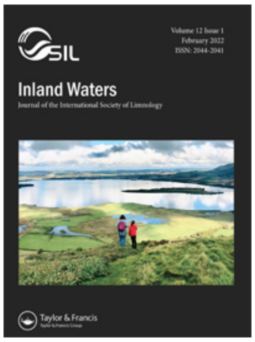

Taylor & Francis
Taylor & Francis Group

Inland Waters

ISSN: (Print) (Online) Journal homepage: https://www.tandfonline.com/loi/tinw20

Controlling water hyacinth (*Eichhornia crassipes* (Mart.) Solms): a proposed framework for preventative management

Linda May, Anne-Jo Dobel & Collins Ongore

To cite this article: Linda May, Anne-Jo Dobel & Collins Ongore (2022) Controlling water hyacinth (*Eichhornia crassipes* (Mart.) Solms): a proposed framework for preventative management, Inland Waters, 12:1, 163-172, DOI: 10.1080/20442041.2021.1965444

To link to this article: https://doi.org/10.1080/20442041.2021.1965444

Published online: 13 Dec 2021.

INLAND WATERS
2022, VOL. 12, NO. 1, 163–172
https://doi.org/10.1080/20442041.2021.1965444

RESEARCH BRIEF

Controlling water hyacinth (*Eichhornia crassipes* (Mart.) Solms): a proposed framework for preventative management

Linda May [a], Anne-Jo Dobel,[a] and Collins Ongore[b]

[a]UK Centre for Ecology & Hydrology, Edinburgh, Scotland, UK; [b]Kenya Marine and Fisheries Research Institute, Kisumu, Kenya

ABSTRACT

Over the last century, water hyacinth, *Eichhornia crassipes* (Mart.) Solms, has invaded freshwater systems in more than 50 countries, causing changes in biodiversity and widespread ecological damage. It also disrupts fisheries, navigation routes, power generation, and water supply. Although water hyacinth has invaded all tropical and subtropical countries and some parts of the Mediterranean basin, recent climate change models suggest that its distribution may soon expand into higher latitudes as temperatures rise within Europe, unless effective preventative management measures are put in place. In this paper, we explore the potential ecological and socioeconomic impacts of water hyacinth invasion using well-documented case studies from Lake Victoria. We also consider the relative effectiveness of biological, chemical, and mechanical control measures on established populations. We conclude that water hyacinth is almost impossible to remove once established, and that controlling its spread into new areas is probably the most cost-effective way of reducing its impact. We propose a framework for the preventative management of this weed by combining the use of environmental DNA as an early warning system with heightened biosecurity to prevent accidental introductions and the physical removal of invasive plants before they become established. We also recommend that nutrient concentrations be lowered in waterbodies to reduce their susceptibility to water hyacinth invasion and reduce its growth rate if introduced accidentally.

ARTICLE HISTORY
Received 12 September 2020
Accepted 25 July 2021

KEYWORDS
biocontrol; biosecurity; catchments; fisheries monitoring; nutrients

Introduction

Water hyacinth (WH), *Eichhornia crassipes* (Mart.) Solms, recently renamed *Pontederia crassipes* (Pellegrini et al. 2018), is a flowering aquatic plant native to South America, but once introduced into new locations it can grow quickly (Penfound and Earle 1948) to establish expansive colonies of tall, interwoven mats of floating material. In recent years, WH has spread rapidly around the world, clogging waterbodies and limiting their use for irrigation, fisheries, and recreation (Harley 1990). This invasion has caused environmental damage and socioeconomic impacts that include adverse effects on human health and education (Honlah et al. 2019). The spread of this plant has also undermined the development of sustainable blue economies in many areas of the world (Wilson et al. 2005, Ongore et al. 2018), and its range is predicted to expand, especially into more northern parts of Europe, as a result of climate change (Kriticos and Brunel 2016).

A good illustration of the ability of WH to colonise vast geographical areas is its well-documented invasion of waterbodies across Africa. Initially recorded in Egypt in the late 1880s, it then spread to South Africa (1910), Zimbabwe (1937), Mozambique (1946), Ethiopia (1956), Congo (1957), Rwanda and Tanzania (1950s), and Kenya (1982–1983) (Navarro and Kanyama-Phiri 2000). By 1990, WH had reached the Kenyan part of Lake Victoria.

Water hyacinth is believed to have entered Lake Victoria via the Kagera River (Muli 1996), which was delivering about 3.5 ha per day of WH mats into the lake by the early 2000s (Mailu 2001). This pathway illustrates the important role that freshwater connectivity plays in facilitating the spread of invasive species (Chapman et al. 2020).

The environmental and socioeconomic impacts of WH are well documented for the Lake Victoria basin, especially by Mailu (2001). The data collected provide important insights into the detrimental effects this plant can have on lakes and the local communities they support. This study uses this large body of evidence to compare the effectiveness of preventative management and restorative management in terms of controlling the spread of WH and reducing its impact.

CONTACT Linda May lmay@ceh.ac.uk UK Centre for Ecology & Hydrology, Bush Estate, Penicuik, Midlothian, Scotland, UK

Methods

Literature for this review, mainly compiled by searching Web of Science (WoS) using the search term "water hyacinth," returned 138 references, of which 46 were book chapters or parts of conference proceedings. An additional 11 unpublished reports were found during a search conducted using Google Scholar. The resultant 149 references were then imported into the reference manager Endnote X8 where they were filtered using the search terms "Lake Victoria" and "Winam Gulf" (61 references). The results formed the knowledge base for this review.

Impacts of water hyacinth

Environmental impacts

The adverse environmental impacts of WH invasion on a waterbody include a decline in water quality, higher water losses due to increased evapotranspiration rates (Lallana et al. 1987), greater siltation rates, and an overall decline in freshwater biodiversity. Areas of a waterbody with high WH infestations have significantly lower water quality than those without WH in terms of their colour, turbidity, and dissolved oxygen concentrations (Mailu 2001). The associated drop in oxygen levels can be sufficient to cause fish kills (Adan 2010), which can decimate fisheries and cause a more general loss of biodiversity. The loading of large quantities of plant organic matter to the lake bed from sunken WH biomass and other macrophyte detritus causes changes in water chemistry, an emerging environmental concern for the Winam Gulf of Lake Victoria in recent years. The Winam Gulf is especially prone to high levels of WH infestation because it is shallow, relatively sheltered, and receives high levels of nutrients from its catchment (Mitchell 1976).

Although fish diversity tends to be greater in areas infested by WH, the most abundant species tend to be those with a high tolerance for low oxygen levels, such as lungfish and catfish (Yongo et al. 2017). By contrast, the most abundant species in waters without WH is *Lates niloticus* (Nile perch), which is highly sensitive to low oxygen concentrations (Yongo et al. 2017). In addition to fish, WH invasion also threatens endemic free-floating macrophytes such as *Azolla pinnata* Decne ex Mett. while encouraging the growth of other aquatic plants such as *Ipomoea aquatica* Forsk., *Enydra fluctuans* Lour., and *Vossia cuspidata* (Roxb.) (hippo-grass) (Gichuki et al. 2012).

The environmental impacts of WH are not all negative, however. Some beneficial services it delivers include water purification, achieved by converting toxic ammonia into usable nitrates and by absorbing heavy metals and organic compounds (Ingole and Bhole 2003). However, even these benefits will have little long-lasting effect on water quality unless the plants are harvested because the stored nutrients and heavy metals will be recycled when the plant dies and decomposes. Ecosystem models used to evaluate the overall effects of WH on Lake Victoria have shown them to be dramatically negative (Nyamweya et al. 2016).

Socioeconomic impacts

The invasion of Lake Victoria by WH has been linked to a 45% decline in fish catches between the 1990s and early 2000s (Mailu 2001, Kateregga and Sterner 2009). The maximum sustainable yield of the lake based on fish stock estimates from 2014 is about US$800 million (note: all monetary estimates are in US$) per year (Odongkara et al. 2005), and the annual average fish catch of ~500 000 tonnes is valued at ~$600 million (LVFO 1999). With a potential sustainable export value of ~$288 million (Albright et al. 2004), the estimated direct economic impact of WH on the fishing industry is a loss in revenue of ~$130 million per year (Guerena et al. 2015). The value is much larger if the value of lost livelihoods within the fishing industry are taken into account.

Other socioeconomic impacts of WH infestations include the migration of fishing communities away from stricken areas, social conflict arising from competition to access areas of open water, fewer tourists, and blocked irrigation canals (Mailu 2001). Opande et al. (2004) investigated how WH had affected the people living around the Winam Gulf between 1995 and 1999 and found that WH had affected their livelihoods by blocking access to fishing and landing sites, disrupting the transport of fish around the lake, and damaging fishing gear. WH has also increased the cost of fishing operations and led to a decline in native fish species (Wawire and Ochiel 2004). Mats of WH have also blocked the breeding and feeding grounds of economically important fish such as tilapia and Nile perch (Twongo and Howard 1998).

The presence of WH mats could possibly provide a benefit by reducing the catchability of some overfished species and allowing fish stocks to recover, thus providing a long-term benefit to fisheries (Kateregga and Sterner 2009). However, catches of the most economically important fish (Nile perch and Nile Tilapia), usually found in open water, have been shown to be negatively correlated with the areal coverage of WH (Fig. 1; Opande et al. 2004, Ongore et al. 2018). This

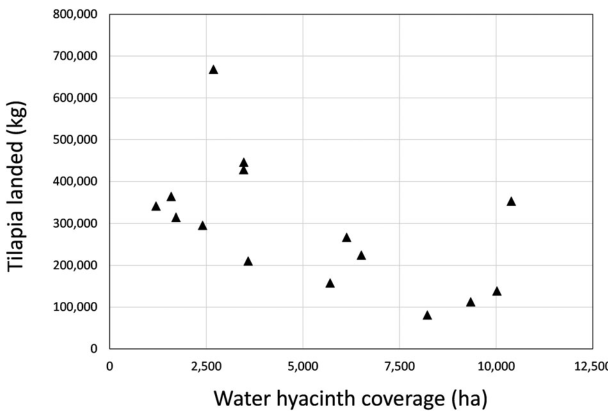

Figure 1. Plot of water hyacinth coverage and the amount of tilapia landed in the Kenyan part of Lake Victoria, February 2016 to April 2017.

relationship is problematic because these species have become commercially important in Kenya, which receives 80–90% of the financial yields from the Nile perch fishery and its associated processing factories (Greboval 1990). Although in general the reduction in fish catches has had serious economic consequences for the fishing communities that depend on them, some communities have been able to sustain their incomes by fishing for lungfish and catfish instead (Kateregga and Sterner 2009).

Water hyacinth infestations can also interrupt the provision of services such as irrigation, water treatment, and water supply, which has a marked impact on local communities (Opande et al. 2004). In 1999, for example, WH blocked a water intake near Kisumu, Kenya, reducing the local water supply by 25% and causing significant water shortages (Mailu 2001, Opande 2002). In addition, mats of WH have been deemed responsible for hydroelectric power production outages (Kateregga and Sterner 2007), with the additional costs of cleaning the intake screens of power plants to keep them operational estimated to be ~\$1 million per power plant per year (Mailu 2001).

Water hyacinth also affects the transport of cargo and people across affected waterbodies, especially in relation to smaller vessels (<700 tonnes) that cannot penetrate the WH mats without incurring unaffordable fuel costs (Mailu 2001). In some severely affected places, such as the Winam Gulf, frequent port closures due to obstruction by WH has caused a decline in sustainable development and significant loss of income to local communities. For example, between 1996 and 1997, the outgoing volume of cargo from Kisumu port alone fell from 84 000 to 34 000 tonnes per year due to a heavy WH infestation (Mailu 2001), amounting to losses of ~\$2.5 million per year per ship, not including associated losses in jobs and income (Guerena et al. 2015). By 2013, the estimated total economic impact of WH on Lake Victoria as a whole had reached ~\$350 million per year (Mkumbo and Marshall 2014).

Human health impacts

The social costs of WH are even greater than the economic costs; they include reduced access to safe water, increased levels of mortality, and higher risks of disease among the local population (Mailu 2001). Floating mats of WH provide more habitat for vectors of diseases such as schistosomiasis, malaria, and human lymphatic filariasis (Masifwa et al. 2001, Chandra et al. 2006). Even as early as 1962, WH infestations had been linked to an increase in mosquito-related problems because they hindered insecticide applications, interfered with predators, increased the amount of habitat available for vectors of disease, and impeded runoff and water circulation (Seabrook 1962). More recently, several studies have shown a positive correlation between increased WH cover and an increased abundance of the snails, *Biomphalaria sudanica* and *Bulinus africanus*, which are common hosts of schistosomiasis within Lake Victoria (Ofulla et al. 2010, Borokoni and Babalola 2012, Gichuki et al. 2012).

While some authors have suggested that WH masses of fibrous, free-floating roots, and semi-submerged leaves and stems enable it to decrease water currents and increase the amount of breeding habitat available to the malaria-causing *Anopheles* mosquito (Minakawa et al. 2008), others have noted that any causal link between WH masses and an increase in cases of malaria remains unproven (Mailu 2001). However, WH is known to harbour the causative agent for cholera, and annual changes in WH coverage within the Kenyan part of Lake Victoria are positively correlated with the number of cholera cases reported in the surrounding area (Feikin et al. 2010). This conclusion is further supported by the fact that *Vibrio cholerae*, which causes cholera, seems to concentrate on the floating tissues of WH under experimental conditions (Spira et al. 1981).

In addition to increasing the transmission of disease, other human health impacts can be directly linked to WH, including local increases in the number of attacks by wild animals, which have been attributed to the plant providing more cover for crocodiles, venomous snakes, and hippos (Ndimele and Jimoh 2011, Patel 2012). Additionally, malnutrition deaths have been reported because WH infestations have reduced fish catches (Navarro and Kanyama-Phiri 2000).

Although the socioeconomic costs of WH damage have been estimated to some degree, many of the social

Table 1. Overview of socioeconomic, social, and ecological costs (in US$) of water hyacinth infestation in Lake Victoria.

	Description	Economic costs	Reference
Socioeconomic	Total economic impact of WH on Lake Victoria	≈$350 million per year	Mkumbo and Marshall 2014
	Economic loss of cargo transport from Kisumu port between 1996 and 1997	≈$2.5 million per ship per year + associated losses of jobs and income	Guerena et al. 2015
	Direct economic impact of WH on the fishing industry	≈$130 million + value of lost livelihoods	Guerena et al. 2015
	Control costs in Louisiana, USA, between 1975 and 2013	$124 million	Wainger et al. 2018
	Hydroelectric power production outages and cleaning costs	Power outages + $1 million per site per year for cleaning	Kateregga and Sterner 2007, Mailu 2001
	Reduction in local water supply	NA	Opande 2002
	Migration of fishing communities, social conflict and competition, fewer tourists, blocked irrigation canals	NA	Mailu 2001
	Blocked access to fishing and landing sites, disruption to the transport of fish around the lake, damage to fishing gear	NA	Adan 2010
	Blocked breeding and feeding grounds for economically important fish such as tilapia and Nile perch	NA	Twongo and Howard 1998
Social	Access to safe water	NA	Mailu 2001
	Increased levels of mortality	NA	Mailu 2001
	increased risk of disease	NA	Mailu 2001
Ecological	Increased losses of water due to higher levels of evapotranspiration	NA	Lallana et al. 1987
	Lower water quality in terms of colour, turbidity, and dissolved oxygen concentrations	NA	Mailu 2001
	Reducing oxygen in water due to reduced air/water interface and reduction in submerged oxygenating plants	NA	Smith-Rogers unpubl.
	Fish kills (loss of biodiversity)	NA	Adan 2010
	Reduction in endemic free-floating macrophytes i.e., *Pistia stratiotes* and *Azolla pinnata*)	NA	Gichuki et al. 2012

and ecological costs are unknown (Table 1). Better cost estimates are critical to improve best practices for controlling and managing WH.

Potential control strategies

Considering the high environmental and socioeconomic impacts, understanding what controls WH distribution and abundance and determining how to limit its spread is essential. When floating freely, its distribution is often influenced by flow in rivers or by winds and currents in lakes. Its high reproductive rate in tropical regions, which is about double that in temperate regions, may possibly be related to higher pollinator abundances (Barrett 1980). While Albano et al. (2011) found that WH seed banks were common in Mediterranean climatic regions of South Africa and had high germination potential, its capacity for vegetative reproduction is most important in determining its spread and invasiveness. In addition, WH becomes abundant in areas that are high in nutrients and where water temperatures are between 20 and 30 °C (Center et al. 2002). In such locations, its growth rate can be up to 1.5% per day (Center and Spencer 1981).

Seasonal and climatic events, such as El Niño, can also affect the growth, spread, and distribution of WH, with heavy rains and rising water levels potentially causing the plant to become dislodged and redistributed (Albright et al. 2004, Williams et al. 2007). Although a potential link has been demonstrated between El Niño and WH abundance using earth observation data (Kiage and Obuoyo 2011), it is not yet known how climate change will alter El Niño (Paeth et al. 2008) and thereby its effect on WH.

Because distribution of WH is largely driven by flow, wind, and currents, hydrological connectivity is important in the spread of this species. Transport of propagules by water is probably the main mechanism for downstream dispersal, whereas transport of propagules by animals and humans is probably the most important vector for upstream dispersal and transfer from one geographical area to another (Jones et al. 2020). In terms of controlling dispersal, these mechanisms can be disrupted by creating artificial barriers, but their effectiveness is highly context dependent. Although in-stream barriers are unlikely to affect the transport of invasive aquatic plants by birds, they can restrict fish movements and disrupt fish-related transport processes (e.g., grazing by grass carp). Once introduced into freshwater systems, transport by humans can occur through accidental transfer (e.g., during the routine maintenance of swimming areas or the movement of boats and fishing gear). The risk of these accidental introductions can be controlled by introducing strict biosecurity measures, although these can be difficult to implement in some parts of the world (EPPO 2008). For example, some evidence indicates that WH has been spread through its use as an ornamental plant in garden ponds (Kriticos and Brunel 2016).

A modelling study by Wilson et al. (2000) indicated that 4 main environmental factors determine the abundance and distribution of WH: water temperature, level of disturbance, and water chemistry factors such as salinity and nutrient concentrations (Wilson et al. 2000). De Casabianca and Laugier (1995) studied the production of WH along a salinity gradient and found that productivity fell as salinity increased. However, in terms of chemical control of WH infestations, manipulating the salinity of a waterbody to suppress WH growth is not a practical option.

In contrast to the situation with salinity, reducing nutrient concentrations to deter WH is a much more feasible solution because growth is accelerated by nutrient enrichment (Carignan et al. 1994, Heard and Winterton 2000, Xie et al. 2004, Moran 2006, Coetzee et al. 2007, 2017, Coetzee and Hill 2012, Burke and Coetzee 2014, Canavan et al. 2014, Ismail et al. 2017). Wilson et al. (2005) showed that in eutrophic systems at 30 °C, WH biomass can increase from 0.01 to 10 kg m^{-2} in only 50 days, and even higher rates of growth were measured under similar conditions by Tucker and Debusk (1981). Understanding the interactions between WH and eutrophication could be key to its effective control (Gao et al. 2016), and the results of Haller and Sutton (1973) suggested that active growth of WH is likely inhibited if phosphorus (P) concentrations fall below 0.1 mg L^{-1}. In addition, many other studies have suggested that WH growth is correlated with concentrations of both nitrogen (N) and P in waterbodies (Bownes et al. 2013, Hill 2014), leading to suggestions that nutrient control measures could help slow its growth rate (Sinkala et al. 2002, Kiage and Obuoyo 2011, Chamier et al. 2012). In a related study, Kiage and Obuoyo (2011) suggested that mismanagement of catchments is the main reason for waterbodies such as Lake Victoria becoming more susceptible to WH invasion, and that improving land management practices to reduce nutrient runoff and soil erosion could be key to finding a solution to the WH problem. Also, P releases from the bed sediments of waterbodies will likely need to be controlled to help suppress WH growth.

Nutrient enrichment can also interfere with the effectiveness of other control measures. For example, Coetzee and Hill (2012) found that biological control agents (e.g., weevils) were far less able to control the spread of WH under high nutrient conditions. They concluded that WH growth is influenced more strongly by bottom-up regulation (nutrients) than by top-down regulation (herbivory), and that the first step in any WH control programme should be to reduce the nutrient status of the target waterbody.

Herbicide use has been proposed to control WH, but while this solution may seem relatively cheap and effective, concerns have been raised about the potentially damaging effects this form of control would have on the wider environment (Woomer 1997). For example, if applied over large areas, WH biomass will sink and undergo anaerobic decomposition, reducing oxygen levels, causing fish kills, and promoting internal release of P from sediments. Concern over the use of herbicides has also been raised in relation to potential effects on non-target plant species (Guerena et al. 2015), many of which are native and beneficial to the healthy functioning of freshwater ecosystems.

In the mid-1990s, the weevils *Neochetina eichhorniae* and *Neochetina bruchi* were introduced into Lake Victoria as biological agents to control the rapidly increasing populations of WH (Ochiel et al. 1998, Mwende and Njoka 2006). These weevils are the natural enemies of WH in their native South America, and the damage they cause (Fig. 2a–b) weakens the plants, making them more susceptible to disease and physical damage (Julien 2001). At first, weevils seemed to deplete the population of WH significantly, reducing its coverage from 20 000 ha in 1995 to <2000 ha by 1999, resulting

Figure 2. Weevil damage on water hyacinth in (a) Kisumu Bay, Lake Victoria and (b) individual leaves of the plant.

in this biocontrol programme being proclaimed one of the most successful in the world (Guerena et al. 2015). However, Williams et al. (2005) argued that weevils alone may not have been responsible for the rapid reduction in WH infestation that was observed, suggesting the decline was more likely to have been caused by an El Niño event that occurred at roughly the same time (1997 and 1998). However, Wilson et al. (2007) were able to demonstrate that WH levels had continued to decline after the El Niño event had ended, eventually collapsing the following year.

Although initially relatively successful as a biocontrol agent in Lake Victoria, the weevils have established a new ecological equilibrium with WH and are now less effective at controlling the weed than when first introduced (Wilson et al. 2007). However, when Hill and Gitonga (2015) surveyed the level of infestation at 23 sites on Lake Victoria and met with local people to discuss their experience of the weed, they concluded that the weevils were still having a considerable influence on the size of individual plants and of the overall population. The biggest weakness in the weevil biological control programme is the difficulty sustaining weevil populations once the WH mats have been decimated. A continuous breeding strategy to maintain the supply of biological control agents and support intensified, frequent introductions timed to coincide with the non-flowering phase of the invasion has been recommended by R. Omondi and others (unpubl.). Researchers are continuing to search for more types of biological control agents, including a number of insects identified as having high host specificity for WH and the appropriate thermal physiologies that suggest their potential for future release into Lake Victoria (May and Coetzee 2013, Hill and Coetzee 2017, Porter et al. 2019).

Weed cutting boats to control WH have been trialled in many places but with limited success. Although larger boats can clear harbour areas in the short term, large mats of WH are so mobile that keeping them clear is difficult. Typically, weed boats are effective only on small or spatially restricted infestations, such as those found on canals and waterways (Pieterse and Murphy 1990). While the shredding and removal of WH has been shown to allow algal reoxygenation of deeper waters on a local scale (Osumo 2001), the method must be tested for more extensive use. Nevertheless, weed removal does provide some opportunities for WH biomass to be recycled into goods that deliver financial and health benefits to local communities, including biogas (Guerena et al. 2015, Omondi et al. 2019), fuel pellets (Langeberg, pers. comm.), fertilisers, feed for fish and farm animals (Gopal 1987, Jarafi 2010), handicrafts (Opande et al. 2004), and other products. However, these small-scale activities do little to offset the much greater economic damage caused by WH (Makhanu 1997) and have never been shown to control its populations.

The case for preventative management

Water hyacinth has invaded freshwater systems in >50 countries across 5 continents (EPPO 2008), causing widespread environmental and socioeconomic damage to freshwater systems. While many attempts have been made to control or eradicate the weed, mostly focused on chemical, physical, and/or biological methods, they have had few lasting effects (Williams et al. 2005, Wilson et al. 2007), possibly because WH flowers produce large numbers of seeds that can remain viable in the bed sediments of infected waterbodies for up to 20 years (Matthews et al. 1977, Gopal 1987), even in areas where sexual reproduction is limited by the scarcity of suitable pollinators (Barrett 1980).

Once a WH infestation has taken hold, little can be done to eradicate it completely, and the control methods discussed earlier, apart from nutrient reduction, can cause further harm to an already stressed environment. Although WH is virtually impossible to remove completely, and suppression and containment make more sense economically, the cost of controlling WH infestations is still huge. In Louisiana, USA, for example, \$124 million was spent controlling WH between 1975 and 2013, although the socioeconomic benefits accrued over the same period were estimated to be ~\$4.2 billion (Wainger et al. 2018). Of the methods trialled, mechanical removal seems to be the preferred way of trying to reduce the nuisance caused by WH in specific problem areas, such as blocked access to landing beaches, ports, and harbours; it also avoids the use of potentially damaging herbicides. However, once the biomass has been removed, its safe disposal becomes a challenging environmental problem because the mechanical process can split WH biomass into thousands of small rhizomes, each of which can create a new plant. (Guerena et al. 2015). Harvesting also creates debris that falls to the bottom of the waterbody, where it decomposes causing deoxygenation problems (Guerena et al. 2015). Even if harvested in large quantities and used for societal benefit (i.e., to produce added value products), large infestations cannot be eliminated completely.

The only practical way to limit the damage this plant can inflict is preventing its introduction to susceptible systems. However, preventative management can only be effective if it involves a combination of measures that include (1) careful management of waterbody connectivity, (2) scrupulous biosecurity to prevent accidental transfer from one place to another, and (3) regular monitoring to enable action to be taken at an early stage if a potential invasion is detected (EPPO 2009). These measures need to include legislation that prohibits the planting or release of WH into uninfected areas (EPPO 2009) and a public education campaign (Kriticos and Brunel 2016) that encourages professional water managers and the public to help monitor the spread of WH into new areas (Nang'alelwa 2008) and to notify sightings to a central data repository via a mobile phone app (https://www.planttracker.org.uk/). Priority areas to survey would be natural waters, impoundments, and water tanks, with a particular focus on waterbodies rich in nutrients (EPPO 2009). In addition, recently developed environmental DNA (eDNA) techniques (Scriver et al. 2015) could provide early warning of potential invasion by WH before it becomes so abundant that its physical presence is visible to the naked eye or can be detected on satellite images.

Sepulveda et al. (2020) explored whether eDNA methods have developed sufficiently to enable their widespread incorporation into the management of aquatic invasive species. They found that, although eDNA techniques are sufficiently reliable for detecting presence and absence, invasive species managers often struggle with the level of uncertainty surrounding its use. Because the cost of taking action is high and currently no eDNA-based decision support tools are available operationally to help interpret the results, manuals on best practices, decision support trees for interpretation of results, and education and training of managers and stakeholders are required before widespread incorporation of eDNA into invasive species management. However, with many such outputs now being developed, eDNA approaches will likely be able to better support invasive species management in the future.

Although WH has invaded all tropical and subtropical countries and some parts of the Mediterranean basin (Kriticos and Brunel 2016), recent climate change models suggest its distribution may soon expand into higher latitudes as temperatures rise (Rodrıguez-Gallego et al. 2004, Hellmann et al. 2008, Rahel and Olden 2008, Kriticos and Brunel 2016). Europe has been identified as an area most at risk of invasion, with the currently limited distribution likely to expand to cover most of France, Greece, Italy, Portugal, and Spain unless effective preventative measures are put in place (Kriticos and Brunel 2016). Because eradication is almost impossible once established, an established preventative management strategy is critical to stop its spread to previously unimpacted areas. We therefore conclude that a framework of measures is needed whereby (1) connectivity is disrupted to prevent active or passive movement from one waterbody to another, (2) biosecurity is increased to reduce the risk of accidental transfer by humans, and (3) excessive nutrient inputs to waterbodies from external sources are reduced to make waterbodies less susceptible to invasion. At the same time, cost-effective monitoring solutions are needed to provide early warning of new invasions to enact effective control measures before the plant becomes established. As Leung et al. (2002) suggested, targeted surveillance and early intervention may provide the ounce of prevention that negates the need for the pound of cure. Operational use of eDNA could, potentially, provide the early warning of species invasions to deliver preventative management effectively.

Acknowledgements

We thank Dr Bryan Spears for his helpful suggestions and constructive comments.

Disclosure statement

No potential conflict of interest was reported by the author(s).

Funding

This work was supported by the UKRI Natural Environment Research Council under the SUNRISE programme [grant number: NE/R000131/1], an Official Development Assistance award.

ORCID

Linda May http://orcid.org/0000-0003-3385-9973

References

Adan MS. 2010. Impacts of water hyacinth on the fishing industry at the Winam Gulf, Kisumu County. Kenyatta University; [accessed 22 April 2021]. http://ir-library.ku.ac.ke/handle/123456789/12776

Albano PE, Coetzee JA, Téllez TR, Hill MP. 2011. A first report of water hyacinth (*Eichhornia crassipes*) soil seed banks in South Africa. S Afr J Bot. 77:795–800.

Albright TP, Moorhouse TG, McNabb J. 2004. The rise and fall of water hyacinth in Lake Victoria and the Kagera River basin, 1989–2001. J Aquat Plant Manag. 42:73–84.

Barrett SCH. 1980. Sexual reproduction in *Eichhornia crassipes* (water hyacinth). II. Seed production in natural populations. J Appl Ecol. 17:113–124.

Borokoni T, Babalola F. 2012. Management of invasive plant species in Nigeria through economic exploitation: lessons from other countries. Manag Biol Invasions. 3(1):45–55.

Bownes A, Hill MP, Byrne MJ. 2013. The role of nutrients in the responses of water hyacinth, *Eichhornia crassipes* (Pontederiaceae) to herbivory by a grasshopper *Cornops* *aquaticum* Bruner (Orthoptera: Acrididae). Biol Control. 67:555–562.

Burke A, Coetzee J. 2014. Effect of nutrient quality and leaf age of water hyacinth, *Eichhornia crassipes*, on the development of its co-evolved herbivore, *Eccritotarsus catarinensis* (Hemiptera: Miridae). Afr Entomol. 22(4):896–899.

Canavan K, Coetzee JA, Hill MP, Paterson ID. 2014. Effect of water trophic level on the impact of the water hyacinth moth *Niphograpta albiguttalis* on *Eichhornia crassipes*. Afr J Aquat Sci. 39(2):203–208.

Carignan R, Neiff JJ, Planas D. 1994. Limitation of water hyacinth by nitrogen in subtropical lakes of the Paraná floodplain (Argentina). Limnol Oceanogr. 39(2):439–443.

Center TD, Hill MP, Cordo H, Julien MH. 2002. Water hyacinth. In: Van Driesche R, Blossey B, Reardon R, Hoddle M, Lyon Suzanne, editors. Biological control of invasive plants in the Eastern United States. Washington (DC): USDA Forest Service Publication; p. 41–64.

Center TD, Spencer NR. 1981. The phenology and growth of water hyacinth (*Eichhornia crassipes* (Mart.) Solms) in a eutrophic north-central Florida lake. Aquat Bot. 10:1–32.

Chamier J, Schachtschneider K, le Maitre DC, Ashton PJ, van Wilgen BW. 2012. Impacts of invasive alien plants on water quality, with particular emphasis on South Africa. Water S Afri. 38:345–356.

Chandra G, Ghosh A, Biswas D, Chatterjee S. 2006. Host plant preference of *Mansonia* mosquitoes. J Aquat Plant Manag. 44:142–144.

Chapman DS, Gunn IDM, Pringle HEK, Siriwardena GM, Taylor P, Thackeray SJ, Willby NJ, Carvalho L. 2020. Invasion of freshwater ecosystems is promoted by network connectivity to hotspots of human activity. Global Ecol Biogeogr. 29(4):645–655.

Coetzee JA, Byrne MJ, Hill MP. 2007. Predicting the distribution of *Eccritotarsus catarinensis*, a natural enemy released on water hyacinth in South Africa. Entomol Exp Appl. 125(3):237–247.

Coetzee JA, Hill MP. 2012. The role of eutrophication in the biological control of water hyacinth, *Eichhornia crassipes*, in South Africa. Biol Control. 57(2):247–261.

Coetzee JA, Hill MP, Ruiz-Téllez T, Starfinger U, Brunel S. 2017. Monographs on invasive plants in Europe N° 2: *Eichhornia crassipes* (Mart.) Solms. Bot Lett. 164(4):303–326.

De Casabianca M, Laugier T. 1995. *Eichhornia crassipes* production on petroliferous wastewaters: effects of salinity. Bioresource Technol. 54:39–43.

[EPPO] European and Mediterranean Plant Protection Organization. 2008. Datasheet on *Eichhornia crassipes*. EPPO Bull. 38:441–449.

[EPPO] European and Mediterranean Plant Protection Organization. 2009. *Eichhornia crassipes*. EPPO PM 9/8(1).

Feikin D, Tabu C, Gichuki J. 2010. Does water hyacinth on East African lakes promote cholera outbreaks? Am J Trop Med. 83:370–373.

Gao L, Li B, Jin L. 2016. Can water hyacinth (*Eichhornia crassipes*) be controlled by reducing nitrogen and phosphorus pollution of water bodies? Appl Ecol Env Res. 14(3):77–91.

Gichuki J, Omondi R, Boera P, Okorut T, Matano AS, Jembe T, Ofulla A. 2012. Water hyacinth *Eichhornia crassipes* (Mart.) Solms-Laubach dynamics and succession in the Nyanza Gulf of Lake Victoria (East Africa): implications for water quality and biodiversity conservation. Sci World J. 2012:1–10.

Gopal B. 1987. Water Hyacinth. Amsterdam (Netherlands): Elsevier Science.

Greboval D. 1990. Appendix F. Socio-economic issues for planning in support of fisheries management. Report of the 5th session of the sub-committee for the development of the fisheries of Lake Victoria, United Republic of Tanzania, 12–14 September 1989; [accessed 12 September 2020]. http://www.fao.org/3/t0336e/T0336E04.htm

Guerena D, Neufeldt H, Berazneva J, Duby S. 2015. Water hyacinth control in Lake Victoria: transforming an ecological catastrophe into economic, social, and environmental benefits. Sustain Prod Consum. 3:59–69.

Haller WT, Sutton DL. 1973. Effect of pH and high phosphorus concentrations on growth of water hyacinth. Hyacinth Control J. 11:59–61.

Harley KLS. 1990. The role of biological control in the management of water hyacinth, *Eichhornia crassipes*. Biocontrol News Inf. 11(1):11–22.

Heard TA, Winterton SL. 2000. Interactions between nutrient status and weevil herbivory in the biological control of water hyacinth. J Appl Ecol. 37(1):117–127.

Hellmann JJ, Byers JE, Bierwagen BG, Dukes JS. 2008. Five potential consequences of climate change for invasive species. Conserv Biol. 22:534–543.

Hill JM. 2014. Investigations of growth metrics and delta N-15 values of water hyacinth (*Eichhornia crassipes*, (Mart.) Solms-Laub) in relation to biological control. Aquat Bot. 114:12–20.

Hill MP, Coetzee JA. 2017. The biological control of aquatic weeds in South Africa: current status and future challenges. Bothalia. 47(2):1–12.

Hill MP, Gitonga W. 2015. Post release evaluation of the biological control of water hyacinth on the Kenyan shores of Lake Victoria. Unpublished report. Grahamstown (South Africa): Rhodes University.

Honlah E, Segbefia AY, Appiah DO, Mensah M, Atakora PO. 2019. Effects of water hyacinth invasion on the health of the communities, and the education of children along River Tano and Abby-Tano Lagoon in Ghana. Cogent Soc Sci. 5:1.

Ingole NW, Bhole AG. 2003. Removal of heavy metals from aqueous solution by water hyacinth (*Eichhornia crassipes*). J Water Supply Res T. 52(2):119–128.

Ismail M, Compton SG, Brooks M. 2017. Interaction between temperature and water nutrient levels on the fitness of *Eccritotarsus catarinensis* (Hemiptera: Miridae), a biological control agent of water hyacinth. Biol Control. 106:83–88.

Jarafi N. 2010. Ecological and socio-economic utilization of water hyacinth (*Eichhornia crassipes* Mart Solms). J Appl Sci Environ Manag. 14(2):43–49.

Jones PE, Consuegra S, Börger L, Jones J, Garcia de Leaniz C. 2020. Impacts of artificial barriers on the connectivity and dispersal of vascular macrophytes in rivers: a critical review. Freshwater Biol. 65(6):1165–1180.

Julien MH. 2001. Biological control of water hyacinth with arthropods: a review to 2000. In: Julien MH, Hill MP, Center TD, Jianqing D, editors. Biological and integrated control of water hyacinth, *Eichhornia crassipes*. Proceedings of the 2nd meeting of the Global Working Group for the Biological and Integrated Control of Water

Hyacinth, Beijing, China, 9–12 October 2000. ACIAR Proceedings. 102:8–20.

Kateregga E, Sterner T. 2007. Indicators for an invasive species: water hyacinth in Lake Victoria. Ecol Indic. 7:362–370.

Kateregga E, Sterner T. 2009. Lake Victoria fish stocks and the effects of water hyacinth. J Environ Dev. 18:62–78.

Kiage LM, Obuoyo J. 2011. The potential link between El Nino and water hyacinth blooms in Winam Gulf of Lake Victoria, East Africa: evidence from satellite imagery. Water Res Manag. 25(14):3931–3945.

Kriticos DJ, Brunel S. 2016. Assessing and managing the current and future pest risk from water hyacinth, (*Eichhornia crassipes*), an invasive aquatic plant threatening the environment and water security. PLoS One. 11(8):e0120054.

Lallana VH, Sabattini RA, Lallana M. 1987. Evapotranspiration from *Eichhornia crassipes, Pistia stratiotes*, *Salvinia herzogii*, and *Azolla caroliniana* during summer in Argentina. J Aquat Plant Manag. 25:48–50.

[LVFO] Lake Victoria Fisheries Organisation Jinja Uganda. 1999. Strategic vision for Lake Victoria (1999–2015). Jinja (Uganda): LVFO.

Leung B, Lodge DM, Finoff D, Shogren JF, Lewis MA, Lamberti G. 2002. An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species. P Roy Soc Lond B Bio. 269:2407–2413.

Mailu AM. 2001. Preliminary assessment of the social, economic and environmental impacts of water hyacinth in the Lake Victoria Basin and the status of control. Biological and integrated control of water hyacinth: *Eichhornia crassipes*. Proceedings of the 2nd meeting of the Global Working Group for the Biological Integrated Control of Water Hyacinth. Beijing, China, 9–12 October 2000; p. 130–139.

Makhanu KS. 1997. Impact of water hyacinth on Lake Victoria. In: Pickford J, editor. Water and sanitation for all – partnerships and innovations. Proceedings of the 23rd WEDC international conference, 1–5 September 1997, Durban, South Africa; p. 165–166.

Masifwa WF, Twongo T, Denny P. 2001. The impact of water hyacinth, *Eichhornia crassipes* (Mart) Solms on the abundance and diversity of aquatic macroinvertebrates along the shores of northern Lake Victoria, Uganda. Hydrobiologia. 452:79–88.

Matthews LJ, Manson BE, Coffey BT. 1977. Longevity of water hyacinth (*Eichhornia crassipes* (Mart.) Solms) seed in New Zealand. Proceedings of the 6th Asian-Pacific Weed Scientific Society conference; p. 263–267.

May B, Coetzee J. 2013. Comparisons of the thermal physiology of water hyacinth biological control agents: predicting establishment and distribution pre- and post-release. Entomol Exp Appl. 147(3):241–250.

Minakawa N, Sonye G, Dida G, Futami K, Kaneko S. 2008. Recent reduction in the water level of Lake Victoria has created more habitats for *Anopheles funestus*. Malaria J. 7:119.

Mitchell DS. 1976. The growth and management of *Eichhornia crassipes* and *Salvinia* spp. in their native environment and in alien situations. In: Varshney CK, Rzoska J, editors. Aquatic weeds in Southeast Asia. The Hague (Netherlands): W. Junk; 396 p.

Mkumbo O, Marshall B. 2014. The Nile perch fishery of Lake Victoria: current status and management challenges. Fish Manag Ecol. 22:56–63.

Moran PJ. 2006. Water nutrients, plant nutrients, and indicators of biological control on waterhyacinth at Texas field sites. J Aquat Plant Manag. 44:109–114.

Muli JR. 1996. Environmental problems in Lake Victoria (East Africa): What the international community can do. Lakes Reserv. 2:47–53.

Mwende K, Njoka S. 2006. Biological control of water hyacinth: a case study of Lake Victoria Kenya. Lake Victoria Environmental Management Project Report; p. 1–5.

Nang'alelwa M. 2008. Public awareness activities on *Eichhornia crassipes* in the Victoria Falls world heritage site, Livingstone, Zambia. EPPO Bull. 38:464–469.

Navarro LA, Kanyama-Phiri GY. 2000. Water hyacinth in Africa and the Middle East: a survey of problems and solutions. Ottawa (ON): International Development Research Centre.

Ndimele P, Jimoh A. 2011. Water Hyacinth (*Eichhornia crassipes* [Mart.] Solms.) in phytoremediation of heavy metal polluted water of Ologe Lagoon, Lagos, Nigeria. Res J Environ Sci. 5(5):424–433.

Nyamweya C, Sturludottir E, Tomasson T, Fulton EA, Taabu-Munyaho A, Njiru M, Stefansson G. 2016. Exploring Lake Victoria ecosystem functioning using the Atlantis modeling framework. Environ Model Softw. 86:158–167.

Ochiel GRS, Mailu AM, Gitonga W, Njoka S. 1998. Biological control of water hyacinth on Lake Victoria, Kenya. Proceedings of the 1st IOBC Water Hyacinth Working Group, 16–19 November 1998, Harare, Zimbabwe; p. 115–118.

Odongkara KO, Abila R, Onyango PO. 2005. Distribution of economic benefits from the fisheries of Lake Victoria. Entebbe regional stakeholders' workshop, Entebbe, Uganda, 24–25 February 2005. Lake Victoria Fisheries Organization (LVFO); p. 124–131.

Ofulla AVO, Karanja D, Omondi R, Okurut T, Matano A, Jembe T, Abila R, Boera P, Gichuki J. 2010. Relative abundance of mosquitoes and snails associated with water hyacinth and hippo grass in the Nyanza Gulf of Lake Victoria. Lakes Reserv. 15:255–271.

Omondi EA, Ndiba PK, Njuru PG. 2019. Characterization of water hyacinth (*E. crassipes*) from Lake Victoria and ruminal slaughterhouse waste as co-substrates in biogas production. SN Appl Sci. 1(8):848.

Ongore CO, Aura CM, Ogari Z, Njiru JM, Nyamweya CS. 2018. Spatial-temporal dynamics of water hyacinth, *Eichhornia crassipes* (Mart.) and other macrophytes and their impact on fisheries in Lake Victoria, Kenya. J Geat Lakes Res. 44(6):1273–1280.

Opande GO. 2002. The distribution and origin of the water hyacinth (*Eichhornia crassipes*), its carpet characteristics, some of its diseases and studies of their causative agents in the Winam Gulf (L. Victoria). Kisumu (Kenya): Maseno University.

Opande GO, Onyango JC, Wagai SO. 2004. Lake Victoria: the water hyacinth (*Eichhornia crassipes* MART. SOLMS), its socio-economic effects, control measures and resurgence in the Winam Gulf. Limnologica. 34(1–2):105–109.

Osumo WM. 2001. Effects of water hyacinth on water quality of Winam Gulf, Lake Victoria. Unpublished report. Kenya Marine Fisheries Research Institute and University of Akureyri and Marine Research Institute, Iceland.

Paeth H, Scholten A, Friederichs P, Hense A. 2008. Uncertainties in climate change prediction: El Niño-Southern Oscillation and monsoons. Global Planet Change. 60(3–4):265–288.

Patel S. 2012. Threats, management and envisaged utilizations of aquatic weed *Eichhornia crassipes*: an overview. Rev Environ Sci Biotech. 11:249–259.

Pellegrini MOO, Horn CN, Almeida RF. 2018. Total evidence phylogeny of Pontederiaceae (Commelinales) sheds light on the necessity of its recircumscription and synopsis of *Pontederia* L. PhytoKeys. 108:25–83.

Penfound WT, Earle TT. 1948. The biology of the water hyacinth. Ecol Monogr. 18(4):447–472.

Pieterse AH, Murphy KJ. 1990. Aquatic weed, the ecology and management of nuisance aquatic vegetation. Oxford (UK): Oxford University Press.

Porter JD, Owen CA, Compton SG, Coetzee JA. 2019. Testing the thermal limits of *Eccritotarsus catarinensis*: a case of thermal plasticity. Biocontrol Sci Technol. 29(6):565–577.

Rahel FJ, Olden JD. 2008. Assessing the effects of climate change on aquatic invasive species. Conserv Biol. 22:521–533.

Rodrıguez-Gallego LR, Mazzeo N, Gorga J, Meerhoff M, Clemente J, Kruk C, Scasso F, Lacerot G, Garcıa J, Quintans F. 2004. The effects of an artificial wetland dominated by free-floating plants on the restoration of a subtropical, hypertrophic lake. Lakes Res. 9:203–215.

Scriver M, Marinich A, Wilson C, Freeland J. 2015. Development of species-specific environmental DNA (eDNA) markers for invasive aquatic plants. Aquat Bot. 122:27–31.

Seabrook EL. 1962. The correlation of mosquito breeding to hyacinth plants. J Aquat Plant Manag. 1:18–19.

Sepulveda AJ, Nelson NM, Jerde CL, Luikart G. 2020. Are environmental DNA methods ready for aquatic invasive species management? Trends Ecol Evol. 35(8):668–678.

Sinkala T, Mwase ET, Mwala M. 2002. Control of aquatic weeds through pollutant reduction and weed utilization: a weed management approach in the lower Kafue River of Zambia. Phys Chem Earth. 27:983–991.

Spira WM, Huq A, Ahmed QS, Saeed YA. 1981. Uptake of *Vibrio cholerae* Eltor from contaminated water by water hyacinth, *Eichhornia crassipes*. Appl Environ Microbiol. 42:550–553.

Tucker CS, Debusk T. 1981. Productivity and nutritive value of *Pistia stratiotes* and *Eichhornia crassipes*. J Aquat Plant Manag. 19:61–63.

Twongo T, Howard G. 1998. Ways with weeds. New Sci. 159:57.

Wainger LA, Harms NE, Magen C, Liang D, Nesslage GM, McMurray AM, Cofrancesco AF. 2018. Evidence-based economic analysis demonstrates that ecosystem service benefits of water hyacinth management greatly exceed research and control costs. PeerJ. 6:e4824.

Wawire NW, Ochiel GR. 2004. Review of the impact of water hyacinth on Lake Victoria: the case of Winam Gulf, Kenya. Lake Victoria Environmental Management Project (LVEMP); [accessed 24 April 2021]. http://repository.eac.int:8080/handle/11671/798

Williams AE, Duthie HC, Hecky RE. 2005. Water hyacinth in Lake Victoria: Why did it vanish so quickly and will it return? Aquat Bot. 81(4):300–314.

Williams AE, Hecky RE, Duthie HC. 2007. Water hyacinth decline across Lake Victoria—was it caused by climatic perturbation or biological control? A reply. Aquat Bot. 87:94–96.

Wilson JR, Ajuonu O, Center TD, Hill MP, Julien MH, Katagira FF, Neuenschwander P, Njoka SW, Ogwang J, Reeder RH, et al. 2007. The decline of water hyacinth on Lake Victoria was due to biological control by *Neochetina* spp. Aquat Bot. 87:90–93.

Wilson JR, Holst N, Rees M. 2005. Determinants and patterns of population growth in water hyacinth. Aquat Bot. 81(1):51–67.

Wilson JR, Rees M, Holst N, Thomas MB, Hill G. 2000. Water hyacinth population dynamics. In: Julien MH, Hill MP, Center TD, Jianqing D, editors. ACIAR Proceedings 102; p. 96–104.

Woomer PL. 1997. Managing water hyacinth invasion through integrated control and utilization: perspectives for Lake Victoria. Afr Crop Sci J. 5:309–324.

Xie Y, Wen M, Yu D, Li Y. 2004. Growth and resource allocation of water hyacinth as affected by gradually increasing nutrient concentrations. Aquat Bot. 79(3):257–266.

Yongo E, Outa N, Omari Ngodhe S. 2017. Effects of water hyacinth (*Eichhornia crassipes* Solm) infestation on water quality, fish species diversity and abundance in the Nyanza Gulf of Lake Victoria, Kenya. Int J Fish Aquat Res. 2:08–10.