

New York State permits the stocking of triploid grass carp in isolated water bodies for aquatic plant management (NYS DEC Bureau of Fisheries 2011). Since the first legal use of triploid grass carp in 1987 at Walton Lake in Orange County, NY (Surprenant 2001), the Bureau of Fisheries has continued to monitor the outcomes of stocking and adjusted recommended rates based on percent coverage of aquatic plants in a water body to prevent the complete removal of submerged aquatic vegetation. As previously noted, EWM is not a preferred species by the grass carp; however, evidence from some New York ponds and lakes suggest the triploid grass carp will consume it after feeding preferentially on more palatable vegetation (Surprenant 2001). Thus, sterile grass carp seem better suited for general aquatic plant management – as often desired in farm ponds or high use recreational areas – rather than for targeted EWM control.

Although concern about the Asian carp invasion of the Great Lakes is primarily focused on bighead carp (*Hypophthalmichthys nobilis*), silver carp (*H. molitrix*), and black carp (*Mylopharyngodon piceus*), feral grass carp remain a source of concern for the Asian Carp Working Group of the Aquatic Nuisance Species Task Force (Conover et al. 2007). In fact, twelve states (AL, ME, MD, MA, MI, MN, MT, OR, ND, RI, VT, WI) prohibit diploid and triploid grass carp (Dibble and Kovalenko 2009).

Pathogens

Surveys for potential bacterial and fungal pathogens that could attack and control Eurasian watermilfoil began in earnest in the 1970s (Shearer 2010). *Mycoleptodiscus terrestris* (Gerdeman), a fungus that breaks down cellulose, was isolated from milfoil in the 1980s and found to lethally attack plants in lab and field trials (Shearer 2010). EcoScience Laboratories developed and registered a commercial formulation of *M. terrestris* in the early 1990s, Aqua Fyte, but failures in its application design and field efficacy led to its discontinuation (Shearer 2010). The US Army Corps of Engineers isolated another isolate of *M. terrestris* from EWM and began combining it with various herbicides (i.e., fluridone, 2,4-D, triclopyr) for a more integrated control approach (Shearer 2010). Combining low doses of an herbicide and fungal pathogen resulted in longer-term and better plant control than when either agent was applied alone (Nelson and Shearer 2005, Nelson and Shearer 2008). The mechanism of interaction between herbicide and pathogen is unknown and may involve a number of different mechanisms given herbicides with different modes of action showed this favorable interaction (Nelson and Shearer 2008). Research is continuing to elucidate these mechanisms and develop a mycoherbicide with the new *M. terrestris* isolate.

Management research needs

The most recent research related to EWM management has focused largely on improving control via aquatic herbicides or insect biological control. Although improving herbicide target specificity and lowering the active dose of chemicals introduced into the environment would likely reduce the potential for non-target effects, it remains to be seen if a new or improved herbicide would actually eradicate EWM and eliminate the need for follow-up maintenance management. Biological control certainly offers potential for long-term suppression of EWM (not eradication), but maintaining adequate herbivore populations to effect control has proven difficult and requires further investigation.

Unfortunately, given the economic and recreational costs of EWM infestations, there is often great pressure on lake managers to take swift, immediate, and sometimes repeated action, often at the expense of research and data collection. In fact, many lakes where EWM control actions have been implemented and studied over the past three decades have been subjected to multiple treatments, either at the plot or whole lake scale, over time (Harman et al. 2005). And rarely, if at all, does a management study pair a treated lake with an untreated but infested control lake where the EWM population can be monitored without intervention (but see Olson et al. 1998). If we could think about management actions in a more experimental (Carpenter 1989) or adaptive management framework

(Williams et al. 2007), we might be able to answer a number of important, but unanswered research questions that have implications for EWM management:

- What are the long-term consequences of EWM management, particularly as it pertains to EWM dynamics, native plant communities, lake food webs and ecosystem processes?
- How does repeated intervention (e.g., chemical applications) affect plant community dynamics (native and invasive) as compared to EWM-infested sites with no intervention? Are there non-target effects at higher trophic levels?
- Are there thresholds for native plant diversity (species richness and abundance) pre-intervention that are necessary to facilitate native plant recovery post-intervention?
- What happens to EWM populations that are not managed over time? Do we have additional evidence of "natural" EWM population decline (see Box 1)? Under what conditions? Due to what factors?
- How do landscape level factors, particularly land use change and climate change, affect EWM invasion and population dynamics?

Spread Prevention

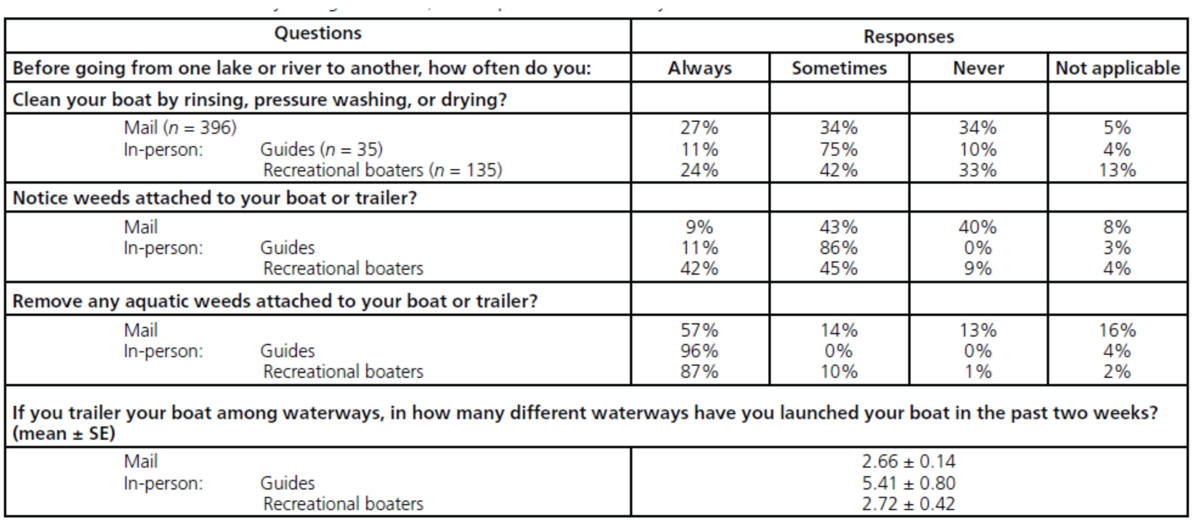
Generally, preventing the establishment and spread of invasive species is more effective and economical than eradicating or managing the impacts of an invasive species after it has already established (Hobbs and Humphries 1995, Leung et al. 2002, Lodge et al. 2006). Particularly for EWM, where management approaches provide only short-term control of EWM (on the order of a few years, at best) at great costs, prevention efforts become all the more critical.

Recreational boating as a transport vector

Recreational boating is an important pathway facilitating the transport of aquatic invasive species (Johnson et al. 2001). Yet, with respect to Eurasian watermilfoil, research evaluating boaters' launching activity and efficacy of cleaning practices suggests that despite significant outreach and education campaigns, there remains much room for improvement. Moreover, few studies have evaluated the efficacy of specific management interventions (e.g. boat launch stewards, boat washing).

To this end, using both mail and in-person surveys, Rothlisberger et al. (2010) polled about 1500 registered boaters in Wisconsin and Michigan about boat launching and cleaning practices (*Table 3*). Results from both survey groups indicated that more than two-thirds of boaters do not *always* take steps (i.e., rinsing, pressure washing, drying) to clean their boats. However, 57% of mail survey participants and approximately 90% of those surveyed in-person who launch their boats in more than one waterway during a season always removed weeds attached to their boat or trailer. Rothlisberger et al. (2010) also report that fishing guides visit two times more unique water bodies (over 5) than recreational boaters (almost 3), and are less likely to clean/dry boats or notice aquatic weeds. These results suggest that despite significant educational efforts in Wisconsin and Michigan to date, boaters are not consistently cleaning and inspecting their boats for aquatic invasive species (AIS). Moreover, professional fishing guides pose the greatest risk for AIS spread and may be important targets for a focused outreach campaign.

Table 3. Questions and responses from mail and in-person surveys of 1500 registered boaters in Wisconsin and Michigan, as reported in Rothlisberger et al. (2010). Sample sizes (in parentheses) are the number of transient boaters (i.e., boaters that launch in > 1 waterway during the season) that responded to the surveys.



Questions	Responses			
Before going from one lake or river to another, how often do you:	**Always**	**Sometimes**	**Never**	**Not applicable**
Clean your boat by rinsing, pressure washing, or drying?				
Mail (*n* = 396)	27%	34%	34%	5%
In-person: Guides (*n* = 35)	11%	75%	10%	4%
Recreational boaters (*n* = 135)	24%	42%	33%	13%
Notice weeds attached to your boat or trailer?				
Mail	9%	43%	40%	8%
In-person: Guides	11%	86%	0%	3%
Recreational boaters	42%	45%	9%	4%
Remove any aquatic weeds attached to your boat or trailer?				
Mail	57%	14%	13%	16%
In-person: Guides	96%	0%	0%	4%
Recreational boaters	87%	10%	1%	2%
If you trailer your boat among waterways, in how many different waterways have you launched your boat in the past two weeks? (mean ± SE)				
Mail	2.66 ± 0.14			
In-person: Guides	5.41 ± 0.80			
Recreational boaters	2.72 ± 0.42			

Data collected about boating behavior and spread prevention in New York State is limited, with a few notable and publically reported exceptions. The Watershed Stewardship Program (WSP) sponsored by Paul Smith's College's Adirondack Watershed Institute has sponsored a boat launch steward and training program since 2000 (Holmlund 2011). In 2010 WSP stewards inspected nearly 9000 boats at seven lakes, and reported almost 100 interceptions of EWM (out of approximately 600 total organism removals). Stewards collected data about visits boats made to other waterways two weeks prior to launching in Adirondack lakes, and they confirmed that a significant number of boats had previously visited waterways infested with invasive species (e.g., as many as 70% of boats inspected at the Second Pond boat launch at Saranac Lake). The majority of boaters polled reported taking one or more spread prevention measures (e.g. boat washing, bilge draining, visual inspection, drying) prior to arriving to the boat ramp.

Since 2006, the Lake George Watershed Steward Program has also inspected boats at launches, conducted boater outreach, and collected quantitative data about boats entering Lake George (DeBolt and Rohne 2010). In 2010 stewards inspected over 2500 boats at 4 launches, intercepting and removing organisms more than 80 times. Fifty-three of those samples were invasive species, and 36 of those were identified as EWM. Prior to launching in Lake George, boaters reported visiting 112 different water bodies in 13 different states, many of which are known to harbor aquatic invasive species (*Figure 9*). Ninety-five percent of boaters indicated taking some kind of spread prevention measure.

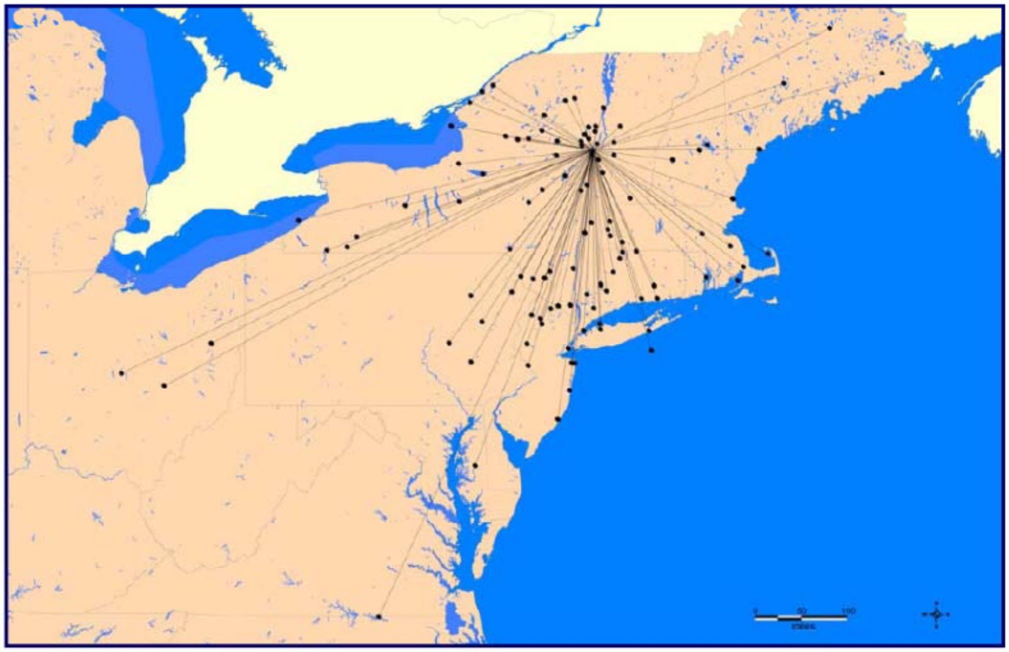


Figure 9. Map showing waterbodies visited within 2 weeks prior to launching at Lake George in 2010, as reported in DeBolt and Rohne (2010).

Taken together, these data collected from lakes in Wisconsin, Michigan and the Adirondacks indicate that there remains ample opportunity for recreational boating to introduce and spread EWM. Moreover, given the frequency of EWM interceptions, the incomplete coverage by lake stewards across the landscape, and inconsistencies in boater cleaning practices, the strengthening of aquatic transport prevention programs should be a strategic priority.

Viability of transported fragments

The ability for EWM to establish and spread in a new water body is somewhat dependent on its ability to survive and overcome the drying conditions of overland transport. Yet, the effects of desiccation on the viability of aquatic plant fragments (particularly EWM) have only recently been examined.

Evans et al. (2011) conducted field and lab experiments to determine EWM desiccation and fragment viability. They estimated that fragments exposed to drying conditions for 3 hours resulted in 87% desiccation, with 100% desiccation occurring after approximately 13 hours of drying. Although most fragments experiencing 3 – 48 hrs of desiccation did not experience re-growth under lab conditions, Evans et al. (2011) reported that a very small fraction of the fragments (including those dried 18-48 hrs, 100% water loss) were actually able to initiate new growth (stems, rootlets) from lateral buds.

Jerde et al. (submitted) also conducted a desiccation experiment to determine how quickly EWM fragments dry out during transport on trailered boats and what, if any, effect desiccation might have on root development and plant growth. They report the highest survival rates for EWM fragments exposed to air for short periods of time (< one hour) or those that are coiled (as would occur when EWM is wrapped around a boat propeller). Fragments and coiled plant pieces experiencing short desiccation periods were also able to set roots in less than two weeks. Given these results, the authors conclude that attention should be paid to boaters visiting neighboring lakes within short travel distances (e.g. 1 hour) on the same day, with inspections of watercraft targeting locations on boats and trailers where EWM can form coiled bunches (e.g., anchors, rollers, propellers).

Efficacy of cleaning practices

Regarding overall efficacy of boat cleaning practices, Rothlisberger et al. (2010) experimentally tested various boat cleaning methods, including low-pressure wash, high-pressure wash, and visual inspection/hand removal for different lengths of time (90 secs vs. 180 secs) on boats and trailers intentionally seeded with either EWM or small-bodied organisms (wetland plant seeds and spiny waterfleas). Both high-pressure washing and visual inspection followed by hand removal resulted in the removal of more than 80% of EWM, significantly more than low-pressure washing (~60%). With regard to small-bodied organisms, high-pressure washing removed 90% of organisms as compared to ~75% removed via low-pressure washing or visual inspection. Time did not significantly affect the amount of materials removed for either class of organisms.

Blumer et al. (2009) experimentally tested whether hot pressure washes (>40°C = 104°F), a recommended practice to eliminate zebra mussels, spiny water fleas, and viral hemorrhagic septicemia from boats and trailers, would also kill EWM fragments. In a lab study that exposed 20cm fragments of EWM to a range of temperatures over multiple exposure times, they found that temperatures less than 60°C (140 °F) did not achieve lethal results, suggesting that using hot water alone to kill EWM is not a feasible strategy.

Thus, high pressure washing (but not necessarily hot water) or visual inspection stations located at and/or near boat launching sites hold promise for reducing the transport of EWM propagules into or out of a water body.

Spread prevention at the landscape scale: “Smart prevention”

Although it is well-established that prevention is a more cost-effective strategy for invasive species than eradication (Leung et al. 2002, Lodge et al. 2006), preventative measures tend to be implemented at the scale of a single water body (e.g. boat stewards at Lake George) rather than in a coordinated way across the landscape. Vander Zanden and Olden (2008) suggest that a “smart” preventative approach should integrate landscape-level thinking with a science-based prioritization scheme.

Vander Zanden and Olden (2008) suggest assessing the vulnerability of sites across the landscape to aquatic invasive species, by specifically considering the three stages (or filters) of species invasion – introduction, establishment, and impact (*Figure 10*). Understanding transport vectors and propagule pressure is essential in determining whether a species will pass through the first filter and arrive at a new site. Even if a species has many introduction opportunities, it may not be able to establish a self-sustaining population if the environmental conditions at the site are not conducive. Thus, the second filter considers ecological conditions necessary for survival and establishment of the aquatic invader. Finally, the third filter considers the impact of the aquatic invader on the native biota or ecosystem, e.g. consequences to food webs, threats to rare or endangered species, or effects to ecosystem services.

Vander Zanden and Olden (2008) recommend using GIS and ecological models to map and predict invasion vulnerability of sites across the landscape.

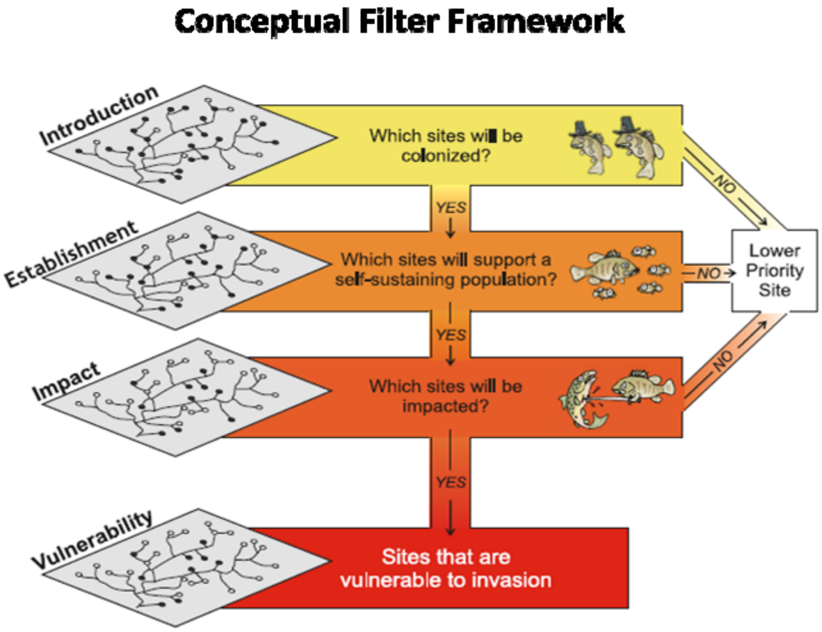


Figure 10. Filter framework for assessing the vulnerability of a water body to aquatic species invasion, as presented in Vander Zanden and Olden (2008).

How might this model of “smart prevention” at the landscape scale be employed for EWM (or other aquatic invasive species) in New York State?

The keys to Vander Zanden and Olden’s approach are integrating knowledge about the primary pathways of introduction, the ecological conditions that promote establishment, and the relative impacts an invader will have across the landscape to assess site vulnerability to invasion. This information, in turn, can be used to target management actions and prevention efforts towards the water bodies most vulnerable to invasion. Below I outline the relevant research and existing tools/information that would inform the application of a “smart prevention” framework for Eurasian watermilfoil in NYS (*Figure 11*).

For Eurasian watermilfoil, recreational boating has been implicated as the primary pathway of introduction, and thus, this pathway has been the focus of models built to predict EWM colonization and spread (Drury and Rothlisberger 2008, Rothlisberger and Lodge 2011). With data about the number of boaters in each county, the location of boat ramps, and a rich, 17-year data set of the EWM invasion history of lakes in Wisconsin as input, Rothlisberger and Lodge (2011) used gravity models to predict EWM propagule pressure and probability of colonization. Yet, they were unable to accurately predict colonization, specifically where and when EWM would invade new sites, something that had been achieved using a similar approach with other aquatic invasive species (Leung et al. 2004, Leung et al. 2006). Rothlisberger and Lodge (2011) suggest that gravity models may have predictive limitations when modeling a scenario where there are few invaded lakes serving as propagule sources and many uninvaded sites with potential for colonization (as was the case at the time for EWM in WI). Additionally, they suggest that changes in boater behavior (cleaning practices) and protective interventions (boat stewards) over time may also affect the model's predictive ability, and might be considered in future iterations of the model.

Gravity models are just one way to predict locations where propagule pressure from a particular aquatic invasive species might be greatest. As evident from Rothlisberger and Lodge (2011), this method may have some limitations. Vander Zanden and Olden (2008) present a case study on rainbow smelt in Wisconsin where they assume the arrival of the invasive smelt is based on the presence and number of boat landings. Consequently, they create a decision rule where lakes with more than one boat launch are the most likely candidates for smelt introduction; thus the number of lakes under consideration in their vulnerability assessment is reduced from 5000 to 1350, without any models. A similarly simple approach might be useful for EWM in New York.

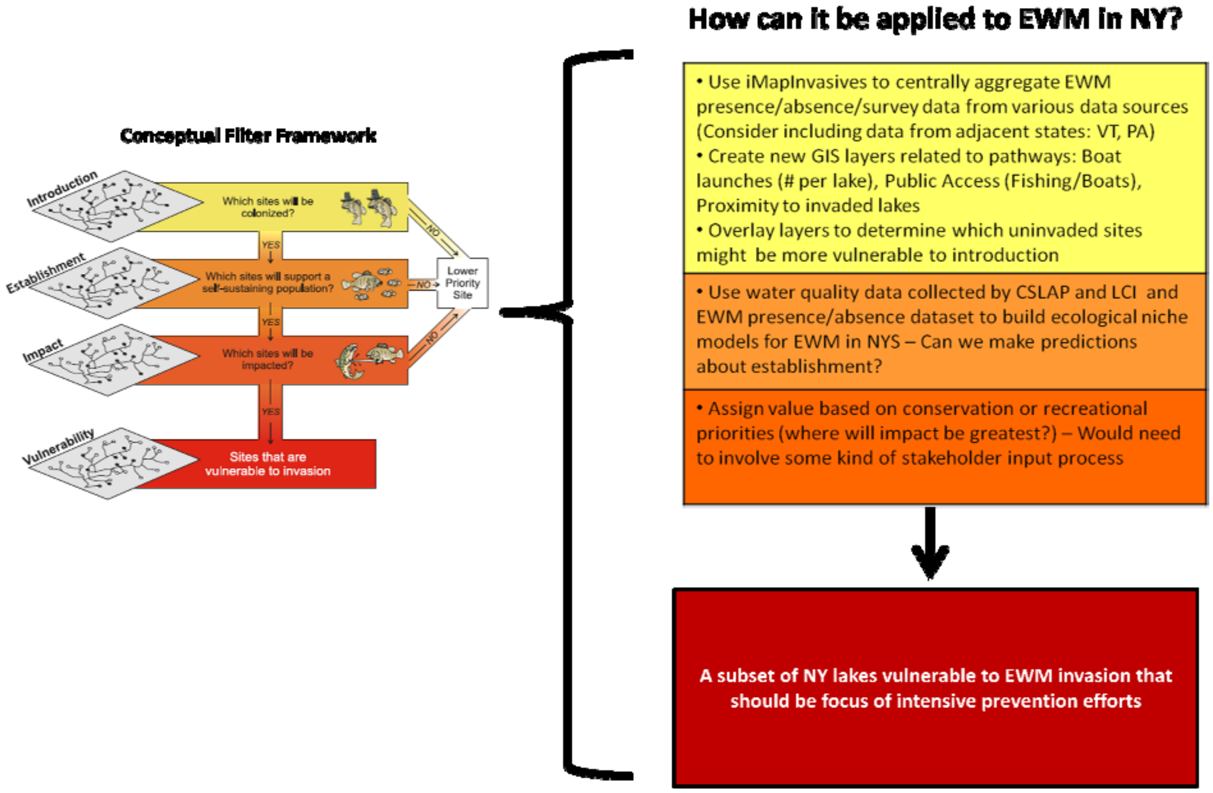
Statistical modeling efforts have also focused on the second filter – the physical, chemical and biological traits of lakes that may promote establishment of EWM (Madsen 1998, Buchan and Padilla 2000, Van den Berg et al. 2003, Roley and Newman 2008). Taken together, these studies suggest several consistent trends (*Table 4*), but no definitive thresholds or rules governing where EWM can establish. For example, although EWM can grow in lakes with a range of alkalinities and trophic conditions, it seems to occur more frequently in lakes with harder water (moderate to high alkalinity) that are meso-- to moderately eutrophic (as evident by TSI, Secchi depths, tubidity). Of course, these studies indicate correlation, not causation, and were often limited in geographic scope. Additionally, the importance of physical and chemical factors vs. transport or human activity factors in the predictive models depended on whether the study was located in a highly invaded area (WI, little significance, Buchan & Padilla 2000) or an area where EWM was less prevalent (MN, great significance, Roley & Newman 2008). To date no such predictive modeling exercise has been conducted with a data set from New York State although water quality data sets exist for a number of lakes (e.g. CSLAP and LCI data).

Table 4. Studies that use physical, chemical, and biological traits of lakes to predict risk of Eurasian watermilfoil invasion.

Reference	Nature of Study	Data Location	Significant Predictive Factors (Total # considered)	Most at Risk of Invasion
Madsen 1998	Literature Review	VT, NY, MI, WI, MN, WA, OR, AL, Ont., BC	Total water column phosphorous Carlson's Trophic State Index (7)	20-60 µg L^{-1} 45-65 TSI
Buchan & Padilla 2000	Logistic regression	WI	%Forest Cover (associated w/ Dissolved Inorganic Carbon) Alkalinity (18)	Less cover Higher alkalinity (Harder)
Van den Berg et al. 2003	Logistic regression	Netherlands	Water depth Turbidity (4)	Deeper Intermediate
Roley & Newman 2008	Linear discriminant function analysis & logistic regression	MN	**Distance to nearest invasion** **Duration of nearest invasion** Lake area Secchi depth Alkalinity Maximum depth (8)	Closer Longer Large lakes Moderate Moderate to high (Harder) Deep lakes

Prioritizing management actions based on the negative impact of invasive species, the third filter, can be challenging (Parker et al. 1999). As discussed in *Box 1*, our knowledge about the ecological impact of EWM is somewhat limited geographically and temporally. Moreover, few studies have collected baseline data prior to invasion and monitored changes in ecological communities and processes over time, in the absence of management intervention. The data we do have suggest that EWM generally has negative consequences on macroinvertebrates, native plants, and fish, but we have yet to study impacts in a way that allows us to rank or prioritize sites for management according to impact magnitude or its potential vulnerability to EWM impact. We currently operate under a framework that all EWM invasions will have a negative ecological impact, making any prioritization efforts difficult.

As Vander Zanden and Olden (2008) point out, however, the third filter in the framework for “smart prevention” could also consider impact in terms of economics and human values, rather than ecological impacts. Significant agency attention and resources have been dedicated to EWM management because the plant detrimentally impacts recreation, impeding boating, swimming and angling as well as generally interferes with the aesthetic quality of water bodies (which in turn negatively affects property values). Identifying those uninvaded lakes that have high recreational value may be a useful approach for determining where to prioritize spread-prevention efforts, and so would identifying uninvaded lakes with high conservation value. Given value assessment is a subjective endeavor, using a formal stakeholder engagement process, like that associated with adaptive management (Williams et al. 2007), would be advisable.



How can it be applied to EWM in NY?

- Use iMapInvasives to centrally aggregate EWM presence/absence/survey data from various data sources (Consider including data from adjacent states: VT, PA)
- Create new GIS layers related to pathways: Boat launches (# per lake), Public Access (Fishing/Boats), Proximity to invaded lakes
- Overlay layers to determine which uninvaded sites might be more vulnerable to introduction

- Use water quality data collected by CSLAP and LCI and EWM presence/absence dataset to build ecological niche models for EWM in NYS – Can we make predictions about establishment?

- Assign value based on conservation or recreational priorities (where will impact be greatest?) – Would need to involve some kind of stakeholder input process

A subset of NY lakes vulnerable to EWM invasion that should be focus of intensive prevention efforts

Figure 11. The filter framework (Vander Zanden and Olden 2008) as potentially applied to EWM in NYS.

Offense or defense?

With landscape-level knowledge of the vulnerability of water bodies to EWM invasion (*Figure 11*), aquatic managers can more strategically implement spread prevention efforts. But does it make more sense to invest resources (boat stewards, intensive monitoring and EDRR) directly at these vulnerable, uninvaded lakes in the name of protection from EWM (*defense*) or is better protection conferred on vulnerable lakes by directing resources towards the containment of EWM at infested water bodies (*offense*)?

Ecological models provide some insight regarding the answer to this question. Drury & Rothlisberger (2008), using simple deterministic models influenced by island biogeography and epidemiology, suggest that the answer depends on how many lakes in the landscape are invaded. They report that early in an invasion (when less than half of the water bodies are invaded), it's more effective to invest in the containment of source populations (offense). When the number of invaded water bodies increases to greater than half, Drury & Rothlisberger (2008) recommend that the strategy should shift to defense, directing efforts towards protection of uninvaded sites. Using a landscape-level approach but with a different type of model (gravity), Rothlisberger and Lodge (2011) offer further support for the offensive strategy regarding EWM; output from their models indicate that containing EWM at invaded lakes reduced the probability of EWM spread more so than protective efforts at uninvaded sites.

The offense-first approach is consistent with theoretical evidence from the terrestrial plant literature, particularly influential work by Moody and Mack (1988). In terrestrial invasive plant management, the tendency is to manage the largest, most obvious invaded patches first. Yet, Moody and Mack (1988) demonstrate that treating/eradicating newly established, isolated satellite populations (the "nascent foci") more effectively reduces the spread of an invasive species than managing the larger patches. Thus, since eradication is nearly impossible once EWM has established, halting EWM spread from lakes where it is well-established prevents its introduction to new satellite foci and slows spread across the landscape.

Moving Forward: Recommendations for New York State

To date, the approach to Eurasian watermilfoil invasions in New York's water bodies has been largely focused on management within an individual water body. Lake residents, natural resource stewards, and government agencies have expended significant effort and resources to reduce EWM populations (at least temporarily) in their respective lakes, and then must remain engaged in a maintenance management situation for perpetuity. Eradication is never achieved, and EWM continues to be detected in new water bodies each year. Stewards are placed at boat ramps in the name of spread prevention, yet their coverage is spotty and dependent on available resources from their sponsoring organization.

A coordinated, landscape level approach is needed to 1) protect EWM-free water bodies from invasion and 2) guide the investment of limited state resources towards EWM management projects that will have the most impact. Although it is well-established that prevention is a more cost-effective strategy for invasive species than eradication (Leung et al. 2002, Lodge et al. 2006), a coordinated "smart prevention” strategy (sensu Vander Zanden and Olden 2008) has not to-date been implemented in New York State. Such an approach would also lend itself to other aquatic invasive species in addition to EWM.

As outlined above (and in *Figure 11*), there are now data and infrastructure in place (e.g., iMapInvasives, PRISMs, boater movement data from some Adirondack lakes) to facilitate the implementation of a landscape prevention strategy. Ideally, a working group – comprised of EWM biologists, GIS and modeling experts, lake stakeholders (e.g., residents, lake associations, state agency personnel), and a facilitator familiar with structured decision-making – would be convened to identify lakes most vulnerable to EWM invasion in NYS and develop a prevention strategy. Perhaps piloting this approach within a smaller landscape, like the Adirondack PRISM, makes the most sense.

At the very least, identifying infested, high-boater traffic lakes and installing boat stewards and inspection stations at those locations would go a long way towards an “offensive” spread prevention strategy. To this end, it would be helpful to have a statewide inventory of where/when boat stewards are currently employed. This information, overlaid with records of infested waters and boater traffic/number of boat launches per lake, could be used to identify water bodies that should be prioritized for coverage as state resources/grant funding opportunities become available.

State support of control and maintenance management activities at EWM-infested water bodies should take a lower priority than prevention efforts. If financial support for these activities should continue (via “eradication” grants or some other program), grantees should be strongly encouraged, perhaps required, to take a more scientifically rigorous approach, e.g., collecting pre- and post- treatment data, monitoring post-treatment effects for longer than 1 year, pairing control and treatment water bodies, publishing outcomes of work in scientific journals or presenting at scientific conferences. Investments in research related to EWM management should be directed towards the insect biological control programs (which hold the most promise for long-term suppression of EWM) as well as long-term studies that evaluate the dynamics of EWM populations left unmanaged.

Although eradication of Eurasian watermilfoil in NYS is an unattainable goal, we do have an opportunity to slow its spread throughout the landscape. Doing so will require leadership and coordination among many stakeholders, particularly NYS DEC and the NY Invasive Species Council, as well as a bold shift in our thinking and priorities.

Literature Cited

Adamec, L. and S. Husak. 2002. Control of Eurasian Watermilfoil in NNR Brehynky Fishpond near Doksy,Czech Republic. Journal of Aquatic Plant Management **40**:45-46.

Aiken, S. G., P. R. Newroth, and I. Wile. 1979. The biology of Canadian weeds. 34. *Myriophyllum spicatum* L. Canadian Journal of Plant Science **59**:201-215.

Bain, M. 1993. Assessing impacts of introduced aquatic species: Grass carp in large systems. Environmental Management **17**:211-224.

Blossey, B. 1999. Before, during and after: The need for long-term monitoring in invasive plant species. Biological Invasions **1**:301-311.

Blossey, B. 2010. Personal communication.

Blumer, D. L., R. M. Newman, and F. K. Gleason. 2009. Can Hot Water Be Used to Kill Eurasian Watermilfoil? Journal of Aquatic Plant Management **47**:122-127.

Bowman, J. A. and K. E. Mantai. 1993. Submersed aquatic plant communities in western New York: 50 years of change. Journal of Aquatic Plant Management **31**:81-84.

Boylen, C. M., N. Mueller, and S. A. Kishbaugh. 2011. The costs of aquatic plant management in New York State. 51st Annual Meeting of the Aquatic Plant Management Society, Baltimore, MD.

Boylen, C. W., L. W. Eichler, J. S. Bartkowski, and S. M. Shaver. 2006. Use of Geographic Information Systems to monitor and predict non-native aquatic plant dispersal through north-eastern North America. Hydrobiologia **570**:243-248.

Boylen, C. W., L. W. Eichler, and J. D. Madsen. 1999. Loss of native aquatic plant species in a community dominated by Eurasian watermilfoil. Hydrobiologia **415**:207-211.

Boylen, C. W., L. W. Eichler, and J. W. Sutherland. 1996. Physical control of Eurasian watermilfoil in an oligotrophic lake. Hydrobiologia **340**:213-218.

Buchan, L. A. J. and D. K. Padilla. 2000. Predicting the likelihood of Eurasian watermilfoil presence in lakes, a macrophyte monitoring tool. Ecological Applications **10**:1442-1455.

Buckingham, G. R. and B. M. Ross. 1981. Notes on the biology and host specificity of *Acentria nivea* (=*Acentropus niveus*). Journal of Aquatic Plant Management **19**:32-36.

Carignan, R. and J. Kalff. 1980. Phosphorous sources for aquatic weeds: Water or sediments? Science **207**:987-989.

Carpenter, S. R. 1980. The decline of *Myriophyllum spicatum* in a eutrophic Wisconsin lake. Canadian Journal of Botany **58**:527-535.

Carpenter, S. R. 1989. Replication and Treatment Strength in Whole-Lake Experiments. Ecology **70**:453-463.

Carpenter, S. R. and D. M. Lodge. 1986. Effects of submersed macrophytes on ecosystem processes. Aquatic Botany **26**:341-370.

Cazenovia, T. o. 2009. Final Environmental Impact Statement: Propose Application of Triclopyr (Renovate OTF) (http://townofcazenovia.org/content/Generic/View/28:field=documents;/content/Documents/File/291.pdf). Cazenovia, NY.

Cheruvelil, K. S., P. A. Soranno, J. D. Madsen, and M. J. Roberson. 2002. Plant architecture and epiphytic macroinvertebrate communities: the role of an exotic dissected macrophyte. Journal of the North American Benthological Society **21**:261-277.

Christopher, S. V. and K. T. Bird. 1992. The effects of herbicides on development of *Myriophyllum spicatum* L. cultured in vitro. Journal of Environmental Quality **21**:203-207.

Cock, M. J. W., H. L. Hinz, G. Grosskopf, and P. Häfliger. 2008. Development of a biological control porgram for Eurasian watermilfoill (*Myriophyllum spicatum*). ERDC/EL TR-08-22, U.S. Army Engineer Research and Development Center, Vicksburg, MS.

Conover, G., R. Simmonds, and M. Whalen, editors. 2007. Management and control plan for bighead, black, grass, and silver carps in the United States. Asian Carp Working Group, Aquatic Nuisance Species Task Force, Washington, D.C.

Conservation, N. D. o. E. 2010. Section 305(b) New York State Clean Lakes Assessment. Page 33. NYS Department of Environmental Conservation, Albany, NY.

Cooke, G. D., A. B. Martin, and R. E. Carlson. 1990. The effect of harvesting on macrophyte regrowth and water quality in LaRue Reservior, Ohio. Journal of Iowa Academy of Science **97**.

Couch, R. and E. Nelson. 1985. *Myriophyllum spicatum* in North America. Pages 8-18 *in* First international symposium on watermilfoil (*Myriophyllum spicatum*) and related Haloragaceae species. Aquatic Plant Management Society, Vancouver, British Columbia.

Creed, R. P. 1998. A biogeographic perspective on Eurasian watermilfoil declines: Additional evidence for the role of herbivorous weevils in promoting declines? Journal of Aquatic Plant Management **36**:16-22.

Creed, R. P., Jr. and S. P. Sheldon. 1995. Weevils and Watermilfoil: Did a North American Herbivore Cause the Decline of an Exotic Plant? Ecological Applications **5**:1113-1121.

Creed, R. P. and S. P. Sheldon. 1993. The effect of feeding by a North American weevil, Euhrychiopsis lecontei, on Eurasian watermilfoil (Myriophyllum spicatum). Aquatic Botany **45**:245-256.

Crowell, W. 1999. Minnesota DNR tests the use of 2,4-D in managing Eurasian watermilfoil. Aquatic Nuisance Species Digest **3**:42-46.

Crowell, W., N. Troelstrup, L. Queen, and J. Perry. 1994. Effects of harvesting on plant communities dominated by Eurasian watermilfoil in Lake Minnetonka, MN. Journal of Aquatic Plant Management **32**:56-60.

Cuda, J. P., R. Charudattan, M. J. Grodowitz, R. M. Newman, J. F. Shearer, M. L. Tamayo, and B. Villegas. 2008. Recent advances in biological control of submersed aquatic weeds. Journal of Aquatic Plant Management **46**:15-32.

Dean, J. 2011. Personal communication.

DeBolt, E. and K. Rohne. 2010. Lake George Watershed Lake Steward Program End of Season Report 2010. The Lake George Association, Lake George, NY.

Deming, A. L. 2002. Augmentation of the Aquatic Moth, *Acentria ephemerella*, for Control of Eurasian watermilfoil, *Myriophyllym spicatum*: Final Report to the Lake Champlain Basin Program (http://www.lincolnpond.org/2002-LP-report-Aquatic-Moth-Final.doc). Essex County Cornell Cooperative Extension, Elizabethtown, NY.

Dibble, E. D. and S. L. Harrel. 1997. Largemouth bass diets in two aquatic plant communities. Journal of Aquatic Plant Management **35**:74-78.

Dibble, E. D. and K. Kovalenko. 2009. Ecological Impact of Grass Carp: A Review of the Available Data. Journal of Aquatic Plant Management **47**:1-15.

Drury, K. L. S. and J. D. Rothlisberger. 2008. Offense and defense in landscape-level invasion control. Oikos **117**:182-190.

Eichler, L. W. 2010. Annual Report - Darrin Fresh Water Institute Aquatic Plant Identification Program. Darrin Fresh Water Institute, Bolton Landing, NY.

Eichler, L. W. 2011. Personal communication.

Eichler, L. W., R. T. Bombard, B. J. W. Sutherland, and C. M. Boylen. 1995. Recolonization of the littoral zone by macrophytes following the removal of benthic barrier material. Journal of Aquatic Plant Management **33**:51-54.

Eichler, L. W., R. T. Bombard, J. W. Sutherland, and C. W. Boylen. 1993. Suction harvesting of Eurasian watermilfoil and its effect on native plant communities. Journal of Aquatic Plant Management **31**:144-148.

Eichler, L. W., E. A. Howe, and C. W. Boylen. 2001. The use of stream delta surveillance as a tool for early detection of Eurasian watermilfoil. Journal of Aquatic Plant Management **39**:79-82.

Eiswerth, M. E., T. D. Darden, W. S. Johnson, J. Agapoff, and T. R. Harris. 2005. Input-Output Modeling, Outdoor Recreation, and the Economic Impacts of Weeds. Weed Science **53**:130-137.

Eiswerth, M. E., S. G. Donaldson, and W. S. Johnson. 2000. Potential environmental impacts and economic damages of Eurasian watermilfoil (Myriophyllum spicatum) in western Nevada and northeastern California. Weed Technology **14**:511-518.

Engel, S. 1995. Eurasian watermilfoil as a fishery management tool. Fisheries **20**:20-27.

Environmental Laboratory, U. S. A. E. R. a. D. C. 2011. Noxious and Nuisance Plant Management Information System (http://el.erdc.usace.army.mil/pmis/pmishelp.htm). U.S. Army Corps of Engineers, Vicksburg, MS.

Evans, C. A., K. M. Forrest, D. L. Kelting, and L. E. Steblen. 2011. Fragment viability and rootlet formation in Eurasian watermilfoil after desiccation. AWI 2011-02, Adirondack Watershed Institute, Paul Smith's College.

Frodge, J. D., G. L. Thomas, and G. B. Pauley. 1990. Effects of canopy formation by floating and submergent aquatic macrophytes on the water quality of two shallow Pacific Northwest lakes. Aquatic Botany **38**:231-248.

Getsinger, K. D., J. D. Madsen, T. Koschnick, and M. D. Netherland. 2002. Whole lake fluridone treatments for selective control of Eurasian watermilfoil: I. Application strategy and herbicide residues. Lake and Reservoir Management **18**:181-190.

Getsinger, K. D., M. D. Netherland, C. E. Grue, and T. J. Koschnick. 2008. Improvements in the use of aquatic herbicides and establishment of future research directions. Journal of Aquatic Plant Management **46**:32-41.

Getsinger, K. D., E. G. Turner, J. D. Madsen, and M. D. Netherland. 1997. Restoring native vegetation in a Eurasian water milfoil dominated plant community using the herbicide triclopyr. Regulated Rivers-Research & Management **13**:357-375.

Getsinger, K. D. and H. E. Westerdahl. 1984. Field evaluation of Garlon 3A (Triclopyr) and 14-ACE-B (2,4-D BEE) for the control of Eurasian watermilfoil. Miscellaneous Paper A-84-5, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Grace, J. B. and R. G. Wetzel. 1978. Production biology of Eurasian watermilfoil (Myriophyllum spicatum L.): A review. Journal of Aquatic Plant Management **16**:1-11.

Gross, E. M., C. Feldbaum, and C. Choi. 2002. High abundance of herbivorous Lepidoptera larvae (Acentria ephemerella DENIS & SCHIFFERMULLER) on submersed macrophytes in Lake Constance (Germany). Archiv Fur Hydrobiologie **155**:1-21.

Gross, E. M., R. L. Johnson, and N. G. Hairston. 2001. Experimental evidence for changes in submersed macrophyte species composition caused by the herbivore Acentria ephemerella (Lepidoptera). Oecologia **127**:105-114.

Gross, E. M., H. Meyer, and G. Schilling. 1996. Release and ecological impact of algicidal hydrolysable polyphenols in Myriophyllum spicatum. Phytochemistry **41**:133-138.

Halstead, J. M., J. Michaud, S. Hallas-Burt, and J. P. Gibbs. 2003. Hedonic analysis of effects of a nonnative invader (*Myriophyllum heterophyllum*) on New Hampshire (USA) lakefront properties. Environmental Management **32**:391-398.

Harman, W. N., L. P. Hingula, and C. E. MacNamara. 2005. Does long-term macrophyte management in lakes affect biotic richness and diversity? Journal of Aquatic Plant Management **43**:57-64.

Hobbs, R. J. and S. E. Humphries. 1995. An integrated approach to the ecology and management of plant invasions. Conservation Biology **9**:761-770.

Hofstra, D. E., J. S. Clayton, and K. D. Getsinger. 2001. Evaluation of selected herbicides for the control of exotic submerged weeds in New Zealand: II. The effects of turbidity on diquat and endothall efficacy. Journal of Aquatic Plant Management **39**:25-27.

Holmlund, E. 2011. Watershed Stewardship Program Summary of Programs and Research 2010. Adirondack Watershed Institute, Paul Smith's College, Paul Smiths, NY.

Horsch, E. J. and D. J. Lewis. 2009. The effects of aquatic invasive species on property values: Evidence from a quasi-experiment. Land Economics **85**:391-409.

Jacono, C. C. and M. M. Richerson. 2011. *Myriophyllum spicatum* (http://nas.er.usgs.gov/queries/FactSheet.aspx?speciesID=237). USGS Nonindigenous Aquatic Species Database, Gainesville, FL.

Jerde, C. L., M. A. Barnes, E. K. DeBuysser, A. Noveroske, W. L. Chadderton, and D. M. Lodge. submitted. Eurasian watermilfoil fitness loss and invasion potential following desiccation during simulated overland transport.

Jester, L. L., M. A. Bozek, D. R. Helsel, and S. P. Sheldon. 2000. Euhrychiopsis lecontei distribution, abundance, and experimental augmentations for Eurasian watermilfoil control in Wisconsin lakes. Journal of Aquatic Plant Management **38**:88-97.

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

Johnson, R., E. Gross, and N. Hairston. 1998. Decline of the invasive submersed macrophyte Myriophyllum spicatum (Haloragaceae) associated with herbivory by larvae of Acentria ephemerella (Lepidoptera). Aquatic Ecology **31**:273-282.

Johnson, R. L. 2010. Personal communication.

Johnson, R. L. 2011. Personal communication.

Johnson, R. L., J. D. Johnson, J. M. Riggs, and J. A. Toner. 2010a. Cazenovia Lake Plant Community Response to the 2009 and 2010 Application of the Herbicide Triclopyr to Control Eurasian Watermilfoil (http://townofcazenovia.org/content/Generic/View/28:field=documents;/content/Documents/File/485.pdf). Racine-Johnson Aquatic Ecologists, Ithaca, NY.

Johnson, R. L., J. D. Johnson, N. Sledziona, N. E. Pyne, J. A. Toner, and J. M. Riggs-Larson. 2008. Evaluation of the efficacy of herbivorous insects to control Eurasian watermilfoil growth in Lake Bonaparte, New York: 2008 Report.

Johnson, R. L., C. Killourhy, N. Sledziona, J. M. Riggs, and J. A. Toner. 2009. Cazenovia Lake Plant Community Response to the 2009 Application of the Herbicide Triclopyr to Control Eurasian Watermilfoil (http://townofcazenovia.org/content/Generic/View/28:field=documents;/content/Documents/File/366.pdf). Racine-Johnson Aquatic Ecologists, Ithaca, NY.

Johnson, R. L., J. A. Toner, J. D. Johnson, N. Sledziona, N. E. Pyne, J. M. Riggs, and C. Killourhy. 2010b. Evaluation of the efficacy of herbivorous insects to control Eurasian watermilfoil growth in Lake Bonaparte, New York: 2009 Report.

Johnson, R. L., P. J. Van Dusen, J. A. Toner, and N. G. Hairston. 2000. Eurasian watermilfoil biomass associated with insect herbivores in New York. Journal of Aquatic Plant Management **38**:82-88.

Kartesz, J. T. 2011. PLANTS Profile: *Myriophyllum spicatum* L. (http://plants.usda.gov/java/profile?symbol=MYSP2). USDA NRCS National Plants Data Team, Washington, DC.

Keast, A. 1984. The introduced aquatic macrophyte, *Myriophyllum spicatum*, as habitat for fish and their invertebrate prey. Canadian Journal of Zoology-Revue Canadienne De Zoologie **62**:1289-1303.

Kelting, D. L. and C. L. Laxson. 2010. Cost and effectiveness of hand harvesting to control the Eurasian watermilfoil population in Upper Saranac Lake, New York. Journal of Aquatic Plant Management **48**:1-5.

Killgore, K. J. 1984. Use of herbicide/adjuvant for the control of *Myriophyllum spicatum* L. Miscellaneous paper A-84-8, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Kimbel, J. C. and S. R. Carpenter. 1981. Effects of mechanical harvesting on *Myriophyllum spicatum* L. regrowth and carbohydrate allocation to roots and shoots. Aquatic Botany **11**:121-127.

King, R. and B. Laginhas. 2010. Lake George Integrated Aquatic Plant Management Program. Lycott Environmental Inc., Southbridge, MA.

Kirk, J. P. and R. C. Socha. 2003. Longevity and persistence of triploid grass carp stocked in the Santee Cooper reservoirs of South Carolina. Journal of Aquatic Plant Management **41**:90-92.

Koschnick, T. J., W. T. Haller, and L. Glasgow. 2006. Documentation of landoltia (*Landoltia punctata*) resistance to diquat. Weed Science **54**:615-619.

Lembi, C. A. 1996. Assessment of 2,4-D use in aquatic systems in the United States. Pages 179-194.

Leung, B., J. M. Bossenbroek, and D. M. Lodge. 2006. Boats, pathways, and aquatic biological invasions: Estimating dispersal potential with gravity models. Biological Invasions **8**:241-254.

Leung, B., J. M. Drake, and D. M. Lodge. 2004. Predicting invasions: Propagule pressure and the gravity of allee effects. Ecology **85**:1651-1660.

Leung, B., D. M. Lodge, D. Finnoff, J. F. Shogren, M. A. Lewis, and G. Lamberti. 2002. An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species. Proceedings of the Royal Society of London Series B-Biological Sciences **269**:2407-2413.

Lillie, R. A. and J. Budd. 1992. Habitat architecture of *Myriophyllum spicatum* L. as an index to habitat quality for fish nad macroinvertebrates. Journal of Freshwater Ecology **7**:113-125.

Lodge, D. M., S. Williams, H. J. MacIsaac, K. R. Hayes, B. Leung, S. Reichard, R. N. Mack, P. B. Moyle, M. Smith, D. A. Andow, J. T. Carlton, and A. McMichael. 2006. Biological invasions: Recommendations for US policy and management. Ecological Applications **16**:2035-2054.

Lyons, J. 1989. Changes in the abundance of small littoral-zone fishes in Lake Mendota, Wisconsin. Canadian Journal of Zoology **67**:2910-2916.

MacRae, I. V. and R. A. Ring. 1993. Life history of *Cricotopus myriophylli* Oliver (Diptera, Chironomidae) in the Okanagan Valley, British Columbia. Canadian Entomologist **125**:979-985.

MacRae, I. V., N. N. Winchester, and R. A. Ring. 1990. Feeding activity and host preference of the milfoil midge, *Cricotopus myriophylli* Oliver (Diptera: Chironomidae). Journal of Aquatic Plant Management **28**.

Madsen, J. D. 1997. Seasonal biomass and carbohydrate allocation in a southern population of Eurasian watermilfoil. Journal of Aquatic Plant Management **35**:15-21.

Madsen, J. D. 1998. Predicting invasion success of Eurasian watermilfoil. Journal of Aquatic Plant Management **36**:28-32.

Madsen, J. D., L. W. Eichler, and C. W. Boylen. 1988. Vegetative spread of Eurasian watermilfoil in Lake George, New York. Journal of Aquatic Plant Management **26**:47-50.

Madsen, J. D., K. D. Getsinger, and R. M. Stewart. 2002. Whole lake fluridone treatments for selective control of Eurasian watermilfoil: II. Impacts on submersed plant communities. Lake and Reservoir Management **18**:191-200.

Madsen, J. D. and D. H. Smith. 1997. Vegetative spread of Eurasian watermilfoil colonies. Journal of Aquatic Plant Management **35**:63-68.

Madsen, J. D., B. J. W. Sutherland, J. A. Bloomfield, K. M. Roy, L. W. Eichler, and C. M. Boylen. 1989. The Lake George Aquatic Plant Survey: Final Report. New York State Department of Environmental Conservation.

Madsen, J. D., J. W. Sutherland, J. A. Bloomfield, L. W. Eichler, and C. W. Boylen. 1991. The decline of native vegetations under dense Eurasian watermilfoil canopies. Journal of Aquatic Plant Management **29**:94-99.

Madsen, J. D., R. M. Wersal, K. D. Getsinger, and J. G. Skogerboe. 2010. Combinations of endothall with 2,4-D and triclopyr for Eurasian watermilfoil control. Miscellaneous paper APCRP-CC-14, US Army Engineer Research and Development Center, Vicksburg, MS.

Marko, P. D., E. M. Gross, R. M. Newman, and F. K. Gleason. 2008. Chemical profile of the North American native Myriophyllum sibiricum compared to the invasive M. spicatum. Aquatic Botany **88**:57-65.

Mayer, J. R. 1978. Aquatic weed management by benthic semi-barriers. Journal of Aquatic Plant Management **16**:31-33.

Mazzei, K. C., R. M. Newman, A. Loos, and D. W. Ragsdale. 1999. Developmental Rates of the Native Milfoil Weevil, Euhrychiopsis lecontei, and Damage to Eurasian Watermilfoil at Constant Temperatures. Biological Control **16**:139-143.

Michel, A., R. S. Arias, B. E. Scheffler, S. O. Duke, M. Netherland, and F. E. Dayan. 2004. Somatic mutation-mediated evolution of herbicide resistance in the nonindigenous invasive plant hydrilla (Hydrilla verticillata). Molecular Ecology **13**:3229-3237.

Moody, M. E. and R. N. Mack. 1988. Controlling the spread of plant invasions: The importance of nascent foci. Journal of Applied Ecology **25**:1009-1021.

Moody, M. L. and D. H. Les. 2002. Evidence of hybridity in invasive watermilfoil (Myriophyllum) populations. Proceedings of the National Academy of Sciences of the United States of America **99**:14867-14871.

Moody, M. L. and D. H. Les. 2007. Geographic distribution and genotypic composition of invasive hybrid watermilfoil (Myriophyllum spicatum x M-sibiricum) populations in North America. Biological Invasions **9**:559-570.

Mueller, N. 2011. Personal communication.

Nelson, L. S. and J. F. Shearer. 2005. 2,4-D and Mycoleptodiscus terrestris for control of Eurasian watermilfoil. Journal of Aquatic Plant Management **43**:29-34.

Nelson, L. S. and J. F. Shearer. 2008. Evaluation of Triclopyr and Mycoleptodiscus terrestris for Control of Eurasian Watermilfoil (Myriophyllum spicatum). Invasive Plant Science and Management **1**:337-342.

Netherland, M. D. and K. D. Getsinger. 1992. Efficacy of triclopyr on Eurasian watermilfoil: Concentration and exposure time effects. Journal of Aquatic Plant Management **30**:1-5.

Netherland, M. D., K. D. Getsinger, and D. R. Stubbs. 2005. Aquatic plant management: Invasive species and chemical control. Outlooks on Pest Management **16**:100-104.

Netherland, M. D., K. D. Getsinger, and E. G. Turner. 1993. Fluridone concentration and exposure time requirements for control of Eurasian watermilfoil and hydrilla. Journal of Aquatic Plant Management **31**:189-194.

Newman, R. M. 2004. Invited review - Biological control of Eurasian watermilfoil by aquatic insects: basic insights from an applied problem. Archiv Fur Hydrobiologie **159**:145-184.

Newman, R. M. and D. D. Biesboer. 2000. A decline in Eurasian watermilfoil in Minnesota associated with the milfoil weevil, *Euhrychiopsis lecontei*. Journal of Aquatic Plant Management **38**:105-111.

Newman, R. M., M. E. Borman, and S. W. Castro. 1997. Developmental Performance of the Weevil Euhrychiopsis lecontei on Native and Exotic Watermilfoil Host Plants. Journal of the North American Benthological Society **16**:627-634.

Newman, R. M., K. L. Holmberg, D. D. Biesboer, and B. G. Penner. 1996. Effects of a potential biocontrol agent, Euhrychiopsis lecontei, on Eurasian watermilfoil in experimental tanks. Aquatic Botany **53**:131-150.

Newman, R. M. and W. G. Inglis. 2009. Distribution and Abundance of the Milfoil Weevil, Euhrychiopsis lecontei, in Lake Minnetonka and Relation to Milfoil Harvesting. Journal of Aquatic Plant Management **47**:21-25.

Newroth, P. R. 1985. A review of the Eurasian water milfoil impacts and management in British Columbia. Pages 139-153 *in* Proceedings of the First International Symposium on Watermilfoil (*Myriophyllum spicatum*) and related Haloragaceae Species. Aquatic Plant Management Society, Vancouver, British Columbia.

Nichols, S. A. and R. C. Lathrop. 1994. Impact of harvesting on aquatic plant communities in Lake Wingra, Wisconsin. Journal of Aquatic Plant Management **32**:33-36.

Nichols, S. A. and B. H. Shaw. 1986. Ecological life histories of the 3 aquatic nusiance plants, *Myriophyllum spicatum*, *Potamogeton crispus* and *Elodea canadensis*. Hydrobiologia **131**:3-21.

Nicholson, S. A. 1981. Changes in submersed macrophytes in Chautauqua Lake, 1937–1975. Freshwater Biology **11**:523-530.

NYS Department of Environmental Conservation Bureau of Fisheries. 2011. Triploid grass carp in NY ponds. (http://www.dec.ny.gov/outdoor/7973.html) Albany, NY.

NYSFOLA. 2009. Diet for a Small Lake: The Expanded Guide to New York State Lake and Watershed Management. 2nd edition. New York State Federation of Lake Associations, Inc.

Olson, M. H., S. R. Carpenter, P. Cunningham, S. Gafny, B. R. Herwig, N. P. Nibbelink, T. Pellett, C. Storlie, A. S. Trebitz, and K. A. Wilson. 1998. Managing macrophytes to improve fish growth: A multi-lake experiment. Fisheries **23**:6-12.

Parker, I. M., D. Simberloff, W. M. Lonsdale, K. Goodell, M. Wonham, P. M. Kareiva, M. H. Williamson, B. Von Holle, J. E. Byers, and L. Goldwasser. 1999. Impact: toward a framework for understanding the ecological effects of invaders. Biological Invasions **1**:3-19.

Parsons, J. K., K. Hamel, J. D. Madsen, and K. D. Getsinger. 2001. The use of 2,4-D for selective control of an early infestation of Eurasian watermilfoil in Loon Lake, Washington. Journal of Aquatic Plant Management **39**:117-125.

Parsons, J. K., K. S. Hamel, S. L. O'Neal, and A. W. Moore. 2004. The impact of endothall on the aquatic plant community of Kress Lake, Washington. Journal of Aquatic Plant Management **42**:109-114.

Petty, D. G., K. D. Getsinger, and K. B. Woodburn. 2003. A review of the aquatic environmental fate of triclopyr and its major metabolites. Journal of Aquatic Plant Management **41**:69-75.

Pine, R. T. and L. W. J. Anderson. 1991. Plant preferences of triploid grass carp. Journal of Aquatic Plant Management **29**:80-82.

Poovey, A. G., K. D. Getsinger, J. G. Skogerboe, T. J. Koschnick, J. D. Madsen, and R. M. Stewart. 2004. Small-plot, low-dose treatments of triclopyr for selective control of Eurasian watermilfoil. Lake and Reservoir Management **20**:322-332.

Poovey, A. G., J. G. Slade, and M. D. Netherland. 2007. Susceptibility of Eurasian watermilfoil (Myriophyllum spicatum) and a milfoil hybrid (M. spicatum x M. sibiricum) to Triclopyr and 2,4-D amine. Journal of Aquatic Plant Management **45**:111-115.

Quantitative Environmental Analysis, L. 2008. Black Lake Eurasian Watermilfoil Management Plan (http://www.weedinfo.blacklakeny.com/FINAL_Black_Lake_milfoil_plan_07_14_08-1.pdf). Liverpool, NY.

Rawls, C. K. 1975. Mechanical Control of Eurasian Watermilfoil in Maryland with and without 2,4-D Application. Chesapeake Science **16**:266-281.

Reeves, J. L., P. D. Lorchi, M. W. Kershner, and M. A. Hilovsky. 2008. Biological Control of Eurasian watermilfoil by Euhrychiopsis lecontei: Assessing Efficacy and Timing of Sampling. Journal of Aquatic Plant Management **46**:144-149.

Roley, S. S. and R. M. Newman. 2006. Developmental performance of the milfoil weevil, Euhrychiopsis lecontei (Coleoptera : Curculionidae), on northern watermilfoil, Eurasian watermilfbil, and hybrid (Northern x Eurasian) watermilfoil. Environmental Entomology **35**:121-126.

Roley, S. S. and R. M. Newman. 2008. Predicting Eurasian watermilfoil invasions in Minnesota. Lake and Reservoir Management **24**:361-369.

Rothlisberger, J. D., W. L. Chadderton, J. McNulty, and D. M. Lodge. 2010. Aquatic Invasive Species Transport via Trailered Boats: What Is Being Moved, Who Is Moving It, and What Can Be Done. Fisheries **35**:121-132.

Rothlisberger, J. D. and D. M. Lodge. 2011. Limitations of gravity models in predicting the spread of Eurasian watermilfoil. Conservation Biology **25**:64-72.

Senseman, S. A., editor. 2007. Herbicide Handbook. Ninth Edition edition. Weed Science Society of America, Lawrence, KS.

Shearer, J. F. 2010. A Historical Perspective of Pathogen Biological Control of Aquatic Plants. Weed Technology **24**:202-207.

Sheldon, S. P. and L. M. O'Bryan. 1996. The effects of harvesting Eurasian watermilfoil on the aquatic weevil *Euhrychiopsis lecontei*. Journal of Aquatic Plant Management **34**:76-77.

Skogerboe, J. G. and K. D. Getsinger. 2001. Endothall species selectivity evaluation: Southern latitude aquatic plant community. Journal of Aquatic Plant Management **39**:129-135.

Skogerboe, J. G. and K. D. Getsinger. 2002. Endothall species selectivity evaluation: Northern latitude aquatic plant community. Journal of Aquatic Plant Management **40**:1-5.

Skogerboe, J. G., K. D. Getsinger, and L. A. M. Glomski. 2006. Efficacy of diquat on submersed plants treated under simulated flowing water conditions. Journal of Aquatic Plant Management **44**:122-125.

Smith, C. S. and J. W. Barko. 1990. Ecology of Eurasian watermilfoil. Journal of Aquatic Plant Management **28**:55-64.

Smith, C. S. and G. D. Pullman. 1997. Experiences Using Sonar® A.S. Aquatic Herbicide in Michigan. Lake and Reservoir Management **13**:338 - 346.

Smith, H. 2011. Personal communication.

Surprenant, L. 2001. Experiences with using grass carp for aquatic vegetation control in DEC Region 3 with emphasis on Walton Lake. NYS Department of Environmental Conservation. Albany, NY.

Sutter, T. J. and R. M. Newman. 1997. Is predation by sunfish (*Lepomis* spp.) an important source of mortality for the Eurasian watermilfoil biological control agent *Euhrychiopsis lecontei*? Journal of Freshwater Ecology **12**:225-234.

Sutton, D. L. and V. V. Vandiver. 1998. Grass carp: A fish for biological management of hydrilla and other aquatic weeds in Florida. UF/IFAS Agricultural Experiment Station Bulletin No. 867.

Unmuth, J. M. L., D. J. Sloey, and R. A. Lillie. 1998. An evaluation of close-cut mechanical harvesting of Eurasian watermilfoil. Journal of Aquatic Plant Management **36**:93-100.

Valley, R. D. and M. T. Bremigan. 2002. Effects of macrophyte bed architecture on largemouth bass foraging: Implications of exotic macrophyte invasions. Transactions of the American Fisheries Society **131**:234-244.

Van den Berg, M. S., W. Joosse, and H. Coops. 2003. A statistical model predicting the occurrence and dynamics of submerged macrophytes in shallow lakes in the Netherlands. Hydrobiologia **506-509**:611-623.

Vander Zanden, M. J. and J. D. Olden. 2008. A management framework for preventing the secondary spread of aquatic invasive species. Canadian Journal of Fisheries and Aquatic Sciences **65**:1512-1522.

Wagner, T. and C. M. Falter. 2002. Response of an aquatic macrophyte community to fluctuating water levels in an oligotrophic lake. Lake and Reservoir Management **18**:52-65.

Ward, D. M. and R. M. Newman. 2006. Fish predation on Eurasian watermilfoil (Myriophyllum spicatum) herbivores and indirect effects on macrophytes. Canadian Journal of Fisheries and Aquatic Sciences **63**:1049-1057.

Water, N. D. o. 2005. A primer on aquatic plant management in New York State (Draft). Page 63. NYSDEC Division of Water, Albany, NY.

Weber, J. B., P. W. Perry, and R. P. Upchurch. 1965. The influence of temperature and time on the adsorption of paraquat, diquat, 2,4-D and prometone by clays, charcoal, and an anion-exchange resin. Soil Science Society of America Journal **29**:678-688.

Weldy, T. and D. Werier. 2011. *Myriophyllum sibiricum* (http://newyork.plantatlas.usf.edu/Plant.aspx?id=1582). New York Flora Atlas. New York Flora Association, Albany, NY.

Williams, B. K., R. C. Szaro, and C. D. Shapiro. 2007. Adaptive Management: The U.S. Department of the Interior Technical Guide. Washington, D.C.

Wilson, K. and S. R. Carpenter. 1997. Making the weedline work for your lake. Wisconsin Natural Resources Magazine **21**:4-8.

Wilson, S. J. and A. Ricciardi. 2009. Epiphytic macroinvertebrate communities on Eurasian watermilfoil (Myriophyllum spicatum) and native milfoils Myriophyllum sibericum and Myriophyllum alterniflorum in eastern North America. Canadian Journal of Fisheries and Aquatic Sciences **66**:18-30.

Zhang, C. and K. J. Boyle. 2010. The effect of an aquatic invasive species (Eurasian watermilfoil) on lakefront property values. Ecological Economics **70**:394-404.