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Invasive European frogbit (*Hydrocharis morsus-ranae* L.) in North America: an updated review 2003–16

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Abstract

Aims
European frogbit (*Hydrocharis morsus-ranae* L.) is an aquatic plant originating from Europe that has emerged as an invasive species, spreading in the USA and Canada since it was first brought to North America in 1932. It can now be found in many water bodies, from small ponds and long rivers to large lakes such as Lake Ontario and Lake Erie. The continuous spread of this species indicates its success as an invasive species despite legislative attempts to limit its distribution. Catling *et al.* (Catling PM, Miltrow G, Haber E, *et al*. (2003) The biology of Canadian weeds. 124. *Hydrocharis morsus-ranae* L. *Can J Plant Sci* 83:1001–16) wrote a thorough review about this invasive species in North America. Our review aims for a compilation of the most recent available data and recent studies on *H. morsus-ranae* L. and focuses primarily on its environmental uses, ecological impacts and management. The purpose of this review is to offer an organized and updated report on European frogbit that can be used towards future studies with the goal of eradicating this invasive species and providing insights on management of other invasive plants.

Important Findings
Our findings reveal that European forgbit's ecological effects on other species and the invaded environment were shown to be less harmful than previously feared. European frogbit had negative impacts on native plants and reduced dissolved oxygen concentration. However, water chemistry, phytoplankton and zooplankton communities were actually not affected by European frogbit. For fungi, bacteria and macroinvertebrates, studies have showed complex and sometimes conflicting results. We also specifically discussed the new method to control this species using shading and the more recent studies on biological control. Shading with a shade cloth has been shown to effectively remove European frogbit and had minor environmental effects. However, using biological control to combat the spread of the invasive frogbit seems not as successful as we wished.

Keywords: European frogbit, invasive plants, ecological impacts, biological control, shading

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INTRODUCTION

The spread and impact of exotic species, especially plant species, has become a global conservation concern (Catling *et al.* 2003; Mack *et al.* 2000; Redmond and Stout 2018). About 6–10% of introduced aquatic plants in the world are considered seriously invasive (Houlahan and Findlay 2004; Williamson 1996). Introduced plants become invasive mainly due to their biological and ecological traits such as genotypic richness, germination timing, drought resistance and various dispersal patterns (e.g. Antunes *et al.* 2018; Clark *et al.* 2018; Collins *et al.* 2018; Gioria *et al.* 2018). European frogbit (*Hydrocharis morsus-ranae* L.), a native of Europe, was first brought to North America in 1932 at the Ottawa botanic garden (intentional introduction) and later noticed as an escapee in 1939 in the Rideau Canal (Dore 1968). This species then spread into the Ottawa and the St. Lawrence Rivers (Dore 1968) and into the USA in 1974 probably by seed and hibernacula (Roberts *et al.* 1981). In Canada, European frogbit is currently found in the Rideau and Ottawa River systems, the St. Lawrence River, Lake Ontario, Lake Erie, the Kawartha Lakes and other lakes and rivers in south central and south western Ontario (Catling and Porebski 1995; Catling *et al.* 2003). In the USA, it is present in Michigan, Wisconsin, Vermont, New York and Washington states (Catling *et al.* 2003). Recently, it was found further south and has been observed in several places in New York State - Sterling Creek in Cayuga County and the Audubon Center and Sanctuary in Southern Chautauqua County (Zhu *et al.* 2008). European frogbit has become a source of concern due to its high invasion success (Catling and Porebski 1995; Catling *et al.* 2003; Zhu *et al.* 2008). For this reason, it is illegal to buy, sell or transport the European frogbit in some states (e.g. Washington and Minnesota).

Catling *et al.* (2003) wrote a review on European frogbit more than a decade ago. It discussed topics such as biological features of frogbit (e.g. growth and reproduction), habitat, records of its invasion and distribution, the possible beneficial and harmful effects of its invasion, along with its response to attempted control agents. When discussing various control methods, like chemicals or hand-pulling, biological control methods could be the most promising solution when it comes to containing the spread of European frogbit (Catling *et al.* 2003), making further studies on the use and effectiveness of those potential biological control agents more necessary.

Our review aims to provide new information (either recent studies from 2003 to 2016 or older studies that not covered in the 2003 review), regarding the biological attributes of European frogbit and detail the impacts European frogbit has had on invaded ecosystems. Findings on the effectiveness of various control methods are particularly mentioned, including certain biological control methods that were not previously discussed.

In this article, we report reproduction, tissue content and dispersal in regards to the European frogbit's biology and ecology. Light, temperature, water, sediment and trophic state are also discussed in relation to the environment from which it grows. Findings on the ecological impacts include its effects on dissolved oxygen levels, native species and human activities, while summary of management discusses some of the more common techniques of invasion control and management.

BIOLOGY AND ECOLOGY

European frogbit (*H. morsus-ranae* L.) belongs to the Hydrocharitaceae family. It is a perennial free-floating aquatic plant with leathery heart-shaped leaves and well-developed roots. It has a closed root meristem (the inner cell layer of the cap complex forms the epidermis) and its trichoblast develops from the proximal sister cell (Clowes 2000). The lacunae of *Hydrocharis* plants are the product of anticlinal radial file cell division and cell death (Seago *et al.* 2005). Hydrocharitaceae also possess packet lysigeny (Seago *et al.* 2005). While it lacks diaphragms, Hydrocharitaceae possess aerenchyma tissue in a honeycomb pattern (Seago *et al.* 2005). It belongs to a group of plant species where certain trans-splicing events had occurred more recently in comparison to other plant species such as Nymphaeales and Chloranthaceae (Qiu and Palmer 2004).

European frogbit is often found in still, slow-moving shallow waters such as ponds, ditches, wetlands, marshes and swamps, backwaters, beaver dams, canals, sluggish creeks and also wind sheltered and wave-protected areas of lakes and rivers (Catling *et al.* 2003; Cook and Lüönd 1982). As a floating leaf species, frogbit (as well as emergent leaf species) contain more phenolics than species with submerged leaves (Smolders *et al.* 2000). This could be because such species may be more susceptible than submerged species to predation, since phenolics appear to help protect plants from pathogens and herbivores (Lodge 1991). Studies done by Schoelynck *et al.* (2010) also show that aquatic plants like European frogbit tend to spend their energy predominantly for cellulose or BSi (biogenic silica) with lignin.

Reproduction

European frogbit is dioecious and has different reproductive structures in male and female individuals (Tuschnjakowa 1929). It is a herbaceous aquatic plant, flowers (small white flowers) erratically and seldom fruits (Cook and Lüönd 1982). In fact, reproduction by seeds is rarely reported and it reproduces vegetatively through development of stolon buds and turions (Catling *et al.* 2003). The production of gemmae or bulbils constitutes a normal mode of propagation in *H. morsus-ranae* (Lanzoni 1928). The turions develop in the fall, separate from the plant and sink to the bottom where they overwinter, and then rise to the surface in the spring to form a new plant (Countryman 1978). A single European frogbit plant can produce 100–150 turions in a season. Stolons, running from the center of the plant, produce juvenile plants, which tangle with other juvenile frogbit stolons and free-floating frogbit roots to create dense mats.

Tissue content

European frogbit is able to store metals and nutrients in its body. When 29 plant species in littoral vegetation of Lake Wadag of Poland were studied, the highest annual zinc accumulation rate was found in European frogbit (Grzybowski *et al.* 2000). Sviridenko *et al.* (1988) also reported that the species may be the best accumulator of nickel ions (70.6 μg/g dry weight; biological accumulation coefficient = 480). Engin *et al.* (2015) confirmed its ability to take up high amounts of iron, manganese and zinc as well (although this only seems to be the case when there are low concentrations of heavy metals within the environment; similarly for copper as in Shang *et al.* 2013). Therefore, this plant species can be used for the removal of heavy metals from waste water to protect water quality. This is one of its prominent environmental uses (Maleva *et al.* 2004; Polechońska and Samecka-Cymerman 2016; Reddy 1984). Results from Polechońska and Samecka-Cymerman (2016) showed frogbit was successful in collecting high amounts of cobalt, chromium, copper, iron, potassium, manganese, nickel, lead and zinc and was useful in treating polluted environments. Huang *et al.* (2006) compared frogbit with seven other aquatic plants such as *Gynura crepidioides*, *Sagittaria trifolia* and *Lemna minor* and found that frogbit had the highest accumulation capacity of heavy metals than other plants, which led to elevated concentrations of copper, lead, cadmium and zinc in its roots, stems and leaves. Engin *et al.* (2015) also argued that frogbit could be used as an indicator of environmental pollution due to its high accumulation of heavy metals and nutrients. Scholz and Anderson (2003) reported frogbit is often found in water with high concentrations of various elements including zinc (0.32 mg/L), barium (1.08 mg/L), iron (7.75 mg/L), magnesium (8.09 mg/L) and manganese (5.75 mg/L) whereas other aquatic plants such as yellow iris and water starwort only exist in water with detectable amounts of those elements. European frogbit has indicated eutrophication in lakes like Lake Mälaren and Lake Hjälmaren in Sweden (Andersson 2001). European frogbit can also be used to remove nitrogen and phosphorus to alleviate eutrophication. Shu (2013) used frogbit in the residential wastewater treatment and found it is efficient in reducing the total nitrogen content in water. However, the plant cannot tolerate very high nitrogen and phosphorus concentrations and it can die when nutrients reach the levels of 551 mg/L nitrogen and 50 mg/L phosphorus (Wan *et al.* 2006).

In addition to heavy metals, European frogbit can accumulate high concentration of some chemical compounds. Dormant turions of *H. morsus-ranae* contain a relatively high level of spermidine while the putrescine and ornithine levels are high in young leaves of *H. morsus-ranae* (Villanueva *et al.* 1985). Vernalized turions of *H. morsus-ranae* contain nearly three times more lysine, the cadaverine precursor, than do the dormant turions (Villanueva *et al.* 1985). Fu *et al.* (2005) conducted pharmacognosy of frogbit and found it contains amino acids, polysaccharides, organic acids and saponins.

Dispersal

Water, wind and animals are three major dispersal vectors of aquatic plants (Lacoul and Freedman 2006). Presence of European frogbit was negatively related to wind exposure, suggesting a higher invasion success in the sheltered areas of water bodies (B. Zhu, unpublished data). Humans are also responsible for an accelerated introduction of invasive plant species beyond their natural range (Ding *et al.* 2008; Lacoul and Freedman 2006). For example, a study found that human influence is the major factor to explain the presence of *H. morsus-ranae* in lakes of southern part of the Kenozersky National Park, Russia (Vekhoff 2000). European frogbit can be transported to new environments as either plantlets or turions hitchhiking on boats, waterfowl, and boat trailers, or carried by flowing or wind-driven currents. It can also be spread through improper disposal by water gardeners. Halvorsen (1989) hypothesized that dispersal by log rafting and relic occurrence along with dispersal by water birds were the possible ways of introduction of *H. morsus-ranae* to the Skien, Telemark, South Norway.

European frogbit invasion may be facilitated by cultural eutrophication process. Data gathered from Johnston and Brown (2013) from the Great Lakes showed how a rise in invasive plant species is connected to high phosphorous levels, which supports findings from Rejmánek *et al.* (2005) that invasions by plant species are more successful when there are more available nutrients. The spread of European frogbit may also depend upon its local abundance and its availability on land and water. Riis and Sand-Jensen (2002) studied a number of aquatic macrophytes including frogbit in Denmark streams and found a positive relationship between local abundance and range size, which means that plants spread to more areas when they are more abundant.

GROWTH ENVIRONMENT

Light and temperature

Light is necessary for turion germination in *H. morsus-ranae* (Richards and Blakemore 1975). However, day-length and intensity are unimportant for turion germination when compared with temperature (Richards and Blakemore 1975). Exclusion of light resulted in a 90% growth reduction of frogbit roots (Minshall 1959). Nonetheless in some studies, European frogbit showed tolerance in low light conditions. Among 12 ornamental species in Poland studied by Pindel and Wozniak (1998), only *H. morsus-ranae* and *Calla palustris* were able to develop and flower at relatively low light levels.

Temperature plays a critical role on the growth of European frogbit. Turion germination in *H. morsus-ranae* can be controlled by temperature and climatic conditions are indicated to be responsible for the limiting of this species to still and shallow waters in lowland England and Wales (Richards and Blakemore 1975). No germination occurs below 10°C and floating of turions does not occur until temperatures approach 20°C. In addition, 2 weeks of temperatures approaching 15°C are necessary for the germination of the majority of turions (Richards and Blakemore 1975). An increase in turbidity of the water from pollution might inhibit germination (Richards and Blakemore 1975). Zhu *et al.* (2008) also showed that water temperatures of 30°C hindered plant reproduction by limiting the number of the plant's bases (i.e. fewer plantlets). Although higher temperatures also seemed to have induced growth by increasing biomass, leaf size and root length, the number of roots, stems, stolons and leaves remained the same.

This species can grow from one hibernaculum to cover an area of 1 m in diameter in one season (Cook and Lüönd 1982). The development of hibernacula in *H. morsus-ranae* is affected by temperature and initiated by photoperiod between 15 and 25°C (Cook and Lüönd 1982). Below 10°C, no hibernacula are formed while above 25°C hibernacula develop almost immediately independent of photoperiod (Cook and Lüönd 1982; Vegis 1955).

Water and sediment

European frogbit does not grow well in sediment with clay particles and needs an organic substrate for its development

(Podbielkowski and Tomaszewicz 1974). It grows in water with low salinity (<0.30 mS/cm) and a salinity of 0.47 and 0.55 mS/cm limits the development of this species (Pindel and Wozniak 1998).

It seems that *H. morsus-ranae* tolerates high concentrations of hydrogen ions (Minshall and Scarth 1952). However, extreme acidities represented by a pH value of <4 had an adverse effect (from hydrogen-ion concentration) on frogbit root (Minshall and Scarth 1952). Such acidity inhibits the growth of the roots by decreasing cell division and cell elongation (Minshall and Scarth 1952). In the roots, cell division at pH 3.5 was 0.66 times the rate at pH 5.0 and this reduction accounted for 3/4 of the inhibition in the growth of the roots (Minshall and Scarth 1952).

Under some unusual conditions such as flooding, European frogbit can react by using the ethylene that builds up inside the plant from heightened water levels to increase the lengths of its shoots (Jackson 2008). Cookson and Osborne (1978), for example, found that applying an ethylene biosynthesis inhibitor to flooded *H. morsus-ranae* and *Ranunculus sceleratus* was successful in stopping shoot elongation.

Trophic state

Catling and Porebski (1995) noted that acidic and/or nutrient-poor waters may not be a suitable environment for *H. morsus-ranae*, but predicted that this species could become prevalent in the northern midwest and prairie regions of North America. Increasing eutrophication of the water under anthropogenic stress increased the expansion of the *H. morsus-ranae* in the taiga zone of Arkhangelsk region, Russia (Vekhov 1994).

It appears that European frogbit prefers mesotrophic lakes (Catling *et al.* 2003; Madsen 1998) although others suggest that this plant species favors water with a high conductivity and high nutrient content (Sager and Christian 2006). Cook and Lüönd (1982) indicate that European frogbit does not grow in oligotrophic waters. In contrast, Zhu *et al.* (2008) reported oligotrophic lakes, such as the Finger Lakes of New York, may be susceptible to the invasion of European frogbit in the near future. These results came from the experiments that involved simulating different trophic conditions using total phosphorus levels: 19 µg/L nearshore concentrations for oligotrophic conditions (open water concentration is much lower) and 36 µg/L nearshore concentration for mesotrophic conditions. Based on the data, oligotrophic conditions were not successful in hindering the growth or reproduction of the European frogbit, implying that various water bodies with a broader range of trophic levels are susceptible to its invasion (Zhu *et al.* 2008).

ECOLOGICAL IMPACTS

European frogbit can form dense mats of intertwined leaves and roots, thereby possibly preventing light penetration, shading out native vegetation, reducing native plant growth and diversity, decreasing water flow, limiting the amount of nutrients and dissolved gases available for native plants beneath the mats, reducing dissolved oxygen (DO) concentrations and affecting native fish, wildlife and ecosystem function (Catling *et al.* 1988; Zhu *et al.* 2008). For example, in some bays of Oneida Lake of New York, a local density of 512 plantlets/m^2 was observed and DO concentration was reported to be as low as 1.9 mg/L underneath the frogbit mat (Zhu *et al.* 2008).

European frogbit has profound negative impacts on native plants. They shade out native plants, particularly submerged ones. Especially during the summer, rapid stoloniferous growth leads to the formation of large masses of interlocking plants that decrease native submerged aquatic plant communities by reducing available light (Catling *et al.* 2003). They also occupy available spaces that would otherwise be occupied by native species (Bain and Mills 2004). Catling *et al.* (1988) demonstrated that the stabilized mats of *H. morsus-ranae* cause a decline in the submerged vascular aquatics below them. For example, European frogbit had an overall deleterious effect on the submerged *Utricularia vulgaris*, which increased by a factor of 13 without frogbit but declined to 1/8 in its presence (Catling *et al.* 1988). Zhu *et al.* (2014) also confirmed the negative effects of European frogbit on native plants while studying shading as a control method during one growth season from June to September. An overall decline in total submerged macrophyte biomass was found, along with a slight change in species richness when European frogbit was present. When 70% shade and 100% shade were applied, the community structures (measured by non-metric multidimensional scaling) were significantly different between the initial and final communities. This further reinforces the concept that light blocking (e.g. by European frogbit mat) could greatly impact submerged macrophyte communities.

Studies on wetland vegetation of the Great Lakes show that invasive species like European frogbit have made it difficult to distinguish individual plant communities from one another due to their broad distribution (Johnston *et al.* 2009). A study performed in a wetland reported introduced species including *H. morsus-ranae* would have a significant negative effect on the native plant community when they became dominant in the plant community (Houlahan and Findlay 2004). It concluded that discouragement of the spread of community dominants, regardless of geographical origin, is the key to conservation of inland wetland biodiversity (Houlahan and Findlay 2004). Furthermore, enclosure–exclosure experiments revealed that the introduction of *H. morsus-ranae* had no effect on community structure of plants (Thomas and Daldorph 1991). Similar findings were also discovered by studies done in the wetlands of the St. Lawrence River (Lavoie *et al.* 2003). Here researchers determined that the data did not strongly support any harm of exotic plants on native wetland species. There were also no connections found between native plant diversity and the invasive species being studied. On the contrary, a presence of invasive plants did not necessarily lead to those plants becoming dominant species in every single site, while other sites showed beneficial outcomes from the presence of introduced plants.

Studies which were conducted to investigate the impacts of European frogbit on fungi and bacteria have shown varied results. Ginns (1986) claimed to not have found any fungi on European frogbit. Since then, it has been shown that organic substances excreted by plants including frogbit can promote the growth of some fungi, which play a major role in the matter and energy cycles (Czeczuga *et al.* 2005). Czeczuga *et al.* (2005) examined the effect of nine species of aquatic plants on the occurrence of aquatic zoosporic fungal species in the water of three water bodies of different trophic status. They, however, reported fewer zoosporic fungal species in *H. morsus-ranae* (16) than *Sparganium ramosum* (23), *L. minor* (24) and *Nuphar luteum* (25). Catling *et al.* (2003) reported that no European frogbit had been found with bacteria growing on them. Similarly Anesio *et al.* (2000) investigated dissolved organic matter from frogbit with irradiation exposure and found that it hindered the bacteria's growth.

Studies also showed complex impacts of European frogbit on macroinvertebrates. Zhu *et al.* (2015) conducted a field experiment in Oneida Lake and found that European frogbit did not affect surface macroinvertebrates, but benthic invertebrate richness and abundance were negatively affected by the European frogbit cover. Interestingly, however, some bottom species actually benefited from the frogbit cover: while there were fewer worms (oligochaetes, leeches and flatworms) found, there were more chironomids when frogbit was present. A greater variety of benthic macroinvertebrate was also reported when frogbit was present. It is likely, as Zhu *et al.* (2015) hypothesized, that the effects of frogbit on macroinvertebrate communities may vary depending on the specific water body the frogbit is found in.

Recent studies have confirmed that European frogbit has negative impacts on native plants and has reduced DO concentration. However, other effects in the ecosystems are not easily predictable. Water chemistry (pH, nitrogen and phosphate), phytoplankton and zooplankton communities were actually not affected by European frogbit when compared to vegetation-free areas (B. Zhu, unpublished data). For fungi, bacteria and macroinvertebrates, studies have showed complex and sometimes conflicting results. Therefore, the negative ecological impacts of European frogbit were not as serious as previously thought. On the contrary, some macroinvertebrate communities may actually benefit from the presence of European frogbit. Therefore, removing this plant can improve oxygen level of the water but may also have unintended negative consequences (Catling *et al.* 1988; Zhu *et al.* 2015), which can make the control and eradication of European frogbit more challenging.

European frogbit can also have impacts on human activities. The dense mats of European frogbit may entangle around motorboat propellers, restrict water traffic, inhibit recreational activity and make fishing and swimming difficult (Hackett *et al.* 2014; Hummel and Kiviat 2004). It may also restrict hunting of diving ducks because they cannot get through dense mats to get food and may leave their previous habitat (Hackett *et al.* 2014). Due to its thick mats and poor water quality associated with its invasion, European frogbit may decrease waterfront property value and influence people's decisions about where to live. Horsch and Lewis (2008) studied >170 lakes in the northern Wisconsin and found an average 13% decrease in property value after the invasion of another plant, Eurasian milfoil. Leggett and Bockstael (2000) also reported a decrease in shoreline property value in response to water quality deterioration.

MANAGEMENT

Eliminating European frogbit on the invaded ecosystems has been a high management priority (Catling *et al.* 2003; Dunster 1990). Prevention is the most effective and cost-efficient form of management. To prevent the spread of aquatic plants, all plant material should be removed from boating and recreational equipment before moving to another water body. Education and monitoring are also essential for prevention and early detection. Where European frogbit is found in a water body, rapid response should be implemented. The following are several common control and eradication methods.

Harvesting and shading

A mechanical harvesting technique is the suggested method to control aquatic vegetation including *H. morsus-ranae*, preferably in spring or very early summer before frogbit has the chance to grow substantially (Catling *et al.* 2003). Hand-pulling can be effective when it is done frequently (Zhu *et al.* 2014). Shading with a shade cloth has been shown to effectively remove European frogbit (25 times less in areas treated with the 70% shade and nearly zero with the 100% shade) and had minor environmental effects (Zhu *et al.* 2015). While the shading technique using 70% or higher coverage was successful in removing European frogbit and in a lake mesocosm experiment even managed to increase dissolved oxygen levels, shading coverage of 70–100% was still shown to decrease submerged plant biomass, as well as cause a shift in community structure and number of species present (Zhu *et al.* 2014).

Chemicals

It has been observed that Gramoxone (Paraquat) at 5 L/ha is effective in the control of European frogbit in ditches (Newbold 1975). Furthermore, both Endothall (as Aquathol) (5 ppm) and Diquat (10 ppm) over the surface of static water in ditches gave good control of European frogbit (Holz 1963). Renard (1963) has also found that Diquat gave excellent control of emergent and submerged plants including frogbit in standing water. According to Renard (1963), treatment (Diquat) should be applied to well-developed plants between June and September and at a water temperature of 15–18°C. The diquat concentration should be <250 ppm because fish such as trout and roach can tolerate this level for relatively short periods.

High rates (12–14.4 kg/ha) of Amitrole-T (aminotriazole) has also been reported to control *H. morsus-ranae* (Hauteur and Canetto 1963). Rates of Dalapon of 10–40 pounds per acre had no effect on frogbit. Other herbicides such as Chlorthiamid, Terbutryn and Cyanatryn have also been indicated to control *H. morsus-ranae* (Catling *et al.* 2003). However, chemical treatment may eliminate other native aquatic plants including beneficial species and may have negative impacts on humans.

Biological control

H. morsus-ranae is a food resource for many animals including insects, rodents (e.g. beaver and mice), water birds, freshwater snails and fish (Bernatowicz and Wolny 1969; Catling and Dore 1982; Froemming 1954; Sviridenko *et al.* 1988; Vaananen and Nummi 2003). For example, insect larvae of *Hydrellia albifrons* Fallén have been observed feeding on *H. morsus-ranae* (Hering 1926). Ding *et al.* (2011) also think that aquatic weevil *Bagous chinensis* Zumpt could potentially develop on aquatic plants like European frogbit. Meanwhile Catling *et al.* (2003) reported observing snails consuming the leaves of European frogbit, but was unaware of the effects this had on the invasive species. Froemming (1954) observed that consumption of *H. morsus-ranae* stimulates egg production of the freshwater snails *Lymnaea stagnalis* and *Rumina decollata*. The embryonated eggs of the former species could easily be collected on the back of leafs of frogbit (Gudkov *et al.* 2006). However, field and laboratory experiments conducted by Zhu (2014) failed to show strong evidence that the presence of snails, specifically *Physa gyrina* Say, caused any significant damage to frogbit. Although there is a possibility that snails could still make useful biological control agents of frogbit, the species of snail selected should be made with consideration to the level of variety in the snail's diet (Zhu 2014). This may include species like *R. decollata* L., a snail species that has been observed eating European frogbit (Froemming 1954). Aside from snails, dabbling duck species (*Anas* spp.) have been documented to consume frogbit in the eutrophic wetlands of central Finland (Vaananen and Nummi 2003).

Fish, particularly grass carp, have been considered to be a possible biological control agent for European frogbit. A study from Stefanidis and Papastergiadou (2007) at Lake Pamvotis, where aquatic vegetation quantity (including European frogbit) was studied, showed a decrease in the number of aquatic vegetation species, which they believed was due to the introduction of certain fish species such as grass carp to the lake. European frogbit could be consumed at a rate of 740 g/kg body weight per day by 2-year-old grass carp (Magomaev 1973). A 3-year-old grass carp were shown to have a higher daily consumption and suggested to be better than 2-year-old for the weed control (Magomaev 1973). Nikolskii (1978) reported a daily consumption rate of 1254 g/kg by grass carp for European frogbit. This species, after *Potamogeton filiformis*, was the most preferred food for grass carp among 27 plant species in warm (30°C) pond waters (Nikolskii 1978). Cross (1969) suggested the introduction of grass carp as a possible method of controlling water weeds. Conversely, grass carp did not prefer *H. morsus-ranae* in another study (Sanders *et al.* 1991).

To date, using biological control to combat the spread of the invasive frogbit seems not as successful as we wished. Success with biological control could often be dependent upon the level of disturbance present within a habitat. According to Elger *et al.* (2004), the level of preference for consumption of a species like European frogbit to a generalist species like the pond snail *L. stagnalis* L. was found to be connected to the disturbance level of that plant's environment. However, their data also showed that with an influx of more available nutrients to the environment, the strength of this relationship diminished.

SUMMARY

The control of invasive species remains a pressing matter, despite legislations that have been implemented to limit their distribution. Maki and Galatowitsch (2004) were successful in obtaining frogbit and other aquatic plant species in Minnesota from US vendors despite the fact that it is 'illegal to possess, import, purchase, transport or introduce' such species in Minnesota. In fact, of their 40 requested purchases, 92% of those purchases were successfully sent. Such results led the researchers to conclude that the horticultural trade contributes greatly to the spread of aquatic plants. They also believed it may aid in the accidental invasion of other organisms as well, given the diversity of organisms that were found within their aquatic plant orders.

Many management practices have been studied and implemented to control the spread of European frogbit. While some techniques were shown to be successful, like hand-pulling, others may risk doing more harm than good. This includes shading and the use of chemicals, which in spite of producing positive outcomes, may adversely affect non-target species. Other methods like biological control continue to remain a promising possibility, although for methods like shading or chemicals, further study is encouraged to perfect their use.

The introduction of exotic species into foreign environments, whether accidental or intentional, continues to be a cause for concern, particularly for native wildlife (e.g. Antunes *et al.* 2018; Huang *et al.* 2018). Our review of recent studies on invasive European frogbit offers information on its biology and growth conditions as well as its effects on other species, which in some instances were shown to be less harmful than previously feared. Despite this, efforts to stop the spread of its invasion have remained a priority. Many types of control methods have been tested in the hopes of containing European frogbit such as hand-pulling or biological control with varying success from researchers. The present data from studies done with these control methods demonstrate that further research is required on these available control methods in order to limit the harm to native species and increase the chances of containing the frogbit invasion. Yet besides containing the invasion, education on how to prevent invasive species from further spread in the future is also highly recommended.

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