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Review

Management and control methods of invasive alien freshwater aquatic plants: A review

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ABSTRACT

Introduced invasive alien aquatic plants (IAAPs) threaten ecosystems due to their excessive growth and have both ecological and economic impacts. To minimize these impacts, effective management of IAAPs is required according to national or international laws and regulations (e.g. the new EU regulation 1143/2014). Prevention of the introduction of IAAPs is considered the most cost effective management option. If/when prevention fails, early detection and rapid response increases the likelihood of eradication of the IAAPs and can minimize on-going management costs. For effective weed control (eradication and/or reduction), a variety of management techniques may be used. The goal or outcome of management interventions may vary depending on the site (i.e. a single waterbody, or a region with multiple waterbodies) and the feasibility of achieving the goal with the tools or methods available. Broadly defined management goals fall into three different categories of, containment, reduction or nuisance control and eradication. Management of IAAP utilises a range of control methods, either alone or in combination, to achieve a successful outcome. Here we review the biological, chemical and mechanical control methods for IAAPs, with a focus on the temperate and subtropical regions of the world and provide a management diagram illustrating the relationships between the state of the ecosystem, the management goals, outcomes and tools.

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1. Introduction

Aquatic plants play an important role in the functioning of aquatic ecosystems (Jeppesen and Sondergaard, 1998). However, introduced invasive alien aquatic plants (IAAPs) may threaten ecosystems due to their excessive growth and have both ecological and economic impacts (Getsinger et al., 2014; Brundu, 2015). Ecologically, they change the macrophyte community composition (Santos et al., 2011; Hussner, 2014), modify macroinvertebrate species richness and abundance (Stiers et al., 2011), deplete oxygen (Shillinglaw, 1981) and alter the food web structure (Villamagna and Murphy, 2010). Moreover, dense macrophyte stands can increase the flood risk by impeding river flow (Holm et al., 1969; Wilcock et al., 1999; Thouvenot et al., 2013), hinder shipping and navigation (Holm et al., 1969) and impair recreational water sports activities, which decreases the value of lakefront property (Halstead et al., 2003). Furthermore, IAAPs clog hydropower dams, which reduces hydropower generation (Clayton and Champion, 2006), and they reduce the water flow and availability in irrigation and drainage systems (Holm et al., 1969).

Due to their potential for large scale negative impacts, IAAP risk assessments have been carried out in numerous countries and several species are now subject to management and control (Table 1). The European parliament adopted a regulation on invasive species (Reg. EU no. 1143/2014), which came into effect on the 1st of January 2015 and regulates the introduction and management of IAAPs in Europe, with the aim of rapid eradication of populations already present and prevention of future invasions (Genovesi et al., 2015). In addition to this EU regulation, the management of IAAPs in general is receiving increased attention, but as yet a comprehensive review of management methods for IAAPs is lacking. Recent guides for the management of IAAPs are focused in specific countries (e.g. for New Zealand: De Winton et al., 2013; for the US: Gettys et al., 2014). In this paper, we examine and review the spectra of control methods for IAAPs to guide management decision making, reviewing management experiences from different parts of the world, with a focus on freshwater bodies in the temperate and subtropical regions.

1.1. The crucial role of aquatic vegetation for ecosystem functioning

Whereas IAAPs can be perceived as a threat, generally freshwater macrophytes are appreciated for their major structuring role in shallow aquatic ecosystems (Scheffer et al., 2001; Burks et al., 2006). Native, multispecies macrophyte beds provide habitats that support biodiversity within aquatic systems and macrophytes perform important ecosystem functions, including nutrient retention, enhancing water clarity by trapping sediment (Cotton et al., 2006; Horppila and Nurminen, 2003; Wang et al., 2015), inhibiting algal blooms (Gross, 2003),providing food for herbivores (Carpenter and Lodge, 1986; Van Donk and Van de Bund, 2002; Bakker et al., 2010; Declerck et al., 2011; Massicotte et al., 2015) and habitat for fish (Jeppesen and Sondergaard, 1998). In principle some of these benefits of macrophytes could also be provided by invasive macrophytes (Grutters et al., 2015a,b). Therefore, depending on the ecosystem

Table 1
A selection of the most important aquatic weeds in Europe, New Zealand and the US and their native and introduced range.

Species name	Native range	Invasive range	References
Alternanthera philoxeroides	South America	North America, Europe, Oceania	Champion and Clayton, 2001; Gordon et al., 2012
Azolla filiculoides	North-, Central-, South America	Africa, Asia, Europe, Oceania	Hill, 1999; Roy et al., 2004
Cabomba caroliniana	North America	Asia, Europe, Oceania	Mackey and Swarbrick, 1997; Hussner, 2012; Weber et al., 2008
Crassula helmsii	Australia	Europe, North America	EPPO, 2007; Diaz, 2011
Ceratophyllum demersum	Cosmopolitan	Oceania (New Zealand)	Wells et al., 1997
Eichhornia crassipes	South America	Africa, Asia, Europe, North America, Oceania	EPPO, 2009; Pan et al. 2011
Egeria densa	South America	Asia, Central- and North America, Europe, Oceania	Champion and Clayton, 2001; , Gordon et al., 2012; Hussner, 2012; Haramoto and Ikusima, 1988
Elodea canadensis	North America	Asia, Europe, Oceania, Alaska	Holm et al., 1969; Hussner, 2012; Alaska DNR, 2016
Elodea nuttallii	North America	Asia, Europe, Alaska	Cook and Urmi-König, 1985; Hussner, 2012; Kadono, 2004; Alaska DNR, 2016
Hydrilla verticillata	Asia	Africa, Europa, North America, Oceania	Cook and Lüond, 1982; Langeland, 1996; Champion and Clayton, 2001
Hydrocotyle ranunculoides	North America	Africa, Asia, Europe, Oceania	EPPO, 2006; Ruiz-Avila and Klemm, 1996
Hygrophila polysperma	Asia	North America, Europe, Oceania	Cuda and Sutton, 2000; Champion and Clayton, 2001; Hussner, 2012
Lagarosiphon major	South Africa	Europe, Oceania	Champion and Clayton, 2001
Ludwigia grandiflora	South America	Africa, Europe, North America	EPPO, 2011
Ludwigia peploides	North-, Central-, South America	Africa, Asia, Europe, Oceania	EPPO, 2011; Champion and Clayton, 2001
Myriophyllum aquaticum	South America	Africa, Asia, Europe, North America, Oceania	Hussner and Champion, 2011
Myriophyllum heterophyllum	North America	Europe	Hussner, 2012
Myriophyllum spicatum	Asia, Europe	North America	Aiken et al., 1979
Pistia stratiotes	South America	Africa, Asia, Europe, North America, Oceania	Neuenschwander et al., 2009
Salvinia molesta	South America	Africa, Asia, Europe, North America, Oceania	Luque et al. 2014; Koutika and Rainey, 2015
Trapa natans	Africa, Asia, Europe	North America, Oceania	Champion and Clayton, 2001; Hummel and Kiviat, 2004

state prior to invasion, invading macrophyte species could potentially enrich or endanger ecosystems. For example, if non-native plants colonize algae-dominated, poor quality waters devoid of macrophytes and with low biodiversity (Scheffer et al., 2001), the invasion could result in reduced phytoplankton blooms and introduce habitat structure of benefit to faunal diversity (Jeppesen and Sondergaard, 1998). However, exotic macrophytes invading freshwater systems rich in native vegetation generally have a negative impact on native plant abundance, on associated faunal diversity and alter ecosystem functioning (Wells et al., 1997; Stiers et al., 2011; Simberloff et al., 2013). Thus, the impact of invading macrophytes depends on the invader and on the state of the ecosystem at the time of invasion.

1.2. When does an introduced plant become invasive?

When a species first arrives in a new ecosystem, there is usually little indication whether it will be invasive or not, unless it is already known to be invasive elsewhere in the world, or unless its growth, reproduction and dispersal capacities are well understood. Invasiveness usually depends on the interaction of local habitat features and the alien aquatic plant traits, which together determine whether excessive growth of IAAPs will occur (Alpert et al., 2000).

Disturbed systems are generally more susceptible to successful invasion (MacDougall and Turkington, 2005), partly because disturbances increase fluctuations in resource availability (Davis et al., 2000). Flood control systems and navigation conveyances are subject to periodic management to guarantee water flow and vessel hull clearance. This management often involves the mechanical removal of much or all macrophytes, which physically creates open spaces and stimulates the release of nutrients. Similarly, the oligotrophication of formerly phytoplankton-dominated waters provides open habitats, and IAAPs often successfully colonize such open waters (Hussner et al., 2010). Another factor affecting invasion success is biotic resistance provided by native vegetation (Levine et al., 2004). However, the evidence for competitive biotic resistance by species-rich vegetation is weaker for freshwaters than for marine and terrestrial ecosystems (Alofs and Jackson, 2014). Instead of species richness, increasing vegetation density reduces the establishment probability of some, but not all, IAAPs (Capers et al., 2007).

Beside these effects of habitat quality, various other mechanisms for IAAP success include competition, enemy release, evolution of increased competitive ability, mutualisms, novel defence mechanisms, allelopathy, phenotypic and physiological plasticity, naturalization of related species, empty niche, fluctuating resources, opportunity windows and propagule pressure (Riis et al., 2010; Fleming and Dibble, 2015; Hussner et al., 2015). This multitude of mechanisms illustrates that invasion is not a simple process. Often several factors are required to co-occur in time and space in order to trigger plant invasiveness.

Time lags between the introduction of an alien species and the start of their invasive behaviour are a well-known phenomenon in invasion biology (Kowarik, 1995; Crooks, 2005; Heger and Trepl, 2003). Thus, the timing of many invasion-related events and processes are difficult to forecast and the time course of invasions can include long initial time periods, sometimes decades, of apparent “innocuous” establishment (Kowarik, 1995; Simberloff, 2003; Crooks, 2005). Data on the lag period between introduction and invasiveness are almost absent for IAAPs. In New Zealand, IAAPs generally perform well directly after being introduced (Howard-Williams, 1993). In addition, one of the shortest lag periods after introduction and invasiveness is for *Hydrocotyle ranunculoides* in the UK, where the plant was first listed for sale in 1989 and found to be invasive in Essex (SE England) in 1991 (Newman, pers. observation). Conversely, the rapid invasiveness of IAAPs has not been found in Germany, where invasion speed differs greatly amongst different IAAPs (Hussner et al., 2010). Similarly, on a European scale, the current distribution of introduced plants classified as invasive by the European and Mediterranean Plant Protection Organization (EPPO) is not correlated with the number of years since the introduction of the plant (Hussner, 2012). Thus, not all species that have been classified as (potentially) invasive to European freshwaters have become invasive across the European continent yet. For instance, the two highly invasive species *Ludwigia grandiflora* and *Ludwigia peploides*, which are widespread in France, are still limited in their distribution within the whole of Europe (Hussner, 2012; Thouvenot et al., 2013). Moreover, some highly invasive species, such as *Egeria densa* or *Hydrilla verticillata*, have shown their invasiveness only on limited spatiotemporal scales. Both species still have a limited distribution within Europe, despite being introduced >100 years ago (Hussner et al., 2010; Hussner, 2012), even though the south European climate matches that of their highly invasive ranges in the US and New Zealand (Howard-Williams, 1993; Langeland, 1996; Hofstra et al., 2010). This suggests that taking action (e.g. prevention, eradication, import and trade prohibitions) requires careful consideration of the risk of "waiting to see" versus assuming that ultimately there is likely to be a costly impact. But predicting the probability of invasion success is extremely difficult due to the number of potential mechanisms involved. Recently, aquatic weed risk assessment models (AWRAM Champion and Clayton, 2001; Gordon et al., 2012) have been developed, which will help to identify potential IAAPs by using a range of parameters.

1.3. Social and political context of freshwater invasions: implications for management

Humans have a prominent role in the invasion process in freshwater ecosystems. People change river features and water regimes, act as vectors and pathways for introductions (accidental or intentional), suffer the consequences of IAAPs impacts, and possess the capacity to act and make decisions for the prevention, and management of these species. IAAPs can be viewed as a socioeconomic problem that requires solutions from economics and sociology (Perrings et al., 2000, 2002; García-Llorente et al., 2008; Estévez et al., 2015). According to García-Llorente et al. (2008), different stakeholder groups may have remarkably different perceptions about the impacts of, and benefits generated by IAAPs, and they display different attitudes towards IAAP introduction or eradication. This fact deserves special attention in the decision-making process. It should be noted that most stakeholders and decision makers have a limited perception of the problem and, therefore, education and public awareness campaigns are vital for any successful management of the problems associated with IAAPs. To be effective, educational and informative programs should be tailored to specific target stakeholder groups (García-Llorente et al., 2008). The success of IAAP control and eradication, as well as the policies governing their management in general (e.g. prevention, inspection regulations, voluntary codes of conduct, or economic incentives to reduce threats), is highly dependent on the acceptance and support by all affected stakeholders (Touza et al., 2014). Finally, policy is especially challenging when invasive species generate income for certain stakeholders, and impose damage on others. For example invasive species may benefit stakeholders such as plant nurseries, gardening and landscape design firms, garden owners and phytoremediation advocates, whereas they financially damage others like riparian property owners and nature conservation agencies (Touza et al., 2014). Furthermore, as water quality degrades or the quantity available has to meet rising demands over time, competition among water users intensifies. State boundaries rarely conform to the parameters of ecological units and water and IAAPs do not recognize political boundaries either (Mitchell, 1996). Important watersheds and internationally significant natural areas are often transected by national boundaries. The management of transboundary waters imposes considerable challenges (Riley, 2009) and without an appropriate management regime for plant invasions for the whole ecological unit or landscape scale, there is an increased risk of conflicts and unsuccessful management interventions (EPPO, 2014; Lim, 2014). This requires the development of inter-state and inter-governmental or inter-jurisdictional collaborations and consensus on threats and appropriate actions to manage IAAPs. This aim of this paper is to provide an overview of the existing knowledge on the management of IAAPs and to provide guidance for choosing appropriate management techniques.

2. Methodology

We reviewed the scientific literature, grey literature, and listed best practices from Europe, the USA and New Zealand to provide guidelines that improve the management of IAAPs. Additionally, unpublished papers, reports and expert knowledge were used as they provide information on both successful and unsuccessful management efforts. The expert knowledge of the contributing authors was discussed during a workshop in Düsseldorf, Germany in October 2015. We provide examples of where these approaches have been used successfully, and highlight problems for which further solutions need to be developed.

2.1. Definitions

2.1.1. Freshwater aquatic plants

Several authors have defined 'aquatic plants' and their growth forms (e.g. Sculthorpe, 1967; Cook et al., 1974; Cook, 1985; Symoens et al., 1982; Den Hartog and Segal, 1964; Den Hartog and van der Velde, 1988). Recently, Bolton (2016) defined plants as organisms which carry out chloroxygenic photosynthesis, including algae and cyanobacteria. Here, we consider aquatic vascular plants visible to the naked eye, i.e. without magnification (Symoens et al., 1982). Freshwater vascular plants in a broad sense include hydrophytes and amphibious plants, but also some helophytes "whose photosynthetically active parts are permanently, or at least for several months each year, submerged or floating on the water surface" (Cook et al., 1974).

Hydrophytes can be divided into species that grow (i) under water (submersed hydrophytes – including elodeids e.g. *E. densa*, *Lagarosiphon major*, myriophyllids e.g. *Myriophyllum spicatum* or *Cabomba caroliniana*, parvopotamids e.g. *Stuckenia pectinata*), (ii) species that reach the water surface, but are rooted in the sediment (rooted aquatic plants, forming floating leaves e.g. batrachiids, most magnopotamids e.g. *Aponogeton distachyos*, nupharids and nympheids), and (iii) species that are free-floating (pleustophytes) during at least part of their life-cycle (lemnids e.g. *Lemna minuta* or *Azolla filiculoides*, stratiotids e.g. *Pistia stratioides* or *Eichhornia crassipes*, trapids i.e. *Trapa natans*).

Amphibious plants can live submersed, floating at the water's surface or emergent in terrestrial habitats (glyceriids e.g. *Paspalum distichum*, nasturtiids – e.g. *Ludwigia peploides, L. grandiflora*), some even having distinct aquatic and terrestrial morphologies (alismatids, e.g. *Sagittaria latifolia* or *Myriophyllum heterophyllum* and *Myriophyllum aquaticum*, peplids e.g. *Crassula helmsii*).

Helophytes are rooted in moist or waterlogged sediment (e.g. *Alternanthera philoxeroides*), but the shoots themselves are rarely submersed. In this paper, we will only include helophytes that grow in habitat containing at least a bit of water on top of the sediment, such as *Ludwigia spp*.

In this paper we also group the aquatic plants into three categories based on their growth form, as different growth forms require different management. The categories are: free floating on the water surface; rooted in the sediment with floating or emergent shoots/leaves; sediment rooted or unrooted submerged plants

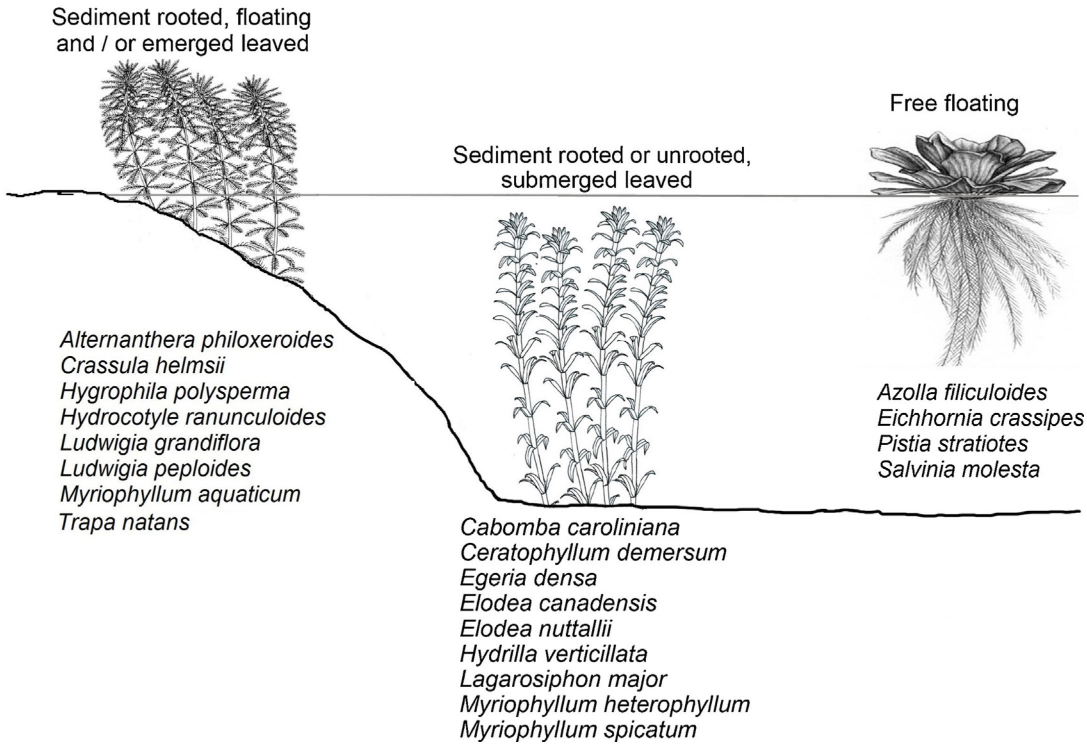

Fig. 1. Growth forms of IAAPs (listed in Table 1).

with their shoots (almost) completely submerged (for a simplified scheme, see Fig. 1).

2.1.2. Terminology

We have followed the definition of Pysek et al. (2014) and considered plant species as alien when they have been introduced into a new area by intentional and unintentional human involvement, or have spread without human involvement into new areas from areas where they are alien. Some of these introduced plants become naturalized, produce reproductive (vegetative and sexual) offspring and show their capacity to spread over large areas and become invasive (according to Pysek et al. (2014)). In this review, we consider "invasive" as a phenotypic trait of a species proliferating abundantly in a given (invaded) habitat. Even though native plants might show an invasive behaviour, we consider a selection of the most invasive alien aquatic plants, which have serious economic and/or ecological impacts.

3. Prevention, early detection and biology of IAAPs

3.1. Prevention is cheaper than control

Preventing the introduction and further spread of IAAPs seems to be the easiest way to reduce current and future negative impacts and management costs associated with IAAPs. The aquarium trade, including internet and mail order shipping (Maki and Galatowitsch, 2004), acts as the most important pathway for IAAP introductions (Brunel, 2009; Hussner et al., 2014a), but other less regulated pathways are involved as well (e.g. Cohen et al., 2007). Both international and national legislation, as well as voluntary codes of conduct (Verbrugge et al., 2014) can help to prevent intended and unintended new introductions of IAAPs and additionally reduce the risk of secondary releases.

Lists of (potential) IAAPs for each region are required, both for prevention and prioritizing actions, and these must be based on existing knowledge about actual or potential impacts, IAAPs biology, habitat and climatic requirements of the IAAPs and the ability to manage incursions. The Aquatic Weed Risk Assessment Model (AWRAM), which was developed in New Zealand (Champion and Clayton, 2001), provides such a tool for identifying potential IAAPs, and was successfully modified for the US (Gordon et al., 2012). The AWRAM classification of species is based on weed biology, historical data, and the environmental tolerance of the species. However, regulation of species is mostly based on a reactive, instead of a proactive approach. Non-native species are 'innocent until proven guilty', and management often targets invasive, problematic species well after they have become established.

3.2. Early warning and rapid response

Early Detection and Rapid Response (EDRR) and Early Warning and Rapid Response (EWRR) programs can significantly reduce the negative impacts of plant invasions and are crucial for effective management and successful eradication (Genovesi et al., 2010). Monitoring sensitive sites (EPPO, 2014), mapping and reporting new infestations, involving the public (citizen science, identification and mapping Apps, e.g. Adriaens et al., 2015) are all key actions within many national strategies. However, species identification is often difficult, which limits the applicability of early detection methods. This is partly the consequence of the high phenotypic plasticity of many aquatic plants in response to environmental factors and site conditions (e.g. Arber, 1920; Dorken and Barrett, 2004; Riis et al., 2010; Eusebio Malheiro et al., 2013) and hybrids between native and alien plants, e.g. within the genus *Myriophyllum* (Thum et al., 2011; LaRue et al., 2013). Because of their high phenotypic plasticity, plants of one species sold in trade can differ morphologically from those growing in the wild. Some groups (i.e. families, genera) of aquatic plants that include both invasive and non-invasive species are not satisfactory taxonomically resolved, e.g. *Myriophyllum* spp. and *Hydrocotyle* spp. (Van de Wiel et al., 2009; Ghahramanzadeh et al., 2013). Novel identification methods include genetic markers and other molecular tools, which in conjunction with reliably identified reference material, can be used to assist in the accurate identification of new incursions (Van de Wiel et al., 2009). Apps have also been developed to enable people to identify and report geolocated sites of IAAPs, which enables rapid response and facilitates mapping of invaded areas (http://aquaplant.tamu.edu/plant-identification/; http://www.helpdeskwater.nl/algemene-onderdelen/structuur-pagina'/stuur/@41290/iwaterplant-app/; http://www.rinse-europe.eu/smartphone-apps).

3.3. Importance of knowledge on weed biology for successful management

3.3.1. Investigating the ecosystems after IAAPs introduction

Once an IAAP has been reported in a new water body, there is a need to investigate its potential ecosystem impacts (within that system, Thouvenot et al., 2013) in a timely manner to provide the best opportunity for developing and implementing a successful and species-selective management programme. The focus of investigations should include delimiting the IAAP infestation, assessing its ecology, reproductive capacity and dispersal mechanisms (Ruaux et al., 2009; Garcia et al., 2015; Heidbüchel et al., 2016; Redekop et al., 2016), as well as documenting the native flora and associated fauna that may be impacted by the IAAP and the uses of the infested site. This latter information is particularly important to adequately assess the pros and cons of potential management tools for the IAAP on native species and ecosystem function. In addition, the native seed (or propagule) bank must be assessed, to develop an understanding of the chances of successful re-establishment of native vegetation from local propagules following management of IAAPs (De Winton et al., 2000). All this information must be taken into account while preparing the management plan for the IAAP in a specific habitat.

3.3.2. Vegetative and sexual reproduction of IAAPs

The successful management of IAAPs requires comprehensive knowledge on the biology of the target species. In particular successful management can be aided by understanding the following biological attributes:

(a) The regeneration capacity of plant fragments.
(b) The timing of development and potential dispersal of overwintering or storage organs (e.g. turions, winter buds, root crowns, rhizomes).
(c) The timing and development of seed production and requirements for seed germination.

Usually IAAPs have a high regenerative potential as they regrow from small (>1cm) plant fragments (Hussner, 2009; Boedeltje et al., 2003; Kuntz et al., 2014). Additionally, some IAAPs produce overwintering organs like turions, which remain viable for years (e.g. in *H. verticillata*, Netherland, 1997; Hofstra et al., 1999). Viable seed production has been rarely observed (or investigated) for IAAPs and may be almost absent in submerged IAAPs, such as most Hydrocharitaceae in Europe (Wolff, 1980), but for several floating and emergent IAAPs seed production and germination have been reported (Ruaux et al., 2009; Haury et al., 2012; Hussner et al., 2014b; D'hondt et al., 2016).

The success of management intervention depends on the likelihood of any regrowth and the unintended spread of the managed IAAPs from vegetative and/or sexual plant organs. Clearly, knowledge of the presence of overwintering organs and seeds is necessary for the development of a suitable management plan. The presence of IAAP seed banks and overwintering organs may limit the success of short-term eradication programs.

3.3.3. Dispersal of IAAPs

Aquatic plants predominantly spread via vegetative means (Boedeltje et al., 2003; Barrat-Segretain, 1996; Heidbüchel et al., 2016). Plant fragments and seeds easily disperse by waterflow and other vectors (Jacobs and MacIsaac, 2009), resulting in the spread of IAAPs. Inappropriate management techniques (for the target IAAP and site) can boost the spread and invasive success of IAAPs, for instance when management releases a large number of plant fragments (Anderson, 1998). The regrowth from these fragments can cause the rapid re-infestation of recently managed waters. Furthermore, vegetative propagules and seeds can be transported to new waterbodies as 'hitch-hikers' on equipment (e.g. boats and diggers; Johnson et al., 2001; Bruckerhoff et al., 2015) and can result in the unintended spread of IAAPs into formerly un-infested waters. The same fragments, overwintering organs and viable seeds can also cause the recolonization of managed water bodies. However, suitability of receiving habitat, such as substrate stability and wave energy, can affect successful establishment (Wittmann et al., 2015a). Overwintering organs and seeds of at least some IAAPs can last in the sediment for years (Hofstra et al., 1999), which may necessitate the development of a long-term (decadal) management programme. In general terms the development of a successful management programme must take into consideration the specific characteristics of the IAAP, the waterbody it has invaded and the outcome that is sought (see Section 3.5).

3.3.4. Different growth forms of IAAPs require different management methods

IAAPs occur in different growth forms (Fig. 1), which require consideration when potential management solutions are sought. For example, floating species, such as *E. crassipes*, can be more readily scooped out of the water as a whole, either by boats or by land-based vehicles when the system is narrow or shallow (e.g. Laranjeira and Nadais, 2008). Furthermore, a biomass reduction of some free-floating species has been achieved using host-specific biological control agents (Coetzee et al., 2011b), while similar levels of control have not been achieved for submerged plants. Mowing boats, harvesters and also cutters with blades are widely used to reduce the biomass of submerged species. However, eradication of sediment rooted IAAPs by such mechanical means is unlikely, as both shoots and roots must be removed for successful eradication. In addition mechanical harvesting can produce large numbers of plant fragments, which possess a risk for further, unintended spread of the species (Anderson, 1998).

In general, a variety of management techniques have been utilized worldwide in order to completely remove rooted IAAPs from bank and/or open water habitats with variable success. An overview of management methods currently in use, their pros and cons and relative costs, and an indication of the likelihood of successful management of IAAPs is provided in Section 4.

3.4. The crucial role of the habitat type on the potential success of management strategies

Aquatic habitats are highly diverse, and aside from general classifications into flowing or stagnant waters, small or large water bodies, numerous habitat attributes must be considered when assessing the appropriateness of weed management options. In any case, site accessibility is required regardless of the control method utilized, both for the management activity itself but also for monitoring (i.e. assessing effectiveness on the IAAP and any off-target impacts) to inform follow-up or on-going management actions.

Standard management approaches in Europe to date, namely mechanical and chemical control, have been more successful when applied in smooth and straight physical structures (e.g. stormwater drains) compared with natural systems. Bank access and stability in such man-made habitats often also enables machinery access to remove and transport harvested plant material. In contrast, as for most IAAP management in rivers, lakes and reservoirs, trees on the banks, but more importantly structures, snags, and other obstacles within the waterbody greatly impede mechanical harvesting.

In many cases, invasive aquatic weed management also includes dredging, for example in ditches colonized by *Ludwigia* spp. or *M. aquaticum* (Plant Protection Service, 2011b,c).

In coastal areas, there is a unique possibility to use sea water to regulate the distribution of invasive freshwater plants (i.e. inducing salt stress) by opening a barrier weir between the sea or estuarine waters and freshwaters (Grillas et al., 1992; Dandelot et al., 2004; Thouvenot et al., 2012).

3.5. *Defining the goal: containment, reduction, eradication*

There are different reasons why IAAP management might be considered once a species has been introduced into a new site, such as when the IAAP is likely to have detrimental impacts on the ecosystem and/or society. As the ecosystem impacts of IAAPs can differ substantially among species, the type of intervention depends largely on invasiveness and the dispersal potential of the IAAP, and the characteristics of the waterbody. To enable selection of the most appropriate management technique(s), defining the goal or outcome sought from management is of primary importance.

The goal or outcome may vary depending on the site (i.e. flowing, interconnected waterbodies, vs lakes with no significant water outlets) and the feasibility of achieving the goal with the management tools or methods available. Moreover, differences in legislative requirements can drive decisions on management techniques, which may have an effect on the management outcome. However, broadly defined goals fall into three different categories of containment, reduction or nuisance control and eradication.

I. Containment

Containment reflects the goal of ensuring the IAAP does not spread from the infested site. This may be a goal on its own, perhaps because there are no effective tools to control or eradicate the IAAP but its threat to adjacent aquatic systems is recognized, or when IAAPs are considered better than no plants in the system, or it may be a goal that is part of a larger management programme. For effective containment the potential vectors and pathways for plant dispersal must be identified and interrupted. Examples include the use of boats, water sports equipment and fishing gear, which must be checked for any IAAP fragments and cleaned to minimize the risk of plant transfer (Bruckerhoff et al., 2015), or perhaps bans are put in place to prevent the use of an infested waterbody for some activities. For example, if motorboats are determined to be a prime vector for IAAP spread, their use could be banned on the infested lake; alternatively weed cordons can be erected at key boat ramps to minimize the risk of new weed fragments being introduced to, or removed from, high risk waterbodies (Alix et al., 2009; Lass, 2012). When no appropriate eradication or management methods are available for a rapid response to an IAAP infestation, containment could be chosen until suitable management options are developed and validated for use (Fig. 2).

II. Biomass reduction, or nuisance control

When the IAAP has already spread extensively throughout a system, eradication becomes increasingly difficult or impossible (Wilson et al., 2007a; Sarat et al., 2015a). Large-scale weed reduction methods will most likely also affect other functional biotic groups and processes within the ecosystem (Hofstra and Clayton, 2014; Habib and Yousouf, 2014). However, if the IAAP species is not directly a major threat for the invaded system and its surroundings, or if no suitable management method for eradication is available for the species of concern, biomass reduction of the IAAP could be beneficial to reduce the negative effects of the IAAP, for instance on amenity values (e.g. access for boating or fishing) and utility functions. Biomass reduction may also promote native aquatic plant species (Hofstra and Clayton, 2014) and still safeguard the ecosystem functions that the aquatic plants provide. Furthermore, in many cases, total eradication can be achieved with the appropriate methods only after the successful biomass reduction to a certain level, making eradication achievable (Fig. 2).

III. Eradication

In some cases, the responsible authorities may set the management goal to eradicate the IAAP, for instance, the case of the first population of *L. grandiflora* in an old branch of the Leda River, Germany. The smaller the IAAP infestations, including propagules or overwintering organs in the infested water body, the greater the probability of a successful outcome. The California (USA) Hydrilla Eradication Program, began in 1976, is probably the largest scale, successfully sustained eradication programme anywhere and relies on multiple methods to eliminate biomass, coupled with extensive monitoring (Kratville, 2013), long-term treatment strategies and resources. Consequently, the eradication of large IAAP stands may only be achieved after IAAPs biomass reduction (Fig. 2) and long-term commitment.

4. Management options for IAAPs and their pros and cons

The existing management options can be divided into physical (including mechaniical control), biological and chemical control tools (Tables 2–4). The options vary greatly in their scale of application, costs and effectiveness. To decide what the best management option is within different aquatic systems can be challenging. Each option has its own advantages and disadvantages, and both the species and habitat attributes as well as the desired outcome(s) must be taken into account. However, decisions on management approaches are also largely driven by government and legislation, as for example the use of herbicides and biological control agents (except widely stocked grass carp) for the control of IAAPs are not permitted in the European countries (except the UK). Consequently, there are differences in the management techniques that can be used in specific situations based on the geographic region.

4.1. *Mechanical and physical control methods (Table 2)*

Mechanical control methods are the most widely used measures to control alien and native weeds in Europe. Their success varies considerably between the target species (free-floating, emergent and submerged macrophytes) and different habitats (e.g. Lambert et al. (2010) for *Ludwigia* spp.). Mechanical management methods usually produce large numbers of plant fragments, posing a high risk that the targeted IAAP is spread within and/or between sites (depending on whether or not machinery is, or can be adequately cleaned to avoid decontamination).

4.1.1. *Mechanical harvesting and cutting of submerged weeds*

Mechanical harvesting is commonly used, particularly where IAAP beds cover large areas. Harvesting is considered to be most effective for the control of free-floating IAAPs, such as *Eichhornia crassipes*, *Pistia stratiotes* and *Azolla filiculoides* (Laranjeira and Nadais, 2008).

The use of cutter boats is probably the most widespread method for managing submerged IAAPs in Europe (Podraza et al., 2008; Zehnsdorf et al., 2015); while the use of weed harvesters is common in the USA (Gettys et al., 2014). Using cutter boats to reduce weed biomass (or volume) without collection and disposal is relatively cheap compared to other control measures, but cutting of submerged plants is not species specific and eradication of IAAPs is not achievable. Cutting depth may often be limited to a maximum

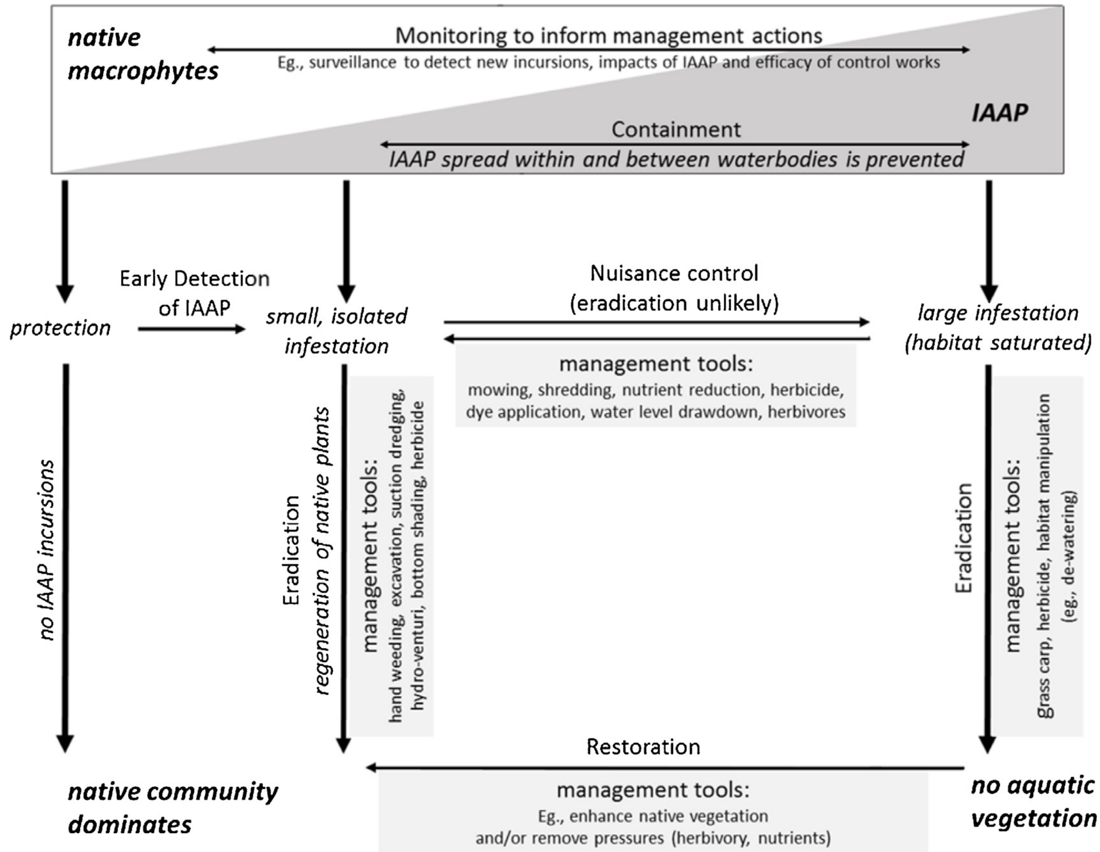

Fig. 2. IAAP management diagram illustrating the relationships between the state of the ecosystem, the management goals, outcomes and tools. The upper banner represents the state of the ecosystem, with the balance of native (white) and IAAP (dark grey) components informing the management goals or actions and examples of tools (light grey boxes) that can be used to achieve outcomes.

water depth of approximately 2 m (Barrett et al., 1999), however some cutter/harvesters in the USA can cut to 5m depth (http://www.clearwaterharvester.com/). Cutting using trailing V-blades is less restricted by depth. With this method, a V-shaped blade is pulled behind a boat and dragged across the sediment like a shovel, cutting the plants on, or just below, the sediment (De Haan et al., 2012) and thereby also uprooting some plants from the sediment. V-blades can be more destructive to the IAAPs and ecosystem than other cutters and harvesters. However, even V-blades are unlikely to completely remove the root system of all IAAP shoots. In the Pacific Northwest, submerged IAAPs have also been mechanically managed using rotovators, which destroy the stem bases of the plants (Madsen, 2000). Like V-blades, rotovating is non-selective and greatly disturbs the sediment and the habitat of benthic organisms, particularly invertebrates.

Mowing and cutting generally produce a large amount of plant fragments, which can easily disperse and form new stands of vegetation. Therefore, a drift barrier may be required to minimize the unintended dispersal of the target species in running water (Haury et al., 2010) and when the IAAPs are limited in their distribution within a water body. In those situations where the IAAP has limited distribution (i.e. not saturated the habitat), there is benefit in removing the cut material to minimize regrowth and dispersal, ideally utilizing boats combining cutting and harvesting (removal) on one platform. Logistical limitations, such as access, offloading and disposal sites for harvested biomass can extend both working time and management costs. Due to the rapid growth of IAAPs, the cut shoots can grow back to the water surface within a few days (Howard-Williams et al., 1996; Podraza et al., 2008), depending on the initial cutting depth. In addition, depending on the weed species, cutting depth and seasonal timing, one cut can provide effective control for a season.

In waters saturated with IAAP biomass there may be no need for harvesting the biomass straight after cutting (depending on what other potential impacts there are from leaving it in the system). Floating cut plant material can be pushed to the shore and dumped, which will reduce the management costs, but will also likely result in local odour issues as the biomass rots onshore, and may then reduce amenity values associated with the waterbody. Landing sites will be required for shoreline disposal of the biomass. In New Zealand and the US disposal is becoming increasingly problematic with regulations limiting where IAAPs can be disposed, with particular concern over the potential for contaminants (e.g. such as trace metal element pollution that can naturally accumulate from geothermal waters (Aggett and Aspell, 1980; Robinson et al., 2006; Clayton and Tanner, 1994)) in harvested weed resulting in contaminated dump sites (Khang et al., 2012).

4.1.2. Mechanical excavation

The use of excavators for IAAP control, while not species specific, can be effective for eradicating some IAAPs. However the machinery does have a limited working reach, confining the utility of this method to narrow waters, such as ditches, channels, drainage or irrigation systems, ponds or small rivers. Excavators can be used either for harvesting floating plants, or by digging rooted floating-leaved, submerged and emerged plants with rhizomatous mats (like *H. ranunculoides*, *Ludwigia* spp. or *M. aquaticum* (Pot, 2003; Plant Protection Service, 2011a,b,c). Excavators were used in addition to hand weeding for the successful (>99% biomass) reduction of *L. grandiflora* in an old river branch (Hussner et al., in press).

Table 2
Mechanical control methods of the most invasive IAAPs (listed in Table 1).

Control method	Target plants	Comments	References
Mechanical harvesting and excavation	free floating species surface reaching submerged species	- not species specific - eradication of free floating species achievable	Clayton, 1996 Hummel and Kiviat, 2004; Laranjeira and Nadais, 2008
Pulverising and shredding	floating rafts or accumulations of submerged weed drift	- not species specific - in water disposal of pulverized or shredded weed results in sinkage, can elevate water nutrients and decrease oxygen	Matheson, 2014 Greenfield et al., 2007
Mowing by boat	all submerged species	- not species specific - manage large areas compared with manual methods	De Winton et al., 2013
		- biomass reduction but fast regrowth of plants, eradication of a species achievable in combination with other methods	Podraza et al., 2008; De Winton et al., 2013
Hand-weeding	many species, like *Alternanthera philoxeroides*, *Ludwigia grandiflora*, *Trapa natans* or *Lagarosiphon major*	- selective - labour intensive	De Winton et al., 2013
		- required scuba diving at water depth >1m	De Winton et al., 2013
		- eradication of a species is achievable in small infestations (e.g. new incursions, outlier populations)	Hummel and Kiviat, 2004; Kelly, 2006; Kelting and Laxson, 2010; De Winton et al. 2013; Clements et al., 2014; Hussner et al., in press
Hydro-Venturi	sediment rooted species *Cabomba caroliniana* *Myriophyllum heterophyllum*	- not species specific - uprooting of plants reduces regrowth - higher biomass reduction and less regrowth than after mowing	Dorenbosch and Bergsma, 2014
Suction-dredging	submerged species	- requires skilled operator and scuba - weed eradication is feasible on a local scale, or early incursion, particularly in conjunction with follow-up hand-weeding and monitoring	Clayton, 1996
Bottom-shading with benthic barriers and substrate	sediment rooted species *Myriophyllum spicatum* *Myriophyllum heterophyllum* *Elodea nuttallii* *Lagarosiphon major* *Cabomba caroliniana*	- benthic barriers: geotextiles and jute mats	Caffrey et al., 2010
		- not species specific, although jute matting may be selective (e.g., *L. major* excluded while regrowth from comparatively fine stemmed native charophytes occurred). Dependent on the aperture or gaps size in the weave of the jute that is used	Laitala et al., 2012; Bailey and Calhoun, 2008; Hofmann et al., 2013; Hofstra and Clayton, 2012; Schooler, 2008
		- eradication of *Lagarosiphon major* at six out of seven treated sites	Caffrey et al., 2010
Water-level drawdown	submerged species	- not species specific - variable results, dependent on scale and duration of drawdown	Clayton, 1996
		- usually high biomass reduction but rapid regrowth of IAAPs	De Winton et al., 2013
Dye application	submerged species	- not species specific - limited to small shallow waterbodies	Denys et al., 2014
Nutrient reduction	all species	- not species specific - may lead to reduced algae in the water, therefore increased water clarity and subsequently increased bottom depth limit for submerged weeds	De Winton et al., 2013

The combination of mechanical and manual removal is frequently used to reduce the biomass and distribution of *H. ranunculoides* in the Netherlands. The balance between these two methods varies, depending on the level of infestation and time of the year (Pot and van der Wal, 2000).

4.1.3. Hydro-Venturi (water jets)

The Hydro-Venturi ventilation is a new removal method, which has been successfully used for the control of *C. caroliniana* and *M. heterophyllum* (van Valkenburg, 2011; Dorenbosch and Bergsma, 2014). The essence of the method is to wash plants out of the sediment, including their root system and subsequently the plants can be removed as they afloat on the water surface. The number of fragments produced is lower than with conventional mechanical methods. It has been successfully applied in clay, sand, peat and soft organic sediments. The timing of this method is essential to guarantee successful biomass reduction (van Valkenburg, 2011; Plant Protection Service, 2011d). The method is particularly suitable for the management of plants with fragile shoots, like *C. caroliniana*, as it is a relatively gentle way of uprooting plants from the sediment. One of the disadvantages is the slow speed of work and the high water turbidity, however these are also disadvantages for most mechanical control methods.

4.1.4. Suction dredging (and diver-assisted suction removal)

This control method is limited to small populations of IAAPs of <0.1 ha (De Winton et al., 2013). Plants are removed with their root systems, which limits the potential for spread as well as for regrowth (e.g. *M. spicatum*; Boylen et al., 1996). In combination with subsequent hand weeding of plant regrowth, suction dredg-

Table 3
The most frequently used classical biological control agents for IAAPs (listed in Table 1).

Growth form	Target plant (species)	Control agent	Comments (host specific, management success, costs)	References
Free-floating	*Azolla filiculoides*	*Stenopelmus rufinasus*	- caused local extinction of Azolla. filiculoides in 81% of the release sites in South Africa	McConnachie et al., 2004
	Eichhornia crassipes	*Neochetina eichhorniae* and *Neochetina bruchi*	- control agents for biomass reduction, not strictly host specific and feed marginally on some Pontederiaceae, in use in US, Australia, Asia	Center et al., 2002; Julien and Griffiths, 1998
			- reduced *Eichhornia crassipes* to 5–10% of its peak level	Wilson et al., 2007b; Van Driesche et al., 2010
	Pistia stratioes	*Neohydronomas affinis*	- used for control in Australia and the US, feeds also on *Lemna* spec., *Spirodela* spec. and *Limnobium* spec. - completes its life cycle only on *Pistia stratiotes*	Neuenschwander et al., 2009; Dray and Center, 2002
			- eradicated *Pistia stratiotes* from sites in South Africa, or large reduction in target weed biomass	Cilliers, 1991
	Salvinia molesta	*Cyrtobagous salviniae*	- target biomass reduced in Australia, destroyed the largest *Salvinia molesta* population in Australia within one year	Julien et al., 2002; Room et al., 1981
			- reduced *Salvinia molesta* by 90% in nine month after its release	Flores and Carlson, 2006
			The following three agents are less effective than *Cyrtobagous salviniae*:	
		Cyrtobagous singularis	- first released as *C. salviniae*, subsequently determined as a separate species and considered less effective than *C. salviniae*	
		Samea multiplicalis	- feeds also on *Pistia stratiotes* and *Azolla filiculoides*	Julien et al., 2002
		Paulinia acumulata	- feeds and completes its life cycle on various floating aquatic plants	
Rooted floating and emerged leaved	*Alternanthera philoxeroides*	*Agasicles hygrophila*	- released in several states in the US, in Australia and NZ; host specific, requires warmer climate for high biomass reduction - defoliated *Alternanthera philoxeroides* stands, often completely	Center et al., 1999; Stewart et al., 1999; Buckingham, 2002; Julien et al., 2012
		Amynothrips andersoni	- released in several states in the US and Australia, also feeds on other plants in the US, but completes its life cycle only on *A. philoxeroides*, needs warmer climate for high effective biomass reduction	
		Arcola malloi	- released in Australia and New Zealand, but failed to establish; was not released in the US due to feeding on four *Amaranthaceae* in tests	
	Crassula helmsii		- no biological control agents have been released, but new studies revealed a variety pathogens and enemies of *C. helmsii*, that are potential candidates for testing	Varia and Shaw, 2011
	Hygrophila polysperma		- no biological control agents have been assessed for *H. polysperma*	Cuda and Sutton, 2000
	Hydrocotyle ranunculoides		- no reports of biological control, but *Lixellus* elongates has been found to feed exclusively on *H. ranunuculoides* in Argentina	Newman and Duenas, 2010

Table 3 (*Continued*)

Growth form	Target plant (species)	Control agent	Comments (host specific, management success, costs)	References
	Ludwigia grandiflora		- *Lysathia* spp. is considered as a potential control agent for *Ludwigia* spp.	Cordo and DeLoach, 1982
	Ludwigia peploides		- *Lysathia* spp. is considered as a potential control agent for *Ludwigia* spp.	Cordo and DeLoach, 1982
	Myriophyllum aquaticum		- Some potential biological control agents exists, eg., *Lysathia* n.sp. (used for control in Southern America), *Listronotus marginicollis*, *Argyrotaenia ivana*, *Choristoneura parallel*, *Parapoynx allionealis* and the fungus *Pythium carolinianum*, but none are in use.	Bernhardt and Duniway, 1984; Cilliers, 1999; Cordo and DeLoach, 1982
	Trapa natans		- numerous insects have been found on *Trapa natans*, but none have fulfilled the requirements of a potential control agent	Pemberton, 2002
Submerged	*Cabomba caroliniana*		- no biological control agents are in use, although some species have been assessed in Australia: - *Paraponyx disminutalis*: not host specific, prefers other species - *Paracles* spec.: not host specific - *Hydrotimetes natans*: host specific in laboratory and field trials	Schooler et al., 2012
	Ceratophyllum demersum		- no biological control reported	
	Egeria densa		- *Hydrellia* sp. Is considered as a potential control agent	Walsh et al., 2013
	Elodea canadensis		- no biological control reported	
	Elodea nuttallii		- no biological control reported	
	Hydrilla verticillata	*Hydrellia pakistanae*	- reduced *Hydrilla verticillata* biomass within a few years after its release	Balcunias et al., 2002; Grodowitz et al., 2003
		Bagous affinis	- sufficiently host specific; strict environmental requirements hinder the esatblishment of stable populations in various sites in the US	
		Bagous hydrillae	- narrow host range, no stable population development after its release in the US	
		Hydrellia balciunasi	- host specific, released in the US but was recovered only from a few sites	
	Lagarosiphon major		- candidate agents are being investigated	Baars et al., 2010; Martin et al., 2013; Earle et al., 2013
	Myriophyllum heterophyllum		- no biological control reported	
	Myriophyllum spicatum	*Acentria ephemerelly*	- not host specific and feeds on a variety of plants	Johnson and Blossey, 2002; Newman, 2004
		Euhrychiopsis lecontei	- also feeds on other *Myriophyllum* species	Newman et al., 2001; Newman and Inglis, 2009;

ing has eradicated IAAPs from small infestations (De Winton et al., 2013). Suction dredging can be highly species specific if it is carried out by a scuba diver. Similarly to the Hydro-Venturi technique, suction dredging works best in soft and sandy sediments, and also temporarily increases water turbidity (although less so than Hydro-venturi technique).

4.1.5. Manual harvesting (hand-weeding)

Hand-weeding is the most species-specific manual control method. It was used for the successful eradication of early infestations of e.g. *L. grandiflora* (EPPO, 2011), including the removal of plants from amongst native *Nuphar lutea* stands (Hussner et al., in press). Hand-weeding is also widely used for the eradication of early infestations and reduction of large population of IAAPs when mechanical harvesting is impossible (Bailey and Calhoun, 2008; Haury and Damien, 2014). However, for species with brittle shoots (e.g. *M. aquaticum*, *M. spicatum* and *H. ranunculoides*), this method is not always successful (Boylen et al., 1996; Kelly, 2006). Hand-weeding is limited to small infestations of IAAPs because it is labour intensive and cost intensive, and the likelihood of eradication from a site can be limited by the plant species (e.g. shoot strength for marginal aquatic plants), and the skill or technique of the SCUBA

Table 4
Chemical control methods of the most invasive IAAPs (listed in Table 1).

Growth form	Target plant (species)	Herbicide	Comments	References
Free-floating	*Azolla filiculoides*			
	Eichhornia crassipes	glyphosate	- 2·88 kg a.i.ha^{-1} killed mats in treated plots within seven days	Olaleye and Akinyemiju, 1996 Kathiresan and Deivasigamani, 2015, Martins et al., 2002
		imazamox	- 372 g a.i. ha^{-1} reduced biomass by 70%	Emerine et al, 2010
		diquat	- 130.8 g a.i. ha^{-1} reduced *Eichhonia crassipes* by 60% within one week after treatment, but the level decreased to 35% after eight weeks	Emerine et al. 2010; Martins et al., 2002; Lugo et al., 1998
		2,4-D	- 1.5 kg ha^{-1} 2, 4-D Na salt optimum doze for biomass reduction	Kathiresan and Deivasigamani, 2015; Martins et al., 2002
		penoxsulam	- > 24.5 g a.i. ha^{-1} reduced Eichhornia crassipes by 100 % within six weeks after the treatment	Wersal and Madsen, 2010
		paraquat	- 1.5 kg ha^{-1} 2, 4-D Na salt optimum doze for biomass reduction	Kathiresan and Deivasigamani, 2015
		bispyribac –sodium	- at foliar rates of >10 g a.e. ha^{-1} 95 % of the treated *Eichhornia crassipes* was controlled within 5 weeks after the treamtment	Glomski and Mudge, 2013; Mudge and Netherland, 2015
		flumioxazin	- not effective	Mudge and Haller, 2012
	Pistia stratiotes	glyphosate	- 3360 g a.i. ha^{-1} caused >99% biomass reduction	Martins et al., 2002
		2,4-D	- 1340 g i.a. h^{-1} resulted in less than 25% biomass reduction	Martins et al., 2002
		terbutryn	- 6 liters ha^{-1} in a 30% mix with water resulted in large scale biomass reduction, but no eradication	Cilliers et al., 1996
		imazamox	- 533 g ai h^{-1} reduced biomass by 70 %	Emerine et al., 2010
		imazapyr	- 250 g i.a. h^{-1} resulted in less than 25% biomass reduction	Martins et al., 2002
		diquat	- >460 g i.a. h^{-1} reduced biomass >97 %	Martins et al., 2002
		bispyribac –sodium	- 59 and 119 g ai ha^{-1} foliar application resulted in biomass reduction of up to 99%	Glomski and Mudge, 2013
		flumioxazin	- 70 g a.i. ha^{-1} reduced biomass by 90 %	Richardson et al., 2008; Mudge and Haller, 2012; Glomski and Netherland, 2013a,b
	Salvinia molesta	2,4-D	- 1340 g a.i. ha^{-1} did not reduce biomass	Martins et al., 2002
		diquat	- 960 g a.i. ha^{-1} reduced biomass >99 %	Martins et al., 2002
		imazamox	- 560 g a.i. ha^{-1} not suitable for control	Emerine et al., 2010
		imazapyr	- 250 g a.i. ha^{-1} reduced biomass <10 %	Martins et al., 2002
			- 560 g a.i. ha^{-1} reduced biomass by 39 %	Emerine et al., 2010
		glyphosate	- 3360 g a.i. ha^{-1} caused about 50 % biomass reduction	Martins et al., 2002
			- 2240 g a.i. ha^{-1} caused 89 % biomass reduction	Emerine et al., 2010
		bispyribac –sodium	- subsurface applications of 10 to 80 lg ai L^{-1} resulted in 8 to 69% biomass reduction - foliar applications of 20 to 60 g ai ha^{-1} reduced biomass by 28 to 73%	Glomski and Mudge, 2013
		fluridone	- 20 µg a.i. L^{-1} reduced biomass by >50 %	Mudge et al., 2012
		penoxsulam	- foliar treatments with 20 µg a.i. L^{-1} decreased biomass by up to 100 % after 12 weeks exposition time	Mudge et al., 2012

Table 4 (*Continued*)

Growth form	Target plant (species)	Herbicide	Comments	References
Rooted floating and emerged leaved	Alternanthera philoxeroides	metsulfuron-methyl	- 0.06 g a.i. L^{-1} reduced biomass by 100%	Clements et al., 2014
		metsulfuron	- several repeat applications are usually required to give adequate control	Clements et al., 2014
		glyphosate	- 2240 g a.i. ha^{-1} reduced biomass by 100%	Emerine et al., 2010
		imazapyr	- 560 g a.i. ha^{-1} reduced biomass by 99 %	Emerine et al., 2010
		triclopyr	- varying success in different studies	Hofstra and Champion, 2010; Dugdale and Champion, 2012
		imazamox	- 560 g a.i. ha^{-1} reduced biomass by 94 %	Emerine et al., 2010
		flumioxazin	- 35.6 g a.i. ha^{-1} reduce biomass by 90 %	Richardson et al., 2008
	Crassula helmsii	glyphosate	- about 50 % coverage reduction	Bridge, 2005
	Hygrophila polysperma	endothall	- 1.5 mg L^{-1} reduced biomass by uo to 90 %, but plants recover to prior biomass	Sutton et al., 1994
		glyphosate	- 4.2 kg a.i. ha^{-1} reduced biomass by 82 %	Fast et al., 2009
		imazamox	- 0.56 kg a.i. ha^{-1} reduced biomass by 82 %	Fast et al., 2009
		imazapyr	- 1.12 kg a.i. ha^{-1} reduced biomass by 95 %	Fast et al., 2009
		triclopyr	- 3.36 kg a.i. ha^{-1} reduced biomass by 100 %	Fast et al., 2009
		flumioxazin	- 158 μg a.i. L^{-1} reduced biomass by 90 %	Haller and Gettys, 2013
		penoxsulam	- < 250 μg a.i. L^{-1} reduced biomass by 90 %	Haller and Gettys, 2013
		trifloxysulfuron	- < 250 μg a.i. L^{-1} reduced biomass by 90 %	Haller and Gettys, 2013
		topramezone	- > 240 μg a.i. L^{-1} reduced biomass by 90 %	Haller and Gettys, 2013
		quinclorac	- > 400 μg a.i. L^{-1} reduced biomass by 90 %	Haller and Gettys, 2013
	Hydrocotyle ranunculoides	glyphosate	- 2.16 kg a.i. ha^{-1} reduced the coverage by about 50 %	Newman and Dawson, 1999
		2,4-D	- 4.23 kg a.i. ha^{-1} reduced coverageby >99%	Newman and Dawson, 1999
	Ludwigia grandiflora	glyphosate	- 2240 g a.i. ha^{-1} reduced biomass by 92 %	Emerine et al., 2010
		imazamox	- 560 g a.i. ha^{-1} reduced biomass by 80 %	Emerine et al., 2010
		imazapyr	- 560 g a.i. ha^{-1} reduced biomass by 93 %	Emerine et al., 2010
	Ludwigia peploides	n.a.		
	Myriophyllum aquaticum	fluridone	- not effective in concentration of 0.5 kg a.i. ha^{-1}	Hofstra et al., 2006
		clopyralid	- not effective in concentration of 1.5 kg a.i. ha^{-1}	Wersal and Madsen, 2007
		glyphosate	- 2240 g a.i. ha^{-1} reduced biomass by 94 %	Emerine et al., 2010
		triclopyr	- 4 kg a.i. ha^{-1} reduced biomass by 100 %	Hofstra et al., 2006
		endothall	- 15 kg a.i. ha^{-1} reduced biomass by 90 %	Hofstra et al., 2006
		dichlobenil	- 20 kg a.i. ha^{-1} reduced biomass by 99 %	Hofstra et al., 2006
		imazamox	- 560 g a.i. ha^{-1} reduced biomass by 81 %	Emerine et al., 2010
			- ≤ 281 g a.i. ha^{-1} not effective, 561 g a.i. reduced biomass by > 60 %	Wersal and Madsen, 2007
		imazapyr	- 560 g a.i. ha^{-1} reduced biomass by 93 %	Emerine et al., 2010
			- > 584 g a.i. ha^{-1} eradicated	Wersal and Madsen, 2007
	Trapa natans	2,4-D	- effective in high concentrations but native plants were also susceptible in North America	Hummel and Kiviat, 2004

Table 4 (*Continued*)

Growth form	Target plant (species)	Herbicide	Comments	References
Submerged	*Cabomba caroliniana*	n.a.		
	Ceratophyllum demersum	endothall	- 0.5 mg L^{-1} reduced biomass by 100 %	Hofstra and Clayton, 2001
		diquat	- 2 mg L^{-1} reduced biomass by up to 100 %	Hofstra et al., 2001
		dichlobenil	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
		tryclopyr	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
	Egeria densa	fluridone	- 10–26 μg a.i. L^{-1} reduced biomass by > 90 %	Parsons et al., 2009
		endothall	- up to 5 mg L^{-1} had no effects	Hofstra and Clayton, 2001
		dichlobenil	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
		triclopyr	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
		diquat	- 0.37 mg L^{-1} caused eradication	Skogerboe et al., 2005; Parsons et al., 2007
	Elodea canadensis	Diquat	- 0.09 mg L^{-1} reduced biomass by >90 %	Glomski et al., 2005
		Fluridone	- 5–10 ug L^{-1} Reduced biomass by > 90%	Alaska DNR, 2016
	Elodea nuttallii	n.a.		
	Hydrilla verticillata	dichlobenil	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
		triclopyr	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
		diquat	- variable results in the US	Blackburn and Weldon, 1970; Van et al., 1987; Glomski et al., 2005; Skogerboe et al., 2005
		endothall	- 0.5 mg L^{-1} reduced biomass by 100 %	Hofstra and Clayton, 2001
			- 2 mg a.i. L^{-1} reduced biomass by > 85 %	Poovey and Getsinger, 2010
		fluridone	- some *Hydrilla verticillata* populations became resistant against fluridone	Puri et al., 2009
		penoxsulam	- 100 μg a.i. L^{-1} reduced biomass by 84 %	Netherland, 2011
		bispyribac –sodium	- 100 μg a.i. L^{-1} reduced biomass by 88 %	Netherland, 2011
		bensulfuron	- 100 μg a.i. L^{-1} reduced biomass by 82 %	Netherland, 2011
		imazamox	- 100 μg a.i. L^{-1} reduced biomass by 73 %	Netherland, 2011
	Lagarosiphon major	diquat	- high effective	Clayton 1996
		endothall	- 0.5 mg L^{-1} reduced biomass by 100 %	Hofstra and Clayton, 2001
		dichlobenil	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
		triclopyr	- 2.5 mg L^{-1} was not effective	Hofstra and Clayton, 2001
	Myriophyllum heterophyllum	fluridone	- 20 μg a.i. L^{-1} reduced biomass by > 80 %	Glomski and Netherland, 2008
		penoxsulam	- 20 μg a.i. L^{-1} reduced biomass by > 80 %	Glomski and Netherland, 2008
		bispyribac -sodium	- 20 μg a.i. L^{-1} had no effect	Glomski and Netherland, 2008
		2,4-D	- 112 kg i.a. ha^{-1} reduced coverage by up to 100 %	Haug and Bellaud, 2013
		triclopyr	- 1.5 mg a.i. L^{-1} caused eradication	Getsinger et al., 2003
	Myriophyllum spicatum	triclopyr	- 1.5 mg a.i. L^{-1} reduced biomass by 88 %	Wersal et al., 2010
		2,4-D	- 1.5 mg a.i. L^{-1} reduced biomass by 88 %	Wersal et al., 2010; Madsen et al., 2015
		endothall	- 1.5 mg a.i. L^{-1} reduced coverage by >50 %	Parsons et al., 2004
		fluridone	- 10–26 μg a.i. L^{-1} reduced biomass by > 90 %	Madsen et al., 2002; Parsons et al., 2009; Valley et al., 2006
		diquat	- 0.37 mg L^{-1} caused eradication	Skogerboe et al., 2006

divers for submerged aquatic plants (Clayton, 1996). Hand- weeding is also used in addition to suction dredging and chemical control as part of an integrated weed management programme (De Winton et al., 2013). Regular hand-weeding can decrease the size of IAAP populations as well as their impacts in small water bodies (e.g. in a gravel pit, Coudreuse et al., 2009), but less so in large ones (e.g. Brière marsh, Haury and Damien, 2012).

4.1.6. *Bottom shading with benthic barriers and substrate*

Benthic barriers such as plastic foils, tarps or sheeting have been used for the control of submerged weeds in lakes (Boylen et al., 1996; Laitala et al., 2012), and for the control of *Ludwigia* spp. on river banks (Sarat et al., 2015b). However long lasting plastic or polyethylene sheeting may reduce dissolved oxygen beneath the sheeting, and may thus affect hydrochemistry and macroinvertebrate habitat (Ussery et al., 1997). More recently, biodegradable jute mats have been used as effective benthic barriers (Caffrey et al., 2010; Hofmann et al., 2013). Jute mats allow gas and water exchange at the water-sediment interphase and native vegetation is able to grow through the mats (Caffrey et al., 2010; Hofstra and Clayton, 2012). Depending on the species and the timeframe for weed management, biodegradation of the jute mats may mean that the jute needs to be replaced for sustained weed reduction (Hofmann et al., 2013). However it is also true, that biodegradation of the jute mat following suppression of the weeds and regrowth of native flora (Caffrey et al., 2010) provides a cost-effective solution without environmental pollution (i.e. no retrieval of plastic sheeting). Shading with benthic barriers is not species specific, and is usually limited to small scale management in static and slow flowing waters with limited wind/wave exposure, and is often used in restricted areas like harbors. Its local success in the longer term also depends on the likelihood of re-invasion by IAAPs if the plants persist in other parts of the same waterbody, and/or the re-invasion pathways and vectors are not managed. In addition sedimentation on an impermeable benthic barrier may, over time, provide new habitat that could be subject to weed invasion (Hofmann et al., 2013).

Next to foils and fabrics, substrates (e.g. sand) can be spread out over the sediment to control IAAPs hindering the regrowth of plants from tubers and seeds on a small scale (substrate capping). It can be used after mechanical harvest of IAAPs to minimize plant regrowth.

4.1.7. Water level drawdown

Water level drawdown is limited to waters with a controlled outflow like ponds or reservoirs and gated irrigation canals and flood control channels, so that the water level can be reduced to a level that the IAAP infested sites are drained to expose the target vegetation to either drying in summer or frost in the winter. The success of this technique is based on the mortality of all plant parts, hence the timing and length of water level drawdown is related to the target species (e.g. propensity to desiccate, extent of the weed beds). Consequently, weed eradication can only be achieved if the water level drawdown is applied for at least several months that either freezing or dryness affects the sediment to the depth where plant fragments are present (De Winton et al., 2013; Barrat-Segretain and Cellot, 2007). Range and frequency of drawdown can also be adjusted to prevent any nuisance weed growth, while for clear water lakes it can still allow deeper growing desirable vegetation (e.g. characean meadows) to remain intact (Clayton et al., 1986). Drawdown is sometimes applied in channel systems during periodic maintenance (Dutartre et al., 2006). In other cases, the reduced water level can be used for easy mechanical removal of IAAPs (Chapman et al., 1974). In the US, the application of soil-active, systemic herbicides to dry (dewatered, or drawndown) canals has become more common because it allows the herbicide to directly target the plant populations and greatly reduces movement of the herbicide into the water even when canals are re-filled.

4.1.8. Dye application

The use of blue dyes to control submerged macrophytes in shallow waters has been undertaken in the UK and The Netherlands. Blue, neutral or black dyes in static water effectively suppressed most submerged macrophyte species, but to date this method has not been effective on IAAPs. Dyes work by absorbing light, reducing chlorophyll efficiency and reducing photosynthesis. In general, the underwater light intensity reaching the plant has to be lowered to 1–4% of the surface irradiance to prevent plant growth, depending on the species' growth form (reviewed by Bornette and Puijalon, 2011). Regular monthly top-ups are required during the growing season to account for losses by dilution (rainfall) and UV degradation. It is not possible to partially treat a lake with dyes, the whole water volume must be treated. Dyes used are non-toxic food grade dyes and have no long-lasting effect on the ecosystem. Side effects have been observed on some *Myriophyllum* species where epiphytes have been controlled, increasing gas exchange and nutrient fluxes in the milfoil leaves, resulting in bigger, healthier *Myriophyllum* plants in some lakes (J. Newman, pers. observation).

4.1.9. Nutrient reduction

Although there are examples where invasive plants proliferate in oligotrophic systems (e.g. *Lagarosiphon major* in New Zealand, De Winton et al., 2009; *Myriphyllum spicatum* and *Potamogeton crispus* in Lake Tahoe, Wittmann et al., 2015a,b) many IAAPs particularly exhibit excessive growth in meso- to eutrophic conditions (Hussner, 2009, 2010; Coetzee and Hill, 2012). Therefore, lowering nutrient availability may benefit native aquatic plants at the expense of IAAPs in some locations. In practice, this method is frequently used to improve water transparency and stimulate submerged plant growth (Jeppesen et al., 2005; Bakker et al., 2013), but is not specifically targeted at reducing IAAPs. In fact, reducing nutrient levels in the water in lake restoration projects can stimulate the excessive growth of submerged plant species, including IAAPs such as *Elodea nuttallii*, as water transparency increases, given that the sediment contains enough nutrients, especially phosphorus (P) for plant growth (Perrow et al., 1997; Hilt et al., 2006). To prevent internal P loading, sediment capping techniques are being applied. Iron (Fe), aluminum (Al), calcium (Ca) or lanthanum-enriched benthonite clay (Phoslock©) can be added to the water column or sediment to bind P (Cooke et al., 1993; Spears et al., 2013). However, when starting with a system that is devoid of macrophytes, the first plant species to establish will benefit from reduced algal growth and improved water transparency, and these are most likely to be IAAPs such as *E. nuttallii* (Hussner et al., 2010; Immers et al., 2015) and *Myriophyllum spicatum* (Jacoby et al., 1994). Native species may take over in the longer term, but such events are rare. In France for example, a decrease of *Ludwigia* cover after decades of proliferation created suitable conditions for a restoration of native plant communities (i.e. Marais d'Orx in France; J. Haury, pers. observation).

4.2. Biological control (Table 3)

Biocontrol (biologial control) in general terms, is described as the use of one organism, the biocontrol agent, to reduce the vigour, reproductive capacity or density of another, the target weed (Cuda et al., 2008). The term biocontrol however, covers a variety of approaches, from classical to augmentive, from inundative to the use of generalist herbivores and even methods that are currently non-operational (Van Driesche et al., 2010). Newman et al. (1998) described the potential of native herbivores for biocontrol of an IAAP.

Classical biocontrol (CBC) can be defined as the introduction of a biocontrol agent into a region that is not part of its natural range to basically suppress populations of target weeds (Harley and Forno, 1992), with CBC agents usually sourced from the native range of the target weed. Augmentive biocontrol, is similar in ecological approach to CBC, but differs in that it refers to a biocontrol agent of local origin, i.e., supplementing numbers of a native or naturalized biocontrol agent to control a weed population. The aim is still to achieve sustained weed suppression as with CBC. An inundative approach usually refers to the use of a pathogen, delivered as a bioherbicide to produce a weed control response (Harley and Forno, 1992). The use of a naturally occurring fungus on aquatic plants in the USA to develop a mycoherbicide is an example of inundative biological control (Shearer, 2008a,b), where application results in plant collapse followed by a decline in the pathogen population (Shearer, 1994).

Generalist herbivores of aquatic macrophytes including insects, waterfowl and fish can have significant suppressive impacts on plant growth, and are considered here as non-operational forms of biocontrol – meaning that they are not actively released or stocked at known densities to achieve specific outcomes. The herbivorous fish grass carp (*Ctenopharyngodon idella*) are the exception, although they feed on a wide range of aquatic plants they are actively stocked and managed for aquatic weed control (Hofstra et al., 2014; Colle, 2014).

With each different approach and biocontrol agent there are a variety of different impacts and degrees of selectivity and effectiveness (Sections 4.2.1 to 4.2.4). In general, the use of a biological control agent that will be actively released for the control of IAAPs is widely restricted and often specific permitting processes must be followed prior to their release. These permission(s) require comprehensive information on the food preference, host range and reproduction of the control agent, to provide surety that there will be no host shifts or off-target impacts by the control agent after its release (Van Driesche et al., 2002).

4.2.1. Classical and augmentive biocontrol with insects

Insects have been used on a range of IAAPs in many countries (Forno and Julien, 2000; Cuda et al., 2008; Coetzee et al., 2011b). Newman (2004) stated that "In fact, many of the best and classic examples of biological control are from aquatic systems: alligator weed, water hyacinth, giant salvinia and water lettuce (McFayden, 1998)" and "However, all of the clearly successful programs have been with emergent or floating aquatic plants" while "the control of submersed plants has proved much more difficult". This has not changed recently, despite significant research investment in biocontrol (e.g. Schmitz and Schardt, 2015).

In the native range of IAAPs a few specialized insect species feeding on IAAPs are often present (McGregor et al., 1996). The specific relationships between these insects and their host plant provide the foundation for the development of CBC programs. Prior to the use of CBC agents host specificity studies are required to avoid host shifts of the control agent after its release (van Driesche et al., 2012). The utility of an insect species as a CBC agent is dependent on its potential to cause harm to the target IAAP. Impacts may include the devastation of plant stems, buoyancy organs and/or meristems of the aquatic weed resulting in biomass reduction (Gross et al., 2001; Newman, 2004). In particular, species belonging to Coleoptera (Chrysomelidae, Curculionidae), Lepidoptera (Arctiidae, Crambidae, Noctuidae) or Diptera (Chironomidae, Ephydridae) have proved effective, selective agents against IAAPs by providing very localised biomass reduction (Newman, 2004; Cuda et al. 2008), but the reduced biomass usually last only a very short term. In addition to the insects that have been deliberately released as CBC agents (Table 3), some other insects have been introduced with their host or by other events. For example, immediately after the first report of the invasive *A. filiculoides* in Germany, the host specific frond-feeding weevil *Stenopelmus rufinasus* was also found (Manzek, 1927), indicating their simultaneous introduction.

Insect herbivory is most effective in biomass reduction against emergent and floating IAAPs like *A. filiculoides*, *E. crassipes* and *P. stratiotes* (Table 3), but less effective for submerged IAAPs due to a number of factors (Coetzee et al., 2011a,b; Forno and Julien, 2000). Fish predation, insect migration to healthy plant patches and weed recovery while insects overwinter ashore reduce the chances of insect populations persisting in sufficient densities for multiple years and reduce the management success (Wheeler and Center, 1997; Ward and Newman, 2006; Grutters et al., 2015a). Furthermore, insects may require specific climatic conditions to sustain populations (Grodowitz et al., 1996). Because of these challenges, biological control of submerged IAAPs is best considered in an integrated approach alongside other types of weed management.

4.2.2. Inundative biological control with mycoherbicide

Several fungal plant pathogens have been found on aquatic plants, and some of these pathogens have been formulated as mycoherbicides and evaluated for their potential as inundative biocontrol agents for target IAAPs (e.g. Charudattan, 1973, 2010; Charudattan et al., 1980; Pennington and Theriot, 1983; Verma and Charudattan, 1993; Harvey and Evans, 1997; Shearer, 1994, 1997, 1998, 2001, 2008a,b, 2010, 2013; Shearer and Jackson, 2003, 2006; Joye, 1990).

The most prominent mycoherbicide candidate, in recent years, uses *Mycoleptodiscus terrestris*, which naturally occurs on several species of submerged aquatic plants (Shearer, 1996, 2010; Hofstra et al., 2009, 2012) and has recognized differences between isolates in the extent of disease symptoms and impacts that the isolates have on different submerged aquatic plants (Hofstra et al., 2004; Shearer, 2010). Development of *M. terrestris* as a mycoherbicide has been the subject of on-going research amongst different agencies and companies over several decades (Shearer, 2010). To date field success, defined as a decline in biomass of weed beds in a lake, has not been as great or predictable as expected from laboratory or mesocosm studies in the USA (Shearer, 1994, 1996) or New Zealand. Combining *M. terrestris* mycoherbicide with other treatments to assess its utility for integrated pest management has been investigated in growth chambers and mesocosm studies with promising synergistic results (Nelson and Shearer, 2005, 2009).

4.2.3. Herbivores – grass carp

Grass carp have a long history of use as biological control agents for submerged IAAPs (Sutton, 1977). Grass carp is the only fish species that consumes large amounts of vascular aquatic plant biomass (>100% of their body weight per day, depending on stocking densities, water temperature and age-class of fish (Wattendorf and Anderson, 1984; Pipalova, 2006)). Pipalova (2006) reported the eradication of aquatic vegetation after stocking 4–8.4 grass carp per metric ton of plant fresh weight. While understocking causes either no control or a selective reduction of aquatic plant species, overstocking results in a complete removal of all aquatic plants in the treated waters (Pipalova, 2006). Due to the risk of unwanted reproduction, infertile triploid grass carp are widely stocked in the USA (Pipalova, 2006). In fact, the triploid (sterile) grass carp have been a major, successful component of the Califorina Hydrilla Eradication Program (Kratfille, 2013). Even though grass carp prefer soft-tissue aquatic plants, they will consume all palatable plants that they have access to, in order of their feeding preference (Pine and Anderson, 1991; Dorenbosch and Bakker, 2011) and this must be considered (amongst other factors eg., ability to contain the grass carp (Dibble and Kovalenko, 2009; Hofstra et al., 2014)) before they are introduced to any new sites. In addition to stocking grass carp for weed control (reduction), grass carp have also been used to successfully eradicate submerged aquatic weeds (Rowe and Champion, 1994). If the restoration of native plant communities is a desired outcome or goal following weed eradication, grass carp must be removed to enable the regrowth of native vegetation (Tanner et al., 1990), however removal of grass carp (depending on the system) is a challenge and current impediment to their use (Hofstra et al., 2014).

4.2.4. Herbivores – non-operational biological control

Native herbivores can consume IAAPs and hence inhibit their establishment, growth and distribution, providing biotic resistance to invading plants. Several studies demonstrate that native generalist herbivores prefer IAAPs over native plants (Gross et al., 2002;

Parker and Hay, 2005; Parker et al., 2006; Grutters et al., 2015a; Redekop et al., in press), but sometimes the converse is observed (Xiong et al., 2008). For example, in the field, a strong preference of native coots (*Fulica atra*) for the invasive *E. nuttallii* has been documented causing localised reduction of weed biomass in Europe (Van Donk and Otte, 1996; Perrow et al., 1997; Irfanullah and Moss, 2004; van de Haterd and Ter Heerdt, 2007; Sarneel et al., 2014). In the USA, native beavers have reduced the abundance of invasive *M. aquaticum* by 90% through consumption (Parker et al., 2007).

For the genus *Myriophyllum*, a host range expansion was found for both native milfoil weevil (*Euhrychiopsis lecontei*) and the milfoil midge (*Cricotopus myriophylli*), where the herbivores perform even better on the exotic IAAP than on the native plant spcies (Borrowman et al., 2015). Cuda et al. (2008) entitled such native herbivores, which show potential as control agents for IAAPs, as neoclassical control agents.

Additionally, introduced herbivores may limit the spread and growth of IAAPs. This includes generalist herbivores in aquatic systems, such as coypu *Myocastor coypus*, which in its alien range feeds on alien aquatic plants like *Ludwigia* (Guichón et al., 2003) or *Hydrocotyle ranunculoides* (Hussner, 2007), and can reduce its cover dramatically (Haury and Damien, 2012). Similar consumption of IAAPs is also observed for introduced red swamp crayfish *Procambarus clarki* (Haury and Damien, 2012).

Aside from grass carp, other fish species like common carp (*Cyprinus carpio*), rudd (*Scardinus erythrophthalmus*), roach (*Rutilus rutilus*) and tropical fishes (various species of tilapia (*Tilapia* spp.) and silver dollar fish (*Metynnis* spp)) exert control on IAAPs. While for instance common carp increase the turbidity and uproot plants during feeding activities (Bajer et al., 2009) while grazing on invasive aquatic plants (Delbart et al., 2013), rudd actively feed on apexes of submerged aquatic plants and due to their selectivity may affect plant community composition (Kapuscinski et al., 2014). However, there is no evidence that common carp and rudd successfully control aquatic plants (Podraza et al., 2008). Furthermore, stocking these two species in high densities may hinder the successful restoration of native vegetation after eradication of IAAPs.

4.3. Chemical control

4.3.1. Herbicides (Table 4)

Herbicides, in general terms, provide a suite of control options that can be applied to achieve reduction or removal of a target species or group of plants, in different types of waterbodies including irrigantion channels, rivers, ponds, and partial or whole lake treatments (Getsinger et al., 1997, 2008, 2014; Netherland, 2014). Selectivity can be achieved by partnering the appropriate chemistry (product mode of action) and use profile (e.g., concentration and exposure time, CET, Getsinger et al., 2008) and application method (Haller, 2014) for the target IAAP, waterbody and management goals.

Products licensed for use around the world include the active ingredients: 2,4-D, bispyribac-sodium, carfentrazone-ethyl, copper, dichlobenil, diquat, endothall, flumioxazin, fluridone, glyphosate, haloxyfop-R-methyl, imazapyr, imazamox, metsulfuron methyl, penoxsulam, terbutryn and triclopyr (Table 4). Most herbicides that have been in use for several decades (e.g. diquat, endothall, fluridone and 2,4-D) have comparatively well-developed use profiles, which means that the product behaviour in aquatic systems is well understood as are the CET relationships for key target (and non-target) species (e.g. Netherland and Getsinger, 1995, 1997; Skogerboe et al., 2004; Clayton and Matheson, 2010). This information along with weed management goals for an IAAP enable the development of site specific and sometimes large scale treatment plans with predictable outcomes. In essence most herbicides act as chemical cutters (with the plant biomass decomposing in the ecosystem), and when used according to the product label have far less environmental impact than sustained mechanical cutting and weed removal, and are more cost effective than other control techniques (Netherland, 2014).

While often used for the management of nuisance weed growth, requiring regular or maintenance application (Netherland, 2014), herbicides have also been instrumental in weed eradication (Champion and Wells, 2014) in both small and large scale management programs. Yet despite these successful outcomes, the use of herbicides to control aquatic vegetation is prohibited in a number of countries (e.g. in many European countries). For example, in Europe, only glyphosate is authorized for regular use on emergent aquatic species. For submerged species in Europe there are no herbicides approved for use due to legislative restrictions, although it may be possible to obtain off label approvals for particular species in particular situations.

However, overreliance on herbicides with a single mode of action to control certain weeds can lead to the selection of weeds resistant to that mechanism of action. Although well recognized with large scale terrestrial crop production, the development of resistance was not anticipated in aquatic plants because of the predominantly vegetative reproduction exhibited by these species (Richardson, 2008). However resistance in aquatic weeds has become apparent with the use of fluridone on *H. verticillata* (Netherland, 2011; Netherland and Jones, 2015) and hybrid watermilfoil (*Myriophyllum spicatum x M. sibiricum*, Berger et al., 2015), hybrids have also been found to be less sensitive to 2,4-D than parental *M. spicatum* (LaRue et al., 2013). Development of resistance to terbutryn in coenocytic filamentous algae was also observed in the UK in the 1980s (Barrett et al., 1999). The first case is an example of selection for a resistant biotype, whereas the latter appears to be resistance developing suddenly in an exposed population after many years of successful treatment. Both mechanisms of resistance appear to be evident within aquatic macrophyte populations. The continued use of currently registered products and the development of new herbicides hinges on appropriate use and stewardship of aquatic herbicides to mitigate further resistance in IAAPs, balanced against the expense to register compounds for use in aquatic systems.

4.3.2. Salt

Use of salt as a “natural” and “total (i.e. not selective)” herbicide against weeds is traditionally used not only in coastal areas, but also in gardens. Experiments on the sensitivity of *L. grandiflora* and *M. aquaticum* to salt showed that it could be a potential method to reduce their populations (Thouvenot et al., 2012). Brackish water is used against invasive macrophytes, reducing invasive macrophyte populations in ditches, for example *Ludwigia* spp. (Chesneau et al., 2015). Furthermore, salt is applied against IAAPs by submersion with sea water (Dandelot et al., 2004) or by locally administering brine (Pierre, 2014, on *Ludwigia*) or by aerial spray, as used for the control of *E. crassipes* (Veith et al., 2008).

4.4. Other indirect control methods

4.4.1. Shading by trees

The likelihood of eradication of the IAAP is improbable when shading by trees is the only method used. However, reduction of invasive plant cover by riparian trees is possible (Bunn et al., 1998), and planting trees on banks has been investigated (Dawson and Kern-Hansen, 1979). Many invasive macrophytes, such as *Ludwigia* spp., are heliophilous (Thouvenot et al., 2013), and experiments to reduce their cover by planting willows on banks and in marshes are presently being conducted (J. Haury, pers. observation). In general terms, although many authors list river or stream shading as a means to reduce or minimize undesirable macrophyte growth, the level of shading required to achieve a desired reduction in weed biomass or cover (which is species dependent based on their compensation point, and related to light attenuation through the water column for submerged plants) is not well defined (Matheson et al., 2012), and managers must consider the time needed from planting (i.e. of shade producing trees) to IAAP control by shading.

4.5. Unsuccessful management methods

A number of alternative (non-operational) techniques have been used, especially regarding the management of amphibious species, with no success. Examples include the use of hot foam in an attempt to eradicate *C. helmsii* in nature reserves in South England (Bridge, 2005); the use of hydrogen peroxide and flame-throwers on *C. helmsii* and *H. ranunculoides* which was also ineffective as only emergent parts were treated and submerged material remained unaffected (Dawson and Henville, 1991; Leach and Dawson, 2000; Invexo, 2013); and the use of liquid nitrogen, which yielded similarly poor results although it was effective on small areas (Leach and Dawson, 2000). For obvious reasons these alternative methods offer no solutions against aquatic invasive plants.

5. Management costs

The cost of management practices varies between the different weed control methods, the target weed species and its abundance, and the weed management goal (De Winton et al., 2013) both in the short and longer term.

Aquatic weed management costs $100 million (USD) per year in the USA (Pimentel et al., 2000). In Florida, the management of *H. verticillata* costs of approx. $14.5 million (USD) per year (Pimentel et al., 2000). In contrast, the *H. verticillata* eradication programme in California costs about $2.5 million annually and has been successful since 1976 in restricting populations to a few sites that are being eradicated. California was the first state to initiate this approach and the success is evident in the absence of this highly invasive species in nearly all water bodies including the massive Sacramento-San Joaquin Delta which supplies water to over 25 million people each year for agriculture, potable and industrial uses.

In Europe, the management costs of free-floating *E. crassipes* are up to €4 million per infested site per year using mechanical harvesting of 50,000 tons of plant material (EPPO, 2009). For the control of *H. ranunculoides* the Dutch water boards spent €1 million in 2000 only (EPPO, 2006), and the costs can increase when plants have accumulated high contents of heavy metals which preclude simple biomass disposing or composting (Pot, 2003). The annual costs for management of *H. ranunculoides* alone amount to €1-3 million in the Netherlands, United Kingdom and Flanders (EPPO, 2006). Management costs for control of *L. grandiflora* and *L. peploides* in France amount to € 2,000-40,000/ha (EPPO, 2011). Cost for a single mechanical treatment of a *C. caroliniana* infestation in the Netherlands using a harvester amounted to €350,000 (Matthews et al., 2013). By comparison, in New Zealand, the cost of mechanical weed clearance is approximately 5 times more expensive than the use of aquatic herbicides.

Costs of the Hydro-Venturi system, when taking into account all preparatory work and aftercare, can be in the order of € 1.35–2.05 m^{-2}. This depends on dimensions of the waterways and environmental conditions such as sediment type (van Valkenburg, 2011). In complex large water systems, where extensive monitoring and barrier systems are required, costs may rise up €5 m^{-2} (Dorenbosch and Bergsma, 2014). Overall, management of IAAPs is costly and varies with the invasive species and management method, with costs increasing as plant biomass increases. But the costs of IAAP management are still low, considering the overwhelming benefits and value of freshwater for life in general.

6. Effects of management on the ecosystems

Management aimed at the reduction or eradication of IAAPs will often not only impact the invasive plants, but also other species in the ecosystem. Non-target species can suffer directly, when control methods are not species specific, or indirectly due to changes in habitat or food availability for example. The impact on other (native) species present in the ecosystem increased with increasing extent of the IAAP infestation that needs to be controlled, but also the management affects the impact on native species. The effect that a management method will have, can be influenced by the timing of management (Richardson, 2008), the specific technique used and characteristics of the ecosystem (e.g. morphology, trophic status, species present; De Winton et al., 2013). As an example, much research has been undertaken on herbicide use profiles (i.e. increasing selectivity and minimising herbicide volumes whilst still achieving efficacy), some of these improvements in use are also based on considerations of seasonal timing.

Although most management techniques have some negative side effects on the system, severe infestations of IAAPs can also have a negative impact on the ecosystem, when not addressed at an early stage of invasion, including impacts to critical habitats for threatened and endangered species. These are all important considerations to take into account when choosing the appropriate management method or combination of methods for a specific IAAP problem.

Herbicides (including salt) can, for instance, kill (physiologically similar) native aquatic plants with negative effects on water clarity and potentially also harm fish and invertebrates although in an indirect manner (for registered products) (Hummel and Kiviat, 2004; Valley et al., 2006; Richardson, 2008). Extensive product registration toxicity tests preclude direct impacts to animals when product label directions are followed (which is the legal requirement in countries where herbicides are permitted). With a registered herbicide indirect impacts could result from the decomposition of treated plant material with locally reduced water quality and associated problems for fauna (Richardson, 2008). However product label conditions and/or local permissions (as in New Zealand and USA for example) mitigate the potential for such impacts by defining portions of the target weed beds, relative to the size of the waterbody that can be treated at any one time. Similarly there is the potential for off-target impacts when herbivores are used as biocontrol agents both directly (e.g. grass carp consumption of non-target plants, including rare native aquatic species if present) and indirectly (e.g. altered foodwebs with CBC agents). Grass carp grazing can result in eradication of the entire submerged vegetation, thereby severely impacting the ecosystem (Richardson, 2008). Furthermore, the foodweb is also inherently altered when introducing additional organisms to the ecosystem. Not only lower trophic levels (plants) can potentially be affected by a biological control agent (including CBC agents), but also higher trophic levels could be affected, for instance predators. In addition, large scale mechanical management is not species specific. Mechanical mowers are often perceived as environmentally neutral, however they can remove and kill aquatic fauna associated with weed beds and cause sediment resuspension (Unmuth et al., 1998; Madsen, 2000; Habib and Yousuf, 2014).

An assessment of environmental effects is considered an essential component of any management programme in response to an IAAP incursion; along with the knowledge of an achievable desirable outcome compared to the consequences of no management programme.

7. Restoration of native vegetation after successful management of IAAPs

From an ecological perspective, the ideal invasive species management programme should focus on maintaining or restoring the native macrophyte community. It is accepted that resident species may reduce the establishment success of individual invaders through competition and other biotic interactions (̈biotic resistancer̈eviewed in Levine et al., 2004) or at least lower the negative ecological impacts of these invaders. The mechanism behind the biotic resistance hypothesis is that species rich communities can use the resources more effectively, which are then not available for the new species (in this case IAAPs). Disturbance and environmental change can lead to empty niches which may lead to resources becoming available for alien species. Preventive management thus ideally starts with the restoration of already affected ecosystems and minimising the disturbance of systems that are in a pristine state. It is also important to prevent the occurrence of empty niches during management practices or to fill them as soon as possible (Van Kleef and Leuven, 2012). Restoration ecologists can achieve this by stimulating (facilitating or planting) native communities that will fill a large part of the available niche space and so best resist invasion. By artificial planting, the establishment of native macrophyte communities can be supported especially in those cases, where seeds banks of native species are lacking (Hilt et al., 2006). However, the role of the propagule bank in restoring native vegetation is largely unknown in some regions, like in Europe (Bakker et al., 2013). In contrast, experience in New Zealand has documented that submerged seed banks possess attributes of longevity and offer a potential means to restore vegetation resources in degraded lakes (De Winton et al., 2000). Charophytes, milfoils and pondweeds have all been recorded amongst the flora that recolonises following weed control actions (e.g., herbicide, grass carp) (Tanner et al., 1990; Rowe and Champion, 1994; Kelly et al., 2012; Hofstra et al., 2014). However, studies have also established a link between the decline of native seed banks and invasion by submerged IAAPs (De Winton and Clayton, 1996) and the consequence of seed burial in reducing seedling (oospore germling) emergence (Dugdale et al., 2001), highlighting the need for timely management intervention.

8. Weed disposal subsidies

This section refers to the use of harvested macrophytes for example in composting, vermiculture, stockfood or for the generation of biogas energy (see e.g. Bates and Hentges, 1976; Edwards, 1980) to potentially offset harvesting costs and reduce landfill disposal. Of particular relevance for the first three uses is that the end product may be utilized in the generation of food for humans. This requires that the material does not contain contaminants (e.g. trace metal elements) present at concentrations that will enter and/or bio-accumulate within the food chain to unsafe levels. Prior to the use of harvested IAAPs biomass, consideration must be given to the composition of the plant matter, and in particular the presence of heavy metals (Pot, 2003). Composting (i.e. recovering nutrients) and the use of harvested plant material in biogas generators are increasingly documented options (Quilliam et al., 2015; Zehnsdorf et al., 2015) along with the potential to isolate and extract high valuable products (like β-sitosterol in *E. nuttallii*, Munoz Escobar et al. (2011)).

9. Conclusion

Although a variety of methods have been developed for the control of invasive aquatic plants, there is no single method that is suitable for every situation. While there is a wealth of experience of IAAP management in both the USA and NZ spanning decades, there is by comparison little experience of successful biomass reduction and/or eradication of IAAPs in Europe, even for species that were introduced more than 50 years ago (e.g. for *M. heterophyllum* and *Elodea* spp.). The success of any management effort largely depends on the leadership of the agency responsible for taking action, their ability to sustain the required funding, the development of an appropriate management plan, and the experience and commitment of the operators. A decision scheme for the management of IAAPs is presented (Fig. 2) to illustrate relationships between native and invaded condition and how the timing of appropriate management (such as early detection) can provide greater surety of attaining an eradication goal (EU legislation) while the native condition is capable of regeneration with less intervention (and associated costs). Preventing the introduction of IAAPs is the most cost effective way to safeguard ecosystems from the negative impacts of invasive species. Yet although trading bans and codes of conduct provide valuable tools to reduce the likelihood of IAAP introduction (Section 3.1), comprehensive monitoring programs are required for early detection and identification of IAAPs (Section 3.2) to provide the best opportunity for eradication. Any delay in the management of IAAPs increases the management costs and decreases the likelihood of eradication, as IAAPs spread quickly and may produce overwintering organs or other long-lived reproductive structures. Expert knowledge on the biology of the targeted IAAPs (Section 3.3) and appropriate management method(s) under the given habitat conditions is required to define the management goal (Section 3.5) and the best management practice (Section 4) and the restoration potential of the site (Section 6).

However while the ability to respond rapidly to a new IAAP incursion is a challenge because the development of an appropriate management plan is site specific (i.e. considers the size, type, biota and function of the waterbody), rapid response could be facilitated by the preparation of a management options assessment (MOA) for each of the listed IAAPs, which includes the biology and ecology of the IAAP and the available options for population control and/or eradication. A detailed MOA for each species would then provide the foundation from which a site-specific management plan could be readily developed in response to an incursion. In most cases, rapid response and an integrated management approach (combination of different methods) provides the best opportunity for eradication at an early stage of invasion. However, expert knowledge and information on species ecology, matched with the appropriate control tools is mandatory for the successful and sustainable management of IAAPs. Consequently, we recommend that the sharing of expert knowledge on IAAP biology and management approaches needs to be encouraged and implemented on a more regular basis, to improve the quality and predictability of outcomes from the management of IAAPs.

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