State of Michigan's

Status and Strategy for Starry Stonewort (*Nitellopsis obtusa* (Desv. in Loisel.) J. Groves) Management

Scope

The first written documentation of starry stonewort (*Nitellopsis obtusa* (Desv. in Loisel.) J. Groves, hereafter SSW) in North America was in 1978 in the St. Lawrence Seaway (Geis 1981); however, a vial containing SSW collected in 1974 in the St. Lawrence Seaway has been confirmed in the New York Botanical Garden's Characeae collection (Karol and Sleith, *In press*). Starry stonewort was first reported in Michigan in 1983 in the St. Clair - Detroit River System (Schloesser et al. 1986). Only recently has SSW been considered an aggressive nuisance in inland lakes (Eichler 2010). An earlier version of this document was a product of an Environmental Protection Agency - Great Lakes Restoration Initiative 205(j) grant between the Michigan Department of Environmental Quality and Central Michigan University (CMU) in 2014 (Hackett et al. 2014). It was significantly revised by CMU and partners on the Michigan Invasive Species Grant Program and reviewed by Michigan Departments of Environmental Quality and Natural Resources for the purposes of:

- Consolidating current science-based knowledge relative to the biology and ecology of SSW.
- Summarizing scientific literature and research efforts that inform management options for SSW in Michigan.
- Identifying future directions for research relative to successful SSW management in Michigan.

This document references peer-reviewed journals and publications. Any chemical, company, or organization that is mentioned was included for its involvement in peer-reviewed, published, publicly shared information, not to imply endorsement of the chemical, company, or organization.

Biology and Ecology

I. Identification

Starry stonewort is a macroalga that resembles true plants with a stem-like central axis (thallus) composed of a string of alternating, long internodal cells and short nodal cells that can grow from 30 to 120 cm (12 to 47 in) long and 2 mm or less (0.1 in) in diameter (Figure 1). Attached at the short nodal cells are a whorl of 5 to 8 longer cells called

Figure 1. Starry stonewort (*Nitellopsis obtusa*) has an axis of alternating long internodal cells (A) and short nodal cells (B). At the nodal cells there is a whorl of long cells forming branchlets (C). Photograph by Paul Skawinski, *Aquatic Plants of the Upper Midwest*

branchlets. Root-like rhizoids anchor SSW to the substrate. Asexual reproductive structures, called bulbils, are produced at any node of SSW, but are found concentrated on rhizoid nodes near the substrate. The star-like shape of SSW bulbils distinguishes it from other charophytes (Figure 2-3).

Figure 2. Starry stonewort is a macroalga that produces star-shaped bulbils. Photograph by Progressive AE

Starry stonewort is dioecious, meaning it has separate male and female individuals. At branchlet nodes, male individuals produce round, orange to red antheridia, from which sperm develop (Figure 4). Female individuals produce round, glossy brown to black, oogonia that develop oospores when fertilized (Rantzien 1963). The male antheridia have been mistaken for oogonia in the past (Pullman and Crawford 2010; per Sleith et al. 2015). Oogonia can be distinguished by the spiraling jacket cells that encircle the oogonia.

Figure 3. A star-shaped bulbil. Photograph by Paul Skawinski, University of Wisconsin Extension Lakes Program

Species that are often mistaken for SSW include other types of macroalgae, especially *Chara* spp. and *Nitella* spp. The genus *Nitellopsis* is represented by a single extant species, *Nitellopsis obtusa* (Soulié-Märsche et al. 2002). The distinctive star-shaped bulbils are the best way to distinguish SSW from other species. Starry stonewort has an asymmetrical branching pattern due to a long bract cell on the branchlet, unlike *Nitella* spp. where branchlets fork at the end with each fork being approximately the same length. The branchlets of fresh SSW retain their shape when held out of water while *Nitella* spp. are limp out of water. Starry stonewort can be distinguished from many non-algal, aquatic vascular plants by their clear filamentous rhizoids and star-shaped bulbils.

Figure 4. Male individuals of starry stonewort produce orange antheridia (i.e., male sex organs; circled in white). Photograph by Paul Skawinski, University of Wisconsin Extension Lakes Program

Paul Skawinski of the Wisconsin Citizen Lake Monitoring Network produced an Invasive Starry Stonewort Identification video that provides excellent tips for distinguishing SSW from other aquatic macrophytes:

https://youtu.be/te9iF5OTdtg. In the video, Skawinski describes SSW and compares it with similar and common aquatic species.

II. Detection

In the Great Lakes region, SSW can be found in large, dense mats or interspersed among native species (Figure 5), often near boat launches and marinas in relatively calm waters (Geis et al. 1981; Schloesser et al. 1986; Nichols et al. 1988; Midwood et al. 2016). Starry stonewort can be found at depths from 1 to 30 m (3 to 98 ft) but it is most often found between 1 and 4 m (3 to 13 ft) in the Great Lakes region (Olsen 1944; Schloesser et al. 1986; Nichols et al. 1988; Brainard and Schlutz 2016; Cahill et al. *in review*). This macroalga is rather cryptic when intermixed with other aquatic species and is often overlooked (Brainard and Schultz 2016). Once the SSW is pulled from the sediment, its star-shaped bulbils attached to clear, thin rhizoids clearly identify it. Bulbils can be found throughout the year (Midwood et al. 2016; Larkin et al. *in review*). Methods used to collect SSW for detection and identification have included Ponar grabs (e.g., Schloesser et al. 1986; Nichols et al. 1988), wading and dredging (e.g., Sleith et al. 2015), anchor drags (Kato et al. 2005), rake methods (e.g., Hilt et al. 2010; Cahill et al. *in review*), and SCUBA/snorkeling with

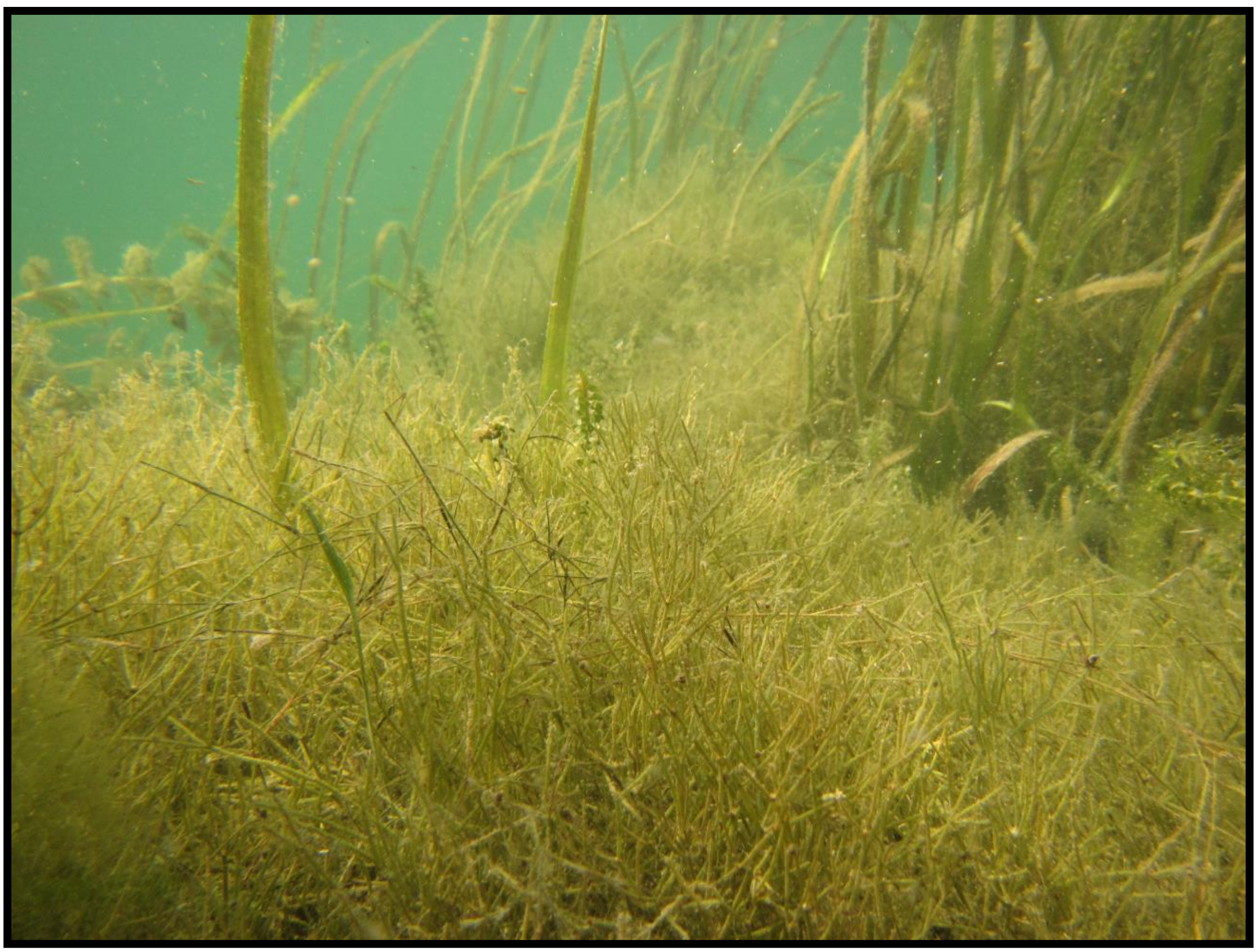

Figure 5. A mat of starry stonewort growing with eel grass (*Vallisneria americana* Michx.), common waterweed (*Elodea Canadenis* Michx.), and Eurasian watermilfoil (*Myriophyllum spicatum* L.). Photograph by Paul Skawinski

transects or quadrats (e.g., Geis et al. 1981; Kato et al. 2014; Brainard and Schulz 2016; Cahill et al. *in review*).

Aerial photographs have been used with botanists or local experts to distinguish emergent and floating aquatic vegetation (e.g., Husson et al. 2013). Submerged aquatic vegetation at water depths greater than 40 cm (16 in) cannot be distinguished using remote sensing technology at this time even when processed with object-based image analysis (Visser et al. 2013). Water absorbs the wavelengths commonly used to remotely sense vegetation (i.e., visible and near infrared). Remote sensing detection would also be limited in its ability to distinguish SSW in mixed stands of other aquatic vegetation.

Little research has been conducted to identify species-distinguishable markers for environmental DNA detection (eDNA) of macrophyte species and none have been published for macroalgae. Scriver et al. (2015) identified markers for ten aquatic invasive plant species distinct from other native species, and successfully identified species from laboratory-generated water samples. Field detection experiments on the aquatic invasive Brazilian waterweed (*Egeria densa* Planch.) in Japan had no false negative or false positive results when compared with field survey results (Fujiwara et al. 2016). Determining aquatic plant concentration with eDNA results has not yet been successful as they have shown fluctuation in laboratory experiments (Fujiwara et al. 2016; Matsuhashi et al. 2016). Starry stonewort may be a candidate for detection with eDNA. If it is possible to detect and differentiate SSW with eDNA, this could improve the true positive detection of SSW when it is growing undetected in mixed stands of aquatic vegetation or in an inaccessible portion of a waterbody. Environmental DNA detection procedures could also reduce the need for labor-intensive field surveys until after SSW was positively detected in an area.

Some work has been conducted to examine algal tissue to determine SSW genetic markers (Palma-Dow et al. 2016). This was developed to aid in SSW identification and specifically addressed positive identification of unidentified tissue samples.

III. Life History and Spread/Dispersal

In its native range, SSW is considered a summer annual in a subgroup of species surviving to the next year via winter bulbils in the substrate; in warmer winters, it may overwinter (Olsen 1944). In one Swedish lake, a detailed phenology was reported: SSW started to grow in April, reached peak biomass by the end of June, then died back during the late summer or fall months beginning near August (Hargeby 1990).

In its North American range, SSW has been documented to have two different phenologies. In the St. Lawrence River and the St. Clair - Detroit River System, much of its growth was noted in July to August, continued until September, and gradually declined during the winter. In mid-March, the SSW died back until July (Geis 1981; Nichols et al. 1988). In Michigan inland lakes, a phenology was observed similar to that observed by Hargeby (1990) in Sweden: vegetative dieback beginning in late August (Cahill et al. *in review*).

Starry stonewort is dioecious (i.e., separate male and female individuals). Starry stonewort populations in a single location can be composed of only a single sex in both its native and North American ranges (Willen 1960; Mann et al. 1999; Kato et al. 2014; Sleith et al. 2015). In its native range, SSW populations rarely produce sexual structures (Willén 1960). Krause et al. (1985) documented an increase in sexual development when SSW colonized shallow water. High water temperature and luminosity during the growing season have been suggested as triggers for sexual development (Willén 1960; Boissezon *In press*).

Starry stonewort can reproduce sexually via oospores in its native range, but it is rarely observed doing so (Willen 1960; Rantzien 1963; Bharathan 1983; Kato et al. 2014). Mature oospores are usually produced only under eutrophic conditions and have a mandatory dormant period of 1 to 3 months before germination (Bharathan 1987). Often the female organs are not fertilized.

In both its native and North American range, SSW reproduces primarily asexually via star-shaped bulbils (Figure 3) and fragmentation. Bulbils are present throughout the year (Midwood et al. 2016; Larkin et al. *in review*) and can sprout in three to five days under the right conditions (Bharathan 1987). Long term viability of bulbils is unknown.

In its North American range, only male or sterile individuals of SSW have been recorded (Mann et al. 1999; Sleith et al. 2015; Figure 4). Given the frequency of single sex populations found in its native range, it is likely that only male individuals have invaded North America at this time (Mann et al. 1999; Sleith et al. 2015). It is possible that female populations are present in North America, but no populations producing female sexual structures have been reported.

IV. Habitat

Native Range:

Starry stonewort is native to Europe and parts of Asia. European and Asian countries that have documented SSW include Belgium, Czech Republic, Denmark, Finland, France, Germany, Great Britain, Italy, Netherlands, Russia, Sweden, Switzerland, Malaysia, and Japan (e.g., Zaneveld 1940; Olsen 1944; Simons and Nat 1996; Kato et al. 2005; Caisová and Gąbka 2009). It is in decline in parts of Europe and endangered in the UK, which has produced detailed documents promoting the conservation of stonewort habitat throughout the country (Stewart 2004; Gołdyn 2009). Starry stonewort was declared extinct in the wild in Japan in 1994 before it was rediscovered in 2003 (Kato et al. 2005).

In its native range, SSW is uncommon, but in locations where it is present, it can be the dominant alga (Simons and Nat 1996; Królikowska 1997). Starry stonewort is found more often in oligo- or mesotrophic lakes, but has been observed in eutrophic conditions (Ozimek and Kowalczewski 1984; Hargeby 1990; Blindow 1992; Królikowska 1997; Bennett et al. 2001; Stewart 2004; Rey-Boissezon and Joye 2015; Schneider et al. 2015). It is often found in areas of low light intensity (e.g., deep waters of 4 to 8 m (13 to 26 ft)) but can be found as shallow as 1 m (3 ft) and deep as 30 m (98 ft; Olsen 1944). The lakes or locations are likely to be in areas protected from strong currents, have high calcium levels, high conductivity, neutral to basic pH, and have low to moderate forest cover in their catchment area (Zaneveld 1940; Olsen 1944; Simons and Nat 1996; Królikowska 1997; Soulié-Märsche et al. 2002; Boissezon 2014; Auderset Joye and Rey-Boissezon 2015; Rey-Boissezon and Joye 2015). Starry stonewort is tolerant of saline conditions and can survive up to a week in up to 17 practical salinity units (PSU; Simons and Nat 1996; Winter et al. 1999). Although it can tolerate fluctuations in salinity, it cannot survive and reproduce in water bodies with salinity consistently higher than 5 PSU and may need a minimal amount of calcium ions in the water to facilitate its tolerance (Okazaki et al. 1996; Winter et al. 1999).

In its native range, SSW can grow in large oligospecific mats of varying densities, usually in cool, lowland freshwater lakes with gently running water and high conductivity (Olsen 1944; Stewart 2004; Rey-Boissezon and Joye 2015). The mats can be dominated by SSW. Starry stonewort has been documented to co-occur with: *Chara aspera* Wildenow, *C. braunii* Gmelin, *C. coralline* Wildenow, *C. glabularis* Thuilier, *C. tomentosa* L., *Nitella hyaline* Agardh, *Ceratophyllum demersum* L., *Myriophyllum spicatum* L., *Potamogeton perfoliatus* L., and *Utricularia vulgaris* L. (Olsen 1944; Best 1987; Blindow 1992; Kato et al. 2005; Hilt et al. 2010).

Invasive Range:

In North America, SSW is present in the St. Lawrence Seaway, Lake Ontario, Lake Erie, Lake Huron, the St. Clair - Detroit River system, Michigan's Lower Peninsula, New York, Minnesota, Wisconsin, Pennsylvania, Vermont, and northern Indiana (Mills et al. 1993; Mills et al. 2007; Sleith et al. 2015; Midwood et al. 2016; Midwest Invasive Species Information Network (MISIN) 2017).

In the St. Lawrence River, Geis et al. (1981) documented SSW growing throughout the littoral zone with the greatest abundance at depths of 3 to 5 m (10 to 16 ft). In the St. Clair - Detroit River system, SSW was found at depths of 0.9 to 3.4 m (3 to 11 ft) in current velocities of 0 to 51.8 cm s^{-1} (Schloesser et al. 1986). Midwood et al. (2016) reported that density of docks, proximity to marinas, and low wave action (i.e., fetch) were good predictors of SSW presence in Prequi'ile Bay, Lake Ontario.

In North America, SSW has been observed to form a dense, vertically thick, oligospecific mats that completely covers the lake bottom or growing individually interspersed with other macrophytes (Figure 5). The SSW mat grows to non-uniform heights instead of mats of uniform heights like other macroalgae (Cahill et al. *in review*). Species that have been recorded to co-occur with SSW in North America include: *Chara* spp. L., *Nitella* spp. Agardh, *Ceratophyllum demersum* L., *Elodea canadensis* Michx., *Heteranthera dubia* (Jacq.) MacMill, *Lemna trisulca* L., *Nuphar variegata* Durand, *Myriophyllum sibiricum* Kom., *M. spicatum* L., *Najas flexilis* (Willd.) Rostk. & Schmidt, *Potamogeton friesii* Rupr., *P. gramineus* L., *P. nodosus* Poir., *P. richardsonii* (A. Benn.) Rydb., *P. zosteriformis* Fernald, *P. crispus* L., *Stuckenia pectinata* (L.) Böerner, *Tolypella intricata* (Trentepohl ex Roth) H. von Leonhardi, *Utricularia macrorhiza* Le Conte, and *Vallisneria americana* Michx. (Geis et al. 1981;

Schloesser et al. 1986; Nichols et al. 1988; Midwood et al. 2016; Brainard and Schulz 2016; Cahill et al. *in review*).

V. Effects from SSW

Brainard and Schultz (2016) published a peer-reviewed paper that examined environmental impacts of SSW. However, the majority of reports are based on anecdotal observations. No published studies could be found examining effects of SSW on food web dynamics, fish habitat, or recreational water use.

a. Negative Effects

In its native range, several groups of macroinvertebrates (i.e., *Asellus aquaticus* L., *Gammarus lacustris* G.O. Sars) have lower relative abundance in areas dominated by SSW compared to other native macroalgae (i.e., *Chara tomentosa*). The decrease in abundance between the two macroinvertebrate species was proposed to be due to the yearly die-back of SSW while *C. tomentosa* is green year-round (Hargeby 1990).

In its North American range, Brainard and Schulz (2016) conducted a quantitative study on the effects of SSW on the macrophyte community in four New York lakes. They found lower macrophyte species richness when SSW biomass increased at shallow (<1 m), intermediate (1 to 2 m), and deep sites in the littoral zone (>2 m).

b. Positive Effects

Laboratory testing in Austria established that SSW has an allelopathic effect on several species of cyanobacteria, inhibiting growth (i.e., *Anabaena cylindrica* Lemmermann*, A. torulosa* Lagerheim ex Bornet & Flahault*, Anabaenopsis elenkinii* V.V.Miller*, Aphanizomenon flexuosum* Komárek & Kovácik*, Cylindrospermum* spp. Kützing ex É.Bornet & C.Flahault, *Microcystis aeruginosa* (Kützing) Kützing*, M. flosaque* (Wittrock) Kirchner; Berger and Schageri 2004). These species of cyanobacteria and related species form toxic algal blooms in freshwater ecosystems that pose a risk to environmental and public health (Hudnell 2008). Field-testing has not been performed to confirm that allelopathy occurs in natural settings or if it has a significant impact on harmful cyanobacterial algal blooms. No allelopathic effects were recorded against eukaryotic cells (Berger and Schageri 2004).

In its native range, SSW can immobilize available phosphorus in its marl encrustation (Blindow 1987; Kufel and Kufel 2002; Siong and Asaeda 2006), potentially leading to less algal growth and higher water clarity in areas supporting large populations of SSW (Hilt et al. 2010).

Current Status and Distribution in Michigan

A vial containing a preserved specimen labeled *Nitellopsis* collected from the St. Lawrence Seaway in 1974 has been confirmed in the New York Botanical Garden's Characeae collection (Karol and Sleith, *In press*), but the first written SSW documentation in North America was in the

St. Lawrence Seaway in 1978 (Geis et al. 1981). In this publication, Geis et al. note that SSW was found dominating the macrophyte community of Goose Bay, north of Alexandria Bay, New York. Five years later Schloesser et al. (1986) collected SSW in the St. Clair - Detroit River System. It was suggested that SSW was introduced in ship ballast water (Geis et al. 1981). Studies in the 1980's and 1990's found SSW growing in oligocultures of eel grass (*Vallisneria americana* Michx.), Eurasian watermilfoil (*Myriophyllum spicatum* L.), Richardson's pondweed (*Potamogeton richardsonii* (A. Benn.) Rydb.), slender naiad (*Najas flexilis* (Willd.) Rostk. & Schmidt), and common waterweed (*Elodea canadensis* L.; Geis et al. 1981; Schloesser et al. 1986). It wasn't until the turn of the 21st century that SSW was perceived as a nuisance and was observed forming monospecific stands in inland lakes (Eichler 2010).

As of May 2017, SSW is present in over half the counties in the southern Lower Peninsula (Figure 6). Most reported sightings occur in southern Michigan in Oakland and Livingston Counties. Populations that were reported as dense were in Lotus, Maceday, and Angelus lakes and Mill Pond Park in Oakland County; Crooked, Gun, and Baker lakes in Barry County; Round Lake in Lenawee County; Black Creek in Macomb County; and Gull Lake in Kalamazoo County (MISIN) 2017). Most confirmed sightings did not contain information on density of populations.

A single unverified sighting was reported to MISIN in the Upper Peninsula in Millecoquins Lake in 2014. Experts dispatched to the site could not confirm the presence of SSW populations.

Management of SSW

I. Prevention

According to habitat modeling efforts of Escobar et al. (2016), much of Michigan is considered to have medium to high suitability for SSW. Preventing the establishment of SSW is preferable to post-establishment management. Starry stonewort is a "Prohibited Species" in Michigan under the Natural Resources and Environmental Protection act 451 of 1994. Under this act it may neither be sold nor grown in the state.

In North America, fragments and bulbils are likely transported by boats and to a lesser extent wildlife and currents from lake to lake (Sleith et al. 2015; Midwood et al. 2016). The following actions may prevent and limit the dispersal of SSW:

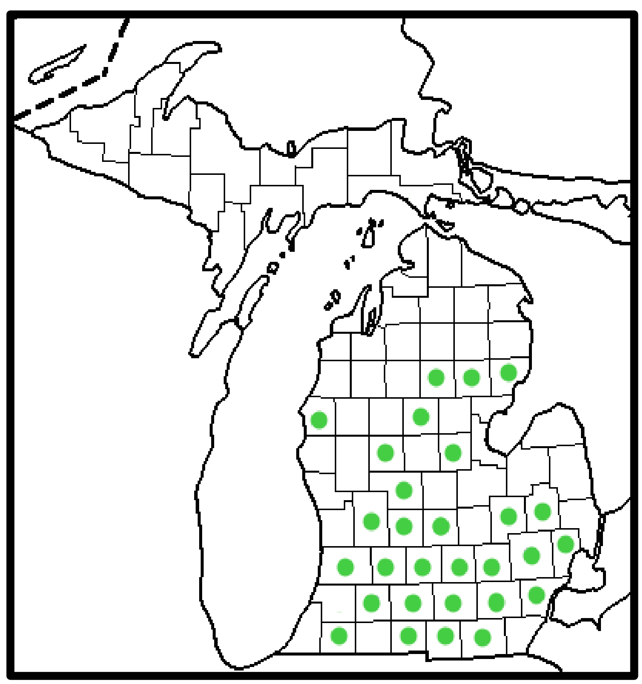

Figure 6. Green dots indicate reported presence of *Nitellopsis obtusa* on the Midwest Invasive Species Information Network (MISIN) submitted by trusted sources or verified. County map was developed by Michigan Flora Online (Reznicek et. al. 2011)

- Build a coalition of local, statewide, and Great Lakes regional partners to monitor for SSW and other aquatic invasive species
- Build a coalition of states that have classified SSW as a restricted or prohibited species
- Provide boat washing stations for high-traffic public lake accesses
- Develop and sustain a water recreation vehicles and trailers inspection program
- Identify water bodies of high risk of infestation using known distribution and dispersal knowledge

II. Management/Control

Although presented separately here, a management plan developed by integrating ecological knowledge, several management techniques, monitoring, and plan adaptation over time – called integrated pest management – is the most effective approach to controlling invasive species.

Few *in situ* experiments have been conducted to evaluate control methods for SSW (Cahill et al. *in review*) and many management recommendations are based on qualitative observations and are lacking untreated controls or pre- and post-treatment monitoring for effectiveness.

In Europe, some lakes where SSW populations are thought to have been extirpated have been rapidly recolonized (Hilt et al. 2010). In Japan, SSW was considered extinct as of 1993 but was rediscovered in a previously occupied lake in 2003 (Kato et al. 2005). The source of SSW recolonization years after populations were reduced or eliminated is believed to be from the germination of viable oospores in the seed bank (Kato et al. 2005; Hilt et al. 2010). This is an important consideration in the management of North American populations. Currently, only males have been detected, but in the event of female colonization and oospore production, management would become further complicated by the potential for populations to reestablish from oospores. Recolonization from bulbils in the seed bank is also plausible (Kato et al. 2005) and a concern in managing invasive populations of SSW. Bulbil viability is currently being investigated (K.G. Karol et al., NYBG, unpub. data).

a. Chemical

Application of copper-based algaecides (e.g., copper sulfate, chelated copper) is a chemical treatment used for SSW control (Cahill et al. *in review*; Larkin et al. *in review).* Cooper-based algaecides have been shown to reduce abundance and inhibit the growth of planktonic and filamentous alga species (e.g., *Raphidocelis subcapitata* (Korshikov) Nygaard, Komárek, J.Kristiansen and O.M.Skulberg, *Spirogyra communice* (Hassall) Kützing, *Microcystis aeruginosa* Kützing, and *Lyngbya magnifica* Gardner; Hallingse and Phlips 1996, Murray-Gulde et al. 2002, Bishop and Rodgers 2011, Tsai 2016). Several studies have been conducted to investigate the impact of copper-based algaecides on charophytes. In drainage canals of Argentina, copper sulfate controlled *Chara contraria* A.Braun ex Kützing and prevented oospore development over the growing season

(Fernández et al. 1987). Guha (1991) documented >90% reductions of *Chara* biomass in rice fields treated with copper sulfate. In laboratory trials, *Chara* germlings treated with chelated copper had reduced height and biomass compared to germlings in untreated controls (Kelly et al. 2012). It is important to note, none of these studies specifically address SSW.

Cahill et al. *in review* conducted a controlled study in Gun Lake, MI to investigate the effect of a series of two copper-based algaecide applications on SSW. In this study, chelated copper was applied during an early summer treatment and a mid-summer treatment. They found no significant differences between treatment and control plots for biomass or SSW mat height at two or four weeks after the early summer treatment or two or four weeks after the mid-summer treatment. Cahill et al. *in review* documented fluctuations in SSW mat height and biomass across the "growing season" and a late August die back, indicating a need for studies inclusive of untreated controls and scientific understanding of SSW ecology and phenology in the development and evaluation of management decisions. Alongside evaluating control of invasive macrophytes, it is imperative to adequately address the impacts of chemical application on native flora and fauna (Hanson and Stefan 1984; Mal et al. 2004) and the potential for legacy effects caused by the accumulation of copper in lakebed sediment (Han et al. 2001; Van Hullebusch et al. 2003; Marcussen et al. 2014).

Copper-based algaecides are sometimes combined with non-copper herbicides when treating SSW. Flumioxazin and endothall are the common non-copper herbicides used in these combination treatments (Larkin et al. *in review*). Endothall has had inconsistent effects on charophyte growth (Serns 1977; Steward 1980; Netherland and Turner 1995; Parsons et al. 2004) and empirical data demonstrating the effectiveness of flumioxazin on charophytes is lacking. Endothall and flumioxazin are broad-spectrum herbicides that can negatively impact the native macrophyte community (Skogerboe and Getsinger 2001; Skogerboe and Getsinger 2002; Glomski and Netherland 2013). Research is needed to evaluate the *in-situ* efficacy of different copper-based algaecides alone and in combination with non-copper herbicides for SSW control.

b. Physical or Mechanical Control

On small or recently introduced populations, manual removal and diver-assisted suction harvesting (DASH) may provide effective SSW control. These methods are time-and labor-intensive and often require repeated visits to maintain control (Bailey and Calhoun 2008; Kelting and Laxson 2010). In Little Muskego Lake and Silver Lake in Wisconsin projects evaluating the use of DASH for controlling newly detected populations of SSW are ongoing.

Mechanical harvesting has been carried out on SSW populations. Cahill et al. *in review* evaluated a late-season mechanical harvesting for SSW control. Biomass and SSW mat height were not significantly different between treatment and control plots following the harvest, indicating mechanical control in late summer coincided with SSW dieback (Cahill et al. *in review*). An evaluation of harvesting for SSW control is needed at various stages of the growing season. There is also a need to assess mechanical harvesting's potential to exacerbate SSW fragment and bulbil dispersal within waterbodies.

Drawdown of water level where it is practical may provide effective SSW control, but it has yet to be investigated. Starry stonewort is less tolerant to desiccation than other macroalgae in its native range (Boissezon 2014), but drawdown conditions may not be dry enough to prevent regrowth from surviving fragments or bulbils.

In Europe, biodegradable benthic barriers successfully controlled the invasive oxygen weed (*Lagarosiphon major* (Roxb.) (Ridley) Moss) while promoting the recovery of the native macrophyte community (Caffrey et al. 2010). An ongoing study in Gun Lake, MI is evaluating the efficacy of 14 oz. single-, double-, and triple-layer biodegradable benthic barriers for SSW control (Monfils et al., CMU, unpub. data).

c. Biological

There are no known species-specific biological controls for SSW. In its native range, eutrophication and competition from other plants limits the growth of SSW (Gołdyn 2009).

III. Indirect Management

No indirect management techniques have been investigated for the control of SSW at the time of this report.

Research Needs

I. Biology and Ecology

Currently, SSW has established populations in Michigan, New York, Indiana, Wisconsin, Minnesota, Vermont, Pennsylvania, and Ontario (Mills et al. 1993; Mills et al. 2007; Sleith et al. 2015; Midwood et al. 2016; MISIN 2017). The lack of a comprehensive survey effort and similarity to native flora (e.g., *Nitella*, *Chara*) make occurrences outside of SSW's distribution likely. Surveys in areas with recently discovered populations would be key to elucidating the full extent of SSW's distribution and potential pathways of introduction. Genetic research may also provide clues regarding primary dispersal pathways that would be useful to support preventative measures. It is imperative that SSW specimens from newly detected populations be vouchered to verify and document the occurrences.

Only male individuals of SSW have been documented to date in North America (Mann et al. 1999; Sleith et al. 2015). It is not clear if invasive SSW populations are composed of solely male individuals or if SSW females are not producing reproductive structures. If SSW females are present, SSW persistence in invaded waterbodies (i.e., oospores in seed bank) and dispersal potential (i.e., transport of oospores by wildlife) could be greater than previously thought. In both its native and invasive range, environmental conditions that trigger sexual development should be examined. The search for female SSW populations in North America could be aided by a better understanding of the locations and years female individuals are likely to become mature in its native range. Genetic work to detect the presence of female SSW would address this question and impact the management strategy that currently does not consider the impacts of sexual reproduction.

Examination of the phenology of SSW in North America is needed. Various late summer treatments are claiming to be effective controls of SSW, but SSW has been reported to die off naturally at that time (Cahill et al. *in review*). Continued monitoring over winter months will be instrumental in discerning the growth and phenology status relative to temperature and seasonality. Studies should cover multiple lakes, multiple years, and preferably multiple climatic zones.

It is known that SSW bulbils can germinate in three to five days (Bharathan 1987), but it is unknown how long bulbils remain viable. Understanding regrowth potential is important to those attempting to control populations of SSW. Similarly, understanding SSW bulbil and fragment tolerance to desiccation is crucial for predicting over-land dispersal and developing effective watercraft decontamination procedures.

To date, modeling of suitable SSW habitat has been coarse, examining a regional scale in North America. Escobar et al. (2016) predicted substantial portions of North America outside of the known distribution could be suitable for SSW. Understanding local characteristics (e.g., lake depth, water chemistry) that characterize SSW occurrence will improve predictions of SSW spread and help guide monitoring efforts. Monitoring efforts would benefit from a set of range-wide standardized procedures; a multi-state sample design and pre- and post-treatment monitoring effort would allow for large-scale studies that could inform best practices for SSW control.

Few studies have evaluated the ecological, economic, and recreational impacts of invasive SSW. In four inland lakes in New York, Brainard and Schultz (2016) documented a negative correlation between SSW abundance and macrophyte species richness. Further research is needed to describe the effects invasive SSW has on fish and invertebrate communities, as well as its economic and recreational impacts. Quantifying the effects of SSW will help prioritize invasive populations for management.

Given the difficulty in detecting early occurrences of SSW, eDNA may be a viable method to use for detection. Currently there are no techniques in place, like Fujiwara et al. (2016) implemented for Brazilian waterweed, to detect SSW from water samples taken in the field. Environmental DNA could improve the efficiency of early SSW detection. When unidentified macrophyte tissue is available for genetic analysis, Angela De Palma-Dow, Maggie Williams, and Jo Latimore from Michigan State University have identified markers that can distinguish SSW from other species. This is based on field collections of tissue and best assists in detecting false positives when there is uncertainty based on morphological character states.

Sleith et al. (2015) built on observations reported by citizens and verified by the Darrin Fresh Water Institute (Eichler 2010) to form an intensive detection strategy throughout New York State. Nearly 400 lakes were systematically sampled throughout the state. Nearshore examinations and dredges were used to determine presence or absence of SSW. Of the 390 lakes sampled, Sleith et al. (2015) found fifteen new records, confirmed twelve occurrence records, and was not able to confirm two occurrence records from the Eichler (2010) report. Building from what we have learned about detection and preferred habitat, a site-selection strategy could be refined by some of the habitat and environmental factors that have correlated with SSW presence in North America (e.g., public marina or boat launches, conductivity, distance to a confirmed infestation) to optimize resources and time. Once the distribution is known, high-risk water bodies can be monitored more intensely for invasion. Monitoring SSW and documenting variation in abundance from year to year would be instrumental in determining the best treatment type and time based on the site.

II. Management

Few quantitative studies have been conducted with untreated controls to document the efficacy of chemical treatment to control SSW (Cahill et al. *in review*). Cahill et al. evaluated the efficacy of a multi-stage management strategy, consisting of two chelated copper treatments and mechanical harvesting over the growing season, for SSW control. Neither the chelated copper treatments nor the mechanical harvesting had a significant impact when compared to untreated controls.

Studies that evaluate the short and long-term efficacy of other chemical treatments, as well as the impact these treatments have on native macrophyte, fish, and invertebrate communities are needed. Untreated control comparisons and quantitative pre- and post-treatment monitoring are required to properly measure the efficacy of any treatment.

Understanding how bulbil or oospore production is impacted by chemical treatment could lead to more effective management strategies. If bulbils or oospores are not affected by treatment or if production of these reproductive structures is enhanced following treatment, repeated applications will likely be required to maintain control.

No research has been published on potential biological controls, and little has been produced regarding mechanical and physical control techniques. Water level drawdown may limit SSW growth, but has yet to be examined quantitatively. Diver-assisted suction harvesting (DASH) was conducted on SSW populations in Little Muskego Lake and Silver Lake, WI and post-treatment monitoring is ongoing. Future research should investigate the potential for physical and mechanical control methods to proliferate SSW fragment and bulbil dispersal. Monitoring of biodegradable benthic barriers deployed in Gun Lake, MI for SSW control is in progress (Monfils et al., CMU, unpub. data). Understanding how bulbil and oospore viability are impacted by shading from benthic barriers could lead to more effective barrier implementation. If bulbils or oospores are capable of germination under benthic barriers a denser material may be required to prevent growth through the barrier.

Future Directions for Michigan and SSW Management

Starry stonewort is an aquatic macroalga native to Europe and Asia. In its native range, it is considered a desirable and/or threatened species confined to unpolluted waters. Male individuals have established invasive populations in many inland lakes in the Great Lakes Region.

Michigan is in a unique position to discover why SSW has become invasive in North America while being considered benign or beneficial elsewhere. Based on the rate we are detecting new populations, it is likely that SSW has already invaded other water bodies in Michigan and the U.S., but has not yet been positively identified. The submersed growth form and difficulty some have with identifying macroalgae makes documentation of the species difficult if not found in large exclusive mats, by which time restoration of habitat can be difficult.

Prevention – Prevention of new colony establishment is the most cost-effective approach to SSW management. Until the current distribution is known, prevention of spread will be difficult. Likely pathways of SSW dispersal are natural waterway currents and transportation of algal fragments or bulbils on boats and boating equipment. The development of a sustainable boat washing or inspection program could aid in containing the spread of this species.

Monitoring – Early detection would make eradication a more realistic option. Adding SSW to existing monitoring programs will assist in early detection and increase the potential of eradication. A cohesive monitoring and reporting system involving local municipalities, non-profit organizations, lake associations, recreation clubs and organizations, and waterfront property owners, would increase the number of known SSW locations and enable early detection and rapid response to new colonies. Connecting waterfront property owners and boaters with resources such as MISIN could improve early detection efforts. Working with herbaria for confirmation, documentation, and vouchering will provide verifiable long-term data that can be used to examine changes in macrophyte communities.

Starry stonewort monitoring would benefit from a direct and targeted monitoring strategy. To develop a targeted monitoring strategy, SSW occurrences and associated environmental variables could be modeled to identify suitable waterbodies for establishment. Suitable waterbodies that have a high-risk of SSW introduction could then be prioritized for monitoring, like Davidson et al. (2015) provided for a suite of invasive macrophytes in the Great Lakes Basin.

Networking data – Statewide monitoring methods would benefit from creating or participating in systems that centralize and provide open access to diversity data (e.g., MISIN, Weed Map – Cooperative Weed Management Area; MiCorps Data Exchange Network – Great Lakes Commission; Nonindigenous Aquatic Species Database – USGS; Biodiversity Information Serving Our Nation (BISON); and Global Biodiversity Information Facility (GBIF); Integrated Digitized Biocollections (iDigBio)). These databases house biological specimen or observation data including species location, verification, photographs, density, and even links to genetic data. Preliminary efforts within the state of Michigan have agencies contributing to regional databases (e.g., MISIN; Cooperative Weed Management Area; Nonindigenous Aquatic Species Database), but participation is not consistent and data standards are not established across programs. Currently state databases are not always networked within an agency, across the state, throughout the region or relative to national efforts.

Participation in a national or global information network will standardize data collecting practices, record comparable data using designated data standards across projects, ease data acquisition, avoid data redundancies, and promote projects with a larger scope of study than the original project for which the data sets were initially collected. Information networks that are continually linked to other resources and updated, can be used to develop effective and efficient monitoring and management plans. When information networks are not linked or periodically synchronized, a person collecting information must independently identify, locate, and consolidate data from separate and often difficult-to-access sources. The result is that information is missed and data collection becomes redundant and inefficient.

Networking with and contributing to state, regional, national, and international databases will advance research in areas that could improve the way aquatic invasive species are managed. Researchers can easily access the data and use it to model suitable habitat, model distribution, research population genetics across many spatial scales, predict new introductions, study changes due to climate change, or locate areas most beneficial for new projects or collections. The public could also use these data to know which species they may be exposed to when visiting specific water bodies.

Rapid response – The ability to rapidly respond to reports in new or high-value locations submitted by the public or through a regular monitoring strategy is essential to battling invasive species. Invasive species are easier to treat if the infestation is small. If the procedure to manage an infestation takes several years to achieve action, the infestation may have grown beyond realistic management. The Maine Department of Environmental Protection has developed a rapid response protocol that attempts to treat infestations of certain aquatic invasive species within 30 days of a newly detected aquatic invasion (MDEP 2006). The workflow begins at confirmation of report, and then delineation of infestation, containment, and primary evaluation. Next steps are treatment selection, plan refinement, and implementation. The infestation should be monitored and evaluated regularly for several seasons to evaluate the treatment and control any re-emerging growth. Although it is called a rapid response, it may not end rapidly.

Management – When managing SSW, it is important to delimit the extent of the infestation, contain already established populations, and protect high-value sites. An integrated plant management plan is needed to manage SSW.

Educating residents on the identification, restrictions, and ecological impacts of SSW could identify areas of infestation, assist in preventing new occurrences, and alert management prior to the establishment of dense mats.

Measuring effective control: Following the treatment of SSW, the effectiveness of treatment can be quantitatively assessed through documenting any year-to-year regrowth, reduction in SSW biomass, height, percent cover, or frequency as well as reduction in bulbil or oospore production. The goal of aquatic invasive species management strategies is to preserve or restore ecologically stable aquatic communities. Minimal chemical, biological, and physical controls should be required to maintain these communities. Any management plan should involve the integration of prevention and control methods that consider factors affecting the long-term ecological stability of an aquatic community.

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Table 1. Objectives, Strategic Actions, Leads, and Expected Outcomes of SSW Management.

Objective	Strategic Action	Who is leading effort in Michigan?	Expected Outcome
Guidance and Outreach for Starry Stonewort Management			
Increase public awareness of prevention methods	• Coordinate and collaborate with local and regional partners of water bodies with an infestation or high likelihood of infestation • Educate public of identification, early-detection, and prevention	• AIS Core Team • Lake Associations • Michigan Inland Lakes Partnerships • MSU Extension • *Nitellopsis* Working Group	• Increase public awareness of SSW • Increase the frequency and use of boat washing programs • Protect high-value sites • Contain established populations
Provide technical guidance to those interested in SSW management	• Framework to prioritize management of SSW populations • Educate stakeholders on available control methods	• *Nitellopsis* Working Group	• Increase management efforts
SSW Monitoring and Data Management			
Develop a mechanism for detecting, monitoring, and reporting AIS species	• Develop a system of identifying water bodies with high likelihood of infestation • Survey waterbodies with high likelihood of infestation	• AIS Core Team • MISIN • Michigan Water Corps	• Develop a more thorough and up-to-date statewide distribution of SSW • Evaluate dispersal pathways and vectors
Contribute regularly to regional, national, and global diversity information networks	• Consolidate Michigan biological and abiotic data • Standardize resources • Standardize data collection • Network existing data • Regularly synchronize data	• MISIN • Weed Map - CWMA • MiCorps • iDIgBio • NAS - USGS • BISON • GBIF	• Develop adaptive monitoring strategy that responds to up-to-date distribution • Promote AIS research of regional, national, and global extents • Prevent data redundancies
Educate public on identification and reporting of AIS in Michigan	• Target users of water bodies that are infested and have high-likelihood of infestation	• MISIN • Michigan Water Corps • Management agencies	• Increase public awareness of AIS • Identify water bodies that need professional confirmation of AIS
Research Needs for SSW Management			
Chemical: Develop treatments to increase long-term control or eradication success	• Develop guidelines for pre-, post-treatment, and control monitoring to determine treatment efficacy	• AIS Core Team • Integrated Invasive Aquatic Plant Management Team	• Effective treatment of infestation resulting in possible eradication of SSW
Biological: Establish biological control methods	• Identify any potential biological control species		• Increase long-term control success
Mechanical: Evaluate effectiveness of current mechanical controls	• Study the effectiveness of hand harvesting, diver-assisted suction harvesting, and mechanical harvesting for reducing/eliminating SSW	• Integrated Invasive Aquatic Plant Management Team	• Determine whether or not long term mechanical removal is a cost-effective management approach
Physical: Evaluate effectiveness of current physical controls	• Study the effectiveness of shading (e.g., benthic barriers) and lake level draw-down for reducing/eliminating SSW	• Integrated Invasive Aquatic Plant Management Team	• Determine whether or not physical controls are a cost-effective management approach

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