Invasive Plants: Impacts and Management in Rivers and Catchments

S.S. Schooler[*], D.G. Williams[**], K. Stokes[***] and M. Julien[*]

[*] CSIRO Entomology, Long Pocket Laboratories, Indooroopilly QLD, Australia, 4068 (E-mail: shon.schooler@csiro.au; mic.julien@csiro.au)
[**] CRC for Freshwater Ecology, University of Canberra ACT 2601 (E-mail: williams@aerg.canberra.edu.au)
[***] CSIRO Entomology, Black Mountain Laboratories, Canberra ACT 2601 (E-mail: kate.stokes@csiro.au)

Abstract

Invasive plants have major implications for river management on account of their diverse social, economic and environmental impacts and the high cost of control. Traditional physical and chemical control methods are problematic in these systems because physical disturbance often increases the spread of aquatic weeds and herbicide use is restricted. Biological control has proven safe and effective for the management of many invasive riparian and aquatic plants. However, with the increasing numbers of invasive plants, many habitats are being over-run by multiple invaders simultaneously. Therefore, we need to combine single species control methods with strategies for managing ecosystems to increase resistance to invasive plants. This requires a greater understanding of how invasive species spread and dominate plant communities, particularly with regard to dynamic river systems. Using a variety of examples we evaluate options for control of invasive riparian and aquatic weeds and relate these to catchment planning and management at the landscape level. We conclude that in order to reduce the associated economic, social and environmental costs it is important to incorporate invasive plant management when designing catchment and river management strategies.

Keywords

Alien plants; aquatic weeds; introduced plants; landscape management; wetlands

INTRODUCTION

There is a burgeoning terminology that is associated with biological invasions. Introduced plant species are often labelled as; nonindigenous, exotic, alien, invasive, naturalised, weeds, or noxious weeds. Confusion in definitions of key vocabulary has led to many unproductive debates in ecology, particularly in quickly expanding fields such as invasion ecology (Colautti and MacIsaac, 2004). To avoid confusion and promote constructive discussion we begin by presenting definitions of key terms. Here we use "invasive species" to denote a species that has been introduced from a different area (native range), becomes established, and spreads without intentional assistance by humans in its new habitat (introduced range). In contrast, an "invasive weed" is an invasive species that has a negative effect on people and/or the environment (Fig. 1).

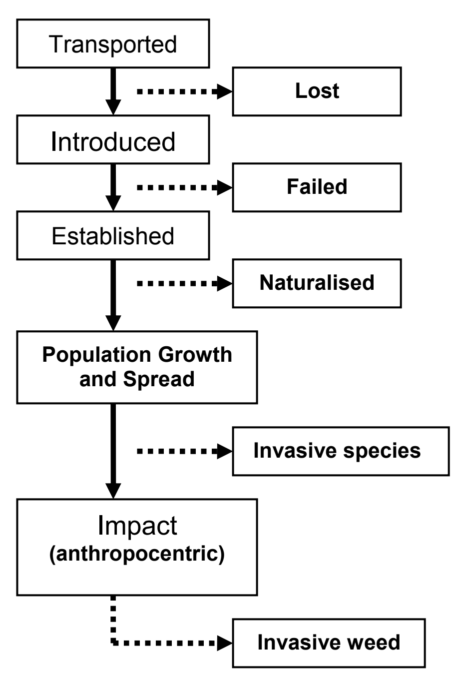


Figure 1. Steps in the process of invasion leading to impacts. Invasive plants may or may not have discernable impacts.

Effects of invasive species on people and the environment

Most introduced plants have no obvious negative impacts. In fact, we rely on many introduced plant species for food and medicine. However, there are some plant species whose populations expand very quickly and are cause for concern. Over time, as more plants are introduced, more become invasive, and more inevitably become problem weeds (Williamson, 1996). In the past we have been particularly concerned regarding plants that directly affect public safety, agricultural productivity, and livestock health. However, we have recently come to realize how dependent we are on the quality of our natural environment, both for recreation and for the services we derive from intact ecosystems. Because of our ever-growing need for fresh water, weeds that impact the abundance and quality of water are of paramount importance.

Invasive plants affect us directly through social and economic impacts, and indirectly through alteration of biotic communities and ecosystem functions (Table 1). These species can directly affect agricultural and grazing productivity by displacing desirable species, reducing soil quality (via increased erosion), and reducing water abundance (via evapotranspiration and impeding irrigation systems) (Mooney, 2005). In addition, invasive aquatic weeds can indirectly affect ecosystem services by altering biotic diversity. From an ecological perspective, we can categorize impacts of invasive plants by the ecological level of organization at which we measure the effect (Parker *et al.,* 1999). Most often we are concerned with effects on populations, communities, and ecosystems, but genetic impacts and effects on individuals may be important as well. Invasive aquatic plants can also directly impact our quality of life through loss of recreational activities. For many, quality of life is also negatively affected by the loss of native species and alteration of biotic community composition.

Table 1. Impacts of invasive aquatic and riparian plants

- Economic Impacts
    - reducing agricultural productivity (displacing species and impeding irrigation)
    - reducing grazing productivity (displacing forage species and poisoning livestock)
    - disrupting ecosystem services
    - reducing land values
    - impeding transportation
- Social Impacts
    - decreasing value of public amenities (tourism, recreation)
    - negatively affecting public health and safety (swimming and disease vectors)
- Environmental Impacts
    - Ecosystem functions
        - reducing water quality
        - altering nutrient cycling and increasing nutrient runoff
        - increasing soil erosion
        - changing river morphology
        - altering water flow
        - decreasing water retention
    - Biotic Populations
        - causing local or regional species extinctions
    - Biotic Communities
        - decreasing plant diversity and productivity
        - decreasing animal diversity and abundance

Challenges in controlling invasive aquatic and riparian weeds

We have many tools for managing weed populations including; physical removal, chemical application, biological control, cultural practices, and integrated methods. Some weeds are effectively controlled through mechanical damage or herbicide application. However, most invasive weeds are difficult to control using any single method and integrating multiple methods may be necessary to achieve the desired level of control (Paynter and Flanagan, 2004). Populations of invasive aquatic weeds are particularly challenging to manage because; (1) aquatic plants often spread by asexual reproduction (plant fragments) and mechanically damaging the plant will not kill the plant, and may even increase its rate of spread, (2) the use of herbicides is greatly restricted in aquatic habitats, particularly near potable water supplies, and many aquatic weeds are resistant to herbicides.

Biological control is often the last recourse for management of weeds that are causing large negative impacts and can't be controlled through other methods (Myers and Bazely, 2003). The ecological theory behind the mechanism of weed biological control is that when species move into new environments they are often introduced without the specialized herbivores that are present in their native range (Keane and Crawley, 2002). The absence of damage from these herbivores gives the introduced plant a competitive advantage over native plants that are being damaged by their complement of native herbivores. This competitive advantage allows the introduced plants to form dense stands and displace native plants. Biological control re-introduces these natural enemies to their host plants in the new range. A biological control program consists of eight basic steps: (1) officially identifying a weed as a biological control target, (2) determining the native range of the weed, (3) surveying in the native range for potential biological control agents, (4) studying the ecology of the potential agents and prioritizing based on host specificity and probability of successful control, (5) receiving permission to import the selected agents into quarantine in the introduced range, (6) conducting host specificity tests on native and economically important plants, (7) receiving permission to release the agents, (8) releasing and monitoring the agents (Julien and White, 1997). Biological control programs are initially expensive, but are often the only effective means of providing safe and sustained control of aquatic weed infestations (see Walton, 2005 for a history of weed biological control in Queensland).

EXAMPLES OF INVASIVE AQUATIC WEEDS

Cabomba

Cabomba (*Cabomba caroliniana*), or water fanwort, is a fast-growing submerged aquatic weed that has the potential to spread throughout the aquatic habitats of Australia (Mackey and Swarbrick, 1997; Ensby, 2000). It is also considered a problem weed in the United States, Canada, Greece, Japan, and China. It grows well in slow-moving water bodies, particularly where nutrient concentrations are high. Cabomba prefers areas of permanent standing water less than 3 m in depth, however it can grow at depths to 5 m (Schooler, unpublished data). The weed is easily recognised by its opposing pairs of finely dissected underwater leaves that are feathery or fan-like in appearance. The small white flowers extend above the water's surface, making weed infestations more visible during the summer months. Reproduction is almost entirely vegetative and any small fragments that include the leaf nodes can grow into a new plant.

Cabomba originates from South America (Orgaard, 1991). The plant's tolerance of fragmentation and ease of cultivation make it a desirable aquarium plant and consequently it was brought into Australia through the aquarium trade (Mackey and Swarbrick, 1997). Cabomba was subsequently introduced into lakes and streams both accidentally, through the dumping of aquarium water, and on purpose, to enable cultivation for later collection and sale. Currently, cabomba is primarily found in rivers and dams of coastal Queensland and New South Wales. However, isolated populations occur from Darwin to Melbourne (Mackey and Swarbrick, 1997). It is easily spread across drainages on water craft, boat trailers and perhaps by waterfowl. Cabomba is a declared weed throughout Australia and it is illegal to propagate, move, or sell this noxious plant. It is listed as one of 20 weeds of national significance (WoNS) in Australia.

Cabomba negatively effects the environment, recreational activities, public safety, and water quality (Mackey and Swarbrick, 1997). The weed can smother native submerged plants such as pondweeds (*Potamogeton spp.*), stoneworts *(Chara spp.*), hornwort (*Ceratophyllum demersum*), and water nymph (*Najas tenuifolia*). Cabomba may also reduce germination of desirable native emergent plants. Alteration of the flora is thought to have reduced populations of platypus and water rats in northern Queensland (Mackey and Swarbrick, 1997). In southern Queensland, cabomba appears to negatively effect populations of the endangered Mary River cod (T. Anderson, pers. comm). The long stems of cabomba impede the movement of boats and can get tangled in propellers, paddles, and fishing lines. This makes many recreational activities less desirable in areas infested with cabomba and thereby reduces tourism. In addition, cabomba is a potential danger to swimmers who may become entangled in the long stems. Cabomba also decreases water quality for human consumption by tainting and discolouring potable water supplies. It interferes with dam machinery, such as valves, pumps, and aerators, with leads to increased costs of maintenance.

Currently, there is little that can be done to control cabomba once it is established (Anderson and Diatloff, 1999). Herbicides are largely ineffective and herbicides use is restricted in or around public water supplies. Some managers are using floating mechanical harvesters to remove cabomba, but these machines are expensive to purchase and operate and are restricted to areas of deep water and wide channels. In addition, they only remove the tops of the plants and the remaining stems soon grow back to the surface. It is likely that the only method that will be effective in managing cabomba is biological control. In 2003, CSIRO Entomology began a project to discover and test biological control agents from cabomba's native range in an effort to find a long-term sustainable solution to this problem.

Water hyacinth

Water hyacinth (*Eichhornia crassipes*) is a free floating plant from Amazonia, South America. It produces sprays of showy mauve flowers and consequently it has been spread around the world by man as an ornamental plant. Water hyacinth reproduces both sexually and vegetatively. Flowering can occur within 10 to 15 weeks after germination (Barrett, 1980). Each inflorescence can produce more than 3,000 seeds which, when released sink and sit in the substrate until water levels wane, which stimulates germination. Seed can remain viable for 17 to 20 years (Barrett, 1980). Each plant also produces off-shoots that form daughter plants. So, in the absence of fluctuating water levels, plant populations continue to expand vegetatively to quickly cover the water's surface.

The blanketing of the surface of water with extensive mats of water hyacinth seriously changes the ecology, values and use of the waterways. Other plants and animals are eliminated or restricted in growth or behaviour, habitats for disease vectors are increased, water quality is decreased, and transportation and recreation (such as boating and fishing) is prevented. The weed causes blockages to waterways which contributes to flooding and damages infrastructure when the heavy mats lodge against fences or bridges. In Papua New Guinea, infestations of water hyacinth in the middle and lower Sepik River and associated lagoons and channels (the highways of that wetland area) threatened the social structure of village populations that were dependent on the river for practically all aspects of living. People died because they could not access medical help, schools and markets could not be attended, and malnutrition and disease increased due to reduced quality of water. The weed mats restricted access to gardens, reduced fishing (the staple protein source) and increased the incidence of disease associated with snail and mosquito vectors. In some instances entire villages were abandoned. In Africa, water hyacinth restricted access to fisheries and severely reduced the incomes of many communities (eg. around Lake Victoria). International transportation of people and goods was reduced when weed mats prevented the docking of vessels at Kisumu in Kenya and Entebbe in Uganda. Water and power were regularly disrupted when turbine intakes became clogged.

Herbicides can be used to destroy mats of water hyacinth and machinery can be employed to remove the floating biomass. However, these methods are not sustainable over time. Simple calculations are essential before employing such activities because it comes down to resources and fortitude. Can the method employed remove the biomass faster than it can grow? If so, can the method be maintained for the duration necessary to remove the entire biomass? If so, can this or another method be employed to stop reinvasion and regrowth? History indicates that, except in small accessible areas or critical locations (such as around water intakes), it is largely a waste of resources attempting to control water hyacinth using herbicidal and mechanical methods.

Biological control was first developed for water hyacinth by the US Department of Agriculture (USDA). Surveys conducted by the USDA in South America identified a range of insects and pathogens with potential for biological control. The first releases were made in Zambia in 1971 and in the USA in 1972. To date, a fungus, a mite and five insects have been released in various countries (Julien and Griffiths, 1998). The most important agents are the two weevils *Neochetina bruchi* and *N. eichhorniae*. These two insects have been released and provide significant control of the weed in numerous countries (Julien *et al.,* 1999; Julien, 2001; Center *et al.,* 2002). The adults of these weevils feed on the leaf blade and petioles, making characteristic scars. Their larvae feed by tunnelling into the lower petiole and rhizome.

This damage severely reduces growth and reproduction and allows the crown of the plant to become waterlogged. The combination of weevil damage and the secondary fungal infections usually result in the water hyacinth mats breaking up and sinking. The process occurs more quickly when temperatures are optimal year round and the host plant is of higher nutritive quality (associated with eutrophic waters) (Room *et al.,* 1986).

Salvinia

Salvinia (*Salvinia molesta*) is a free floating fern from south-eastern Brazil (Forno and Harley, 1979). Sporocarps form amongst the roots but this species does not develop fertile spores and therefore population increase is solely due to vegetative growth. The plant is dispersed when fragments are transported to a new location (Room, 1983, 1990). Similar to water hyacinth, it is considered an attractive water garden and aquaria plant. Consequently it also has been spread around the world through the horticultural trade and is currently a problem in many tropical and sub tropical countries. It thrives in high nutrient waters and in tropical climates. Often floods flush large quantities of the weed downstream or out to seas where it is killed by saline conditions. Within months the few remaining plants grow and multiply to once-again cover the water's surface. Under ideal conditions the biomass of salvinia can double within days (Mitchell and Tur, 1975; Room, 1986).

When salvinia grows unchecked it forms a blanket of vegetation over the water's surface. Further growth causes the weed to become multi-layered and some mats in Papua New Guinea were over a metre thick (Thomas and Room, 1986). Multi-layered mats of salvinia require significant flood events to remove them due to the large biomass (Storrs and Julien, 1996). The movement of such mats can destroy infrastructure, i.e., bridges and fences, or cause blockages that increase flooding in a similar manner as water hyacinth.

Herbicides are available that will kill the weed or reduce its growth for a period (Diatloff *et al.,* 1979; Storrs and Julien, 1996). In addition, the weed can be readily scooped up by machines where it is accessible. The major issue with these methods is that it is impossible to remove all material except from small areas. Regrowth is so fast that repeated applications are required ad infinitum – a costly on-going process. In tropical climates growth may be so fast that it is neither practical nor affordable to treat or remove the weed faster than it can grow (Farrell, 1978)

In the 1970's, surveys in south-eastern Brazil found a small weevil called *Cyrtobagous salviniae*, a natural enemy of the plant. Host specificity studies found that this weevil was specific to salvinia (i.e. it would not attack other plant species) (Forno *et al.,* 1983) and it was subsequently released in Australia in 1980 (Room *et al.,* 1984). Early releases were made on Lake Moondara, Mt Isa. Within 18 months the few hundred weevils that were released had multiplied to a population of millions, which quickly destroyed the dense salvinia mats (Room *et al.,* 1981). Over subsequent years this weevil was distributed, reared and released at many locations in many countries and achieved similar levels of control of salvinia (Julien *et al.,* 2002). In recognition of the extra-ordinary success achieved in Sri Lanka the salvinia team was awarded the UNESCO Science Prize in 1985. In that project the return on investment was calculated to be 53:1 (Doeleman, 1989).

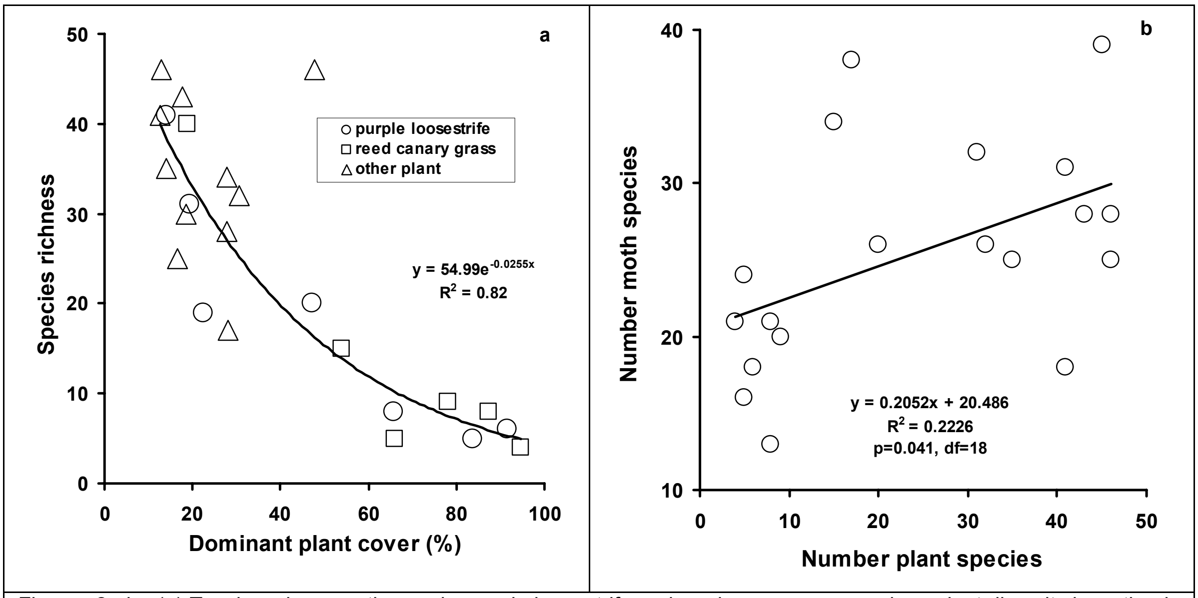
Purple loosestrife

Purple loosestrife (*Lythrum salicaria*) is a wetland plant native to Europe. It is a tall (2-3 m) emergent aquatic plant that prefers shallow standing water with high nutrient concentrations. It probably arrived on the East coast of the United States before 1830 in ballast deposited by trading ships from Northern Europe (Thompson, 1991). Loosestrife was initially described as a native species. However, it was recognized as an aggressive invasive species during the 1930's in wetland pastures along the St. Lawrence River (Thompson *et al.,* 1987). It has subsequently spread across North America aided by road construction and irrigation channels, as well as through the planting of seeds sold in wildflower mixes

(Wilcox, 1989). It is currently considered a noxious weed across temperate North America (Blossey *et al.,* 2001). Two thorough reviews of the biology, ecology, and history of invasion in North America of purple loosestrife are provided by Thompson et al. (1987) and Mal et al. (1992). Purple loosestrife is present in Australia where its status as a native species is currently under dispute.

Management of purple loosestrife has proven very difficult. It is a tall perennial plant that can tolerate standing water due to production of spongy tissue around its stem (aerenchyma) that facilitates oxygen transfer to roots. A single adult plant can produce in excess of 1 million seeds and any plant fragment can produce a new plant through adventitious rooting (Mal *et al.,* 1992). Herbicides might destroy adult plants, but these are soon replaced by seedlings from the prodigious seed bank. Physical removal rarely removes enough material to kill the plant entirely and often increases plant spread when fragments are transported for disposal.

Purple loosestrife is an invasive aquatic weed that displaces native wetland vegetation in wetlands and riparian areas (Blossey, 2001; Schooler, 2003; Landis *et al.,* 2003). Studies have found that as the abundance of loosestrife increases, both plant diversity and associated herbivore diversity decrease exponentially (Fig. 2a and b, Schooler 2003). However, biological control agents were released in 1992 and they have been very effective in reducing loosestrife populations across North America (Schooler, 1998; Landis *et al.,* 2003). Plant community diversity has increased following the reduction of loosestrife abundance in the midwestern USA (Landis *et al.,* 2003). However, a study in the northwestern USA found that after the biological control agents reduced loosestrife populations by 90%, another invasive wetland weed, reed canary grass (*Phalaris arundinacea*: Poaceae), increased in abundance. The result was no observable increase in biotic diversity (Schooler, 1998).



Figures 2a-b. (a) Two invasive aquatic weeds, purple loosestrife and reed canary grass, reduce plant diversity in wetlands of the northwestern United States. Data were ln transformed prior to linear regression analysis ($P < 0.001$). (b) Herbivore (moth) species richness is positively correlated with the number of plant species. This indicates that herbivore diversity will decline with plant diversity as the abundance of the invasive plants increase (data from Schooler 2003).

Alligator weed

Alligator weed (*Alternanthera philoxeroides*) is an invasive plant that originates from South America and is currently spreading in many countries throughout the world including the United States, China, India, Thailand, Burma, New Zealand, and Australia (Julien, 1995). It is primarily associated with aquatic habitats, but can spread into moist terrestrial environments. The plant was first recorded in Australia in 1946 (Julien and Bourne, 1988). It invades agricultural areas and blocks drainage and irrigation channels causing problems on agricultural land (Spencer and Coulson, 1976). It has also eliminated the turf industry near Newcastle due to potential spread from contaminated material. Other concerns of alligator weed include water pollution from plant decomposition and an increase in mosquito breeding areas (Spencer and Coulson, 1976). Alligator weed is currently listed as one of 20 weeds of national significance (WoNS) in Australia.

Alligator weed is a perennial plant that does not produce viable seeds in its introduced range (Ensby, 2001) and reproduces and spreads by adventitious rooting, primarily from stem nodes (Ensby, 2001; Julien et al., 1992). Although herbicides destroy leaves and shoots, they do not cause direct mortality of roots (Tucker, 1994). Physical control methods often increase the spread of the plant. The difficulty of controlling the abundance and spread of alligator weed instigated a program to investigate the potential for biological control methods. The alligator weed flea beetle, *Agasicles hygrophila* (Coleoptera: Chrysomelidae), has been successful in controlling the aquatic form of alligator weed in the warm temperate climates of Australia (Julien, 1981). However the beetle has been unsuccessful in controlling the terrestrial form and does not control the weed in cooler temperate climates (Julien et al., 1995; Julien and Bourne, 1988). CSIRO Entomology is currently studying the prospects for safe and effective biological control of alligator weed in terrestrial habitats and cool temperate climates.

Willows

All willows (*Salix* spp.: Salicaceae) in Australia are introduced from overseas, originally for soil stabilization, river erosion control and as shelterbelt plantings to provide shade for stock. Impacts of concern include channel obstruction and diversion resulting in river bank erosion, increased sediment loads, reduction in channel capacity and increased flooding (ARMCANZ 2000). In comparison to the native river red gum, (*Eucalyptus camaldulensis*), which shares a similar niche, *Salix* species shed leaves at different times and rates, provide different levels of river shade and experience different litter breakdown rates (Schulze & Walker 1997), resulting in changes in abundance, diversity and composition of terrestrial and aquatic invertebrates (Pidgeon and Cairns, 1981; Greenwood *et al.,* 2004), with potential consequences for associated riparian fauna.

The success of *Salix* species in southern Australia is largely attributable to the versatility of their reproductive systems. One of the striking features of the Salicaceae is their capacity to regenerate either by seed or vegetatively under a wide range of environmental conditions (Karrenberg *et al.,* 2002). Natural forms of vegetative reproduction include both the production of stolons (clonal growth) and the rooting of detached branches. The latter is particularly common in *S. fragilis*, a willow with very brittle branches which is now one of the most common willows in Australia. For plants capable of both vegetative reproduction and reproduction by seed, populations which experience limited seedling recruitment and low disturbance are expected to show reductions in density over time due to density-dependent mortality (Watkinson and Powell, 1993). We investigated whether this would occur in populations of *S. nigra* at Blowering Dam, near Tumut in NSW, Australia. The population at Blowering Dam largely consisted of young individuals, most probably established via wind blown seed. Mortality was high in the younger, smaller age classes and decreased substantially once an individual exceeded 35 mm in stem diameter. At a low disturbance site such as Blowering Dam it is predicted that, in the short term, dominance by larger and larger clones will occur at this site, leading to the eventual longer term replacement of *Salix* species by taller stemmed more shade tolerant species.

This has implications for future management regimes. If seed sources can be eliminated, non-interventionist methods would result in the eventual replacement of willows by other tree species. However, this decline may take many decades and other management methods might be necessary depending on the goals of the program.

MODELLING THE SPREAD OF INVASIVE SPECIES

Once invasive species arrive and become established, their management requires not only the targeting of population growth through available control measures but also the assessment of where and how populations are spreading in the landscape. For new invaders there is often a lack of the precise data required to predict population growth and expansion. These data constraints, along with the need for managers to consider both population size and spatial distributions, suggest that simulation modelling has a key role in evaluating management scenarios for invasive species. This is particularly relevant to freshwater aquatic species because flow is a major driver for the dispersal and population growth of such organisms. An initial study, outlined here, has incorporated many of the requirements for modelling the spread of aquatic invasive species, particularly in relation to the issues that may concern water resource managers. These include the relative effects of floods and their timing and frequency and the size of founder populations.

The study area we simulated was a linear waterway of eight sections. The model was run multiple times to examine the effects of varying four parameters; (a) initial population size introduced into the most upstream section of the model system (50, 500, 5000 plants), (b) flood frequency (no floods, one flood every 5 or 10 years), (c) year in which the first flood occurred (year 1, 5, 6 or 10) and (d) the effect of a floodplain being inundated and connected to the main river channel.

The initial population size and flood frequency had major effects on population sizes downriver (Fig. 3a). As floods were set to increase the reproductive rate by a factor of ten, they had the largest effect on population growth and hence the number of individuals dispersing. However, the size of the initial population, which varied by two orders of magnitude, also directly translated to larger populations and faster spread, with a diminishing effect as the carrying capacity of a section was approached (Fig. 3a). In the lowest section, the carrying capacity was approximately 1.2 million and the highest population achieved after 50 years was about 0.9 million. Population size in turn controlled the time to reach the lowest section downriver (Fig. 3b). In the absence of floods, the model allowed only minor dispersal downstream each year. However initial population sizes were still important, since at the two smallest initial population sizes, the lowest river section was not even colonised within the 50 yr time span of the model; whereas with 5000 starters there was a small number arriving downriver at 35 years (Fig. 3b). The potential shortest time to spread downriver was 8 years and at the highest flood frequency and initial population size the time taken was just 13 years (Fig. 3b).

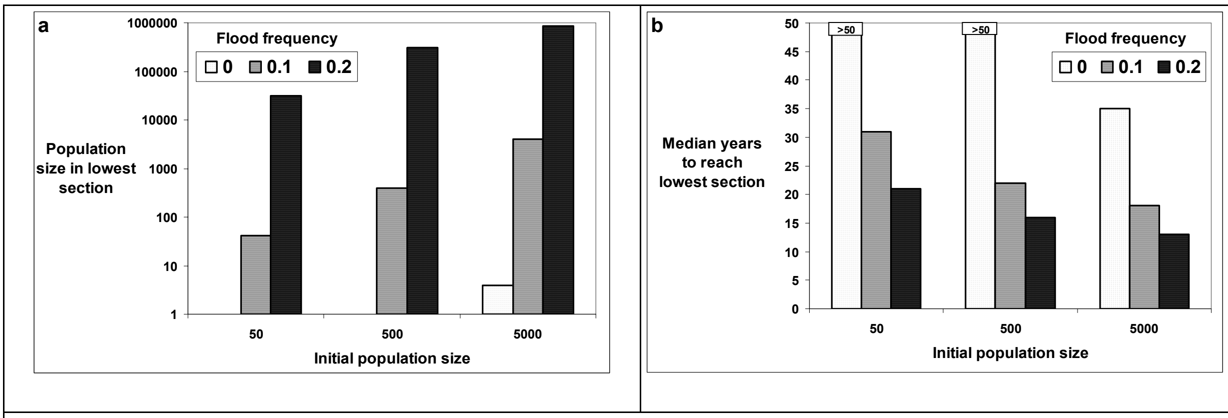


Figure 3. (a) Presence of floods and higher frequency of floods greatly increased population sizes in the lowest reach after 50 years. Initial population size had a lesser effect. (b) The time taken to populate the whole reach is also strongly dependent on presence of floods and their frequency. Without floods, and at the two smallest initial population sizes, the lowest reach was not colonised within 50 years (which was the simulated time frame).

Propagule pressure has been repeatedly identified as a key element for invasion success (Kolar and Lodge, 2001; Lockwood *et al.,* 2005). Propagule pressure consists of both the numbers of individuals initially introduced into a system and the frequency with which they arrive at new sites (Williamson and Fitter, 1996). These two components were separated in our simulations allowing a comparison of their effects on invasion after 50 years. Our simulations were consistent with the observations that increased propagule pressure was a key element in invasion success. However, we found that vector intensity, as reflected in flood frequency, has a dramatically greater effect on final population densities than the initial number of invaders. In studies on environmental flows rather than floods, it has also been concluded that high flow events may be of benefit to invasive species, resulting in increased spread rates and abundances (Howell and Benson, 2000). Given the aim of increasing river health through environmental flows, invasive species issues need to be considered in water management reforms.

CONCLUSIONS

Invasive aquatic weeds can negatively affect our economy, environment, and quality of life. Physical damage and herbicide application are often not effective in managing invasive aquatic weeds. Biological control has proven to be a safe and effective management method. However, biological methods can only control species individually because the safety of biological control programs relies upon releasing only host specific agents. In some cases removing one weed will just result in another taking its place. Ecological research allows us to search for and evaluate alternative means of controlling invasive species. Some of these may be effective in controlling multiple weed species simultaneously, or may identify management practices that reduce the invasibility of ecosystems. On the other hand, some management practices may exacerbate the establishment, spread, and impact of invasive species. Therefore, it is essential that the management of invasive aquatic plants should be included in river and catchment management strategies.

ACKNOWLEDGEMENTS

Thanks to the RiverSymposium Committee for organizing this event. Linda Merrin, Charles Todd and Sabine Schreiber of the CRC for Freshwater Ecology contributed to the modelling project. Thanks also to Rieks van Klinken and Alice Yeates for their constructive reviews of this manuscript.

REFERENCES

Agriculture and Resource Management Council of Australia & New Zealand (ARMCANZ) (2000). *Weeds of national significance: willow* (Salix *taxa*, *excluding* S. babylonica, S. × calodendron *and* S. × reichardtii) strategic plan. National Weeds Strategy Executive Committee, Launceston, Tasmania, Australia.

Anderson, T. and Diatloff, G. (1999). Cabomba management attempts in Queensland. In: *Practical weed management: protecting agriculture and the environment.* 10th Biennial Noxious Weeds Conference, Ballina, Australia, 20th-22nd July 1999.

Barrett, S.C.H. (1980). Sexual reproduction in *Eichhornia crassipes* (Mart.) Solms (water hyacinth). II. Seed production in natural populations. Journal of Applied Ecology **17**, 113-124.

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

Blossey, B., Skinner, L.C. and Taylor, J. (2001). Impact and management of purple loosestrife (*Lythrum salicaria*) in North America. Biodiversity and Conservation, **10**, 1787-1807.

Center, T.D., Hill, M.P., Cordo, H. and Julien, M.H. (2002). Waterhyacinth. In: *Biological Control of Invasive Plants in the Eastern United States*, R. Van Driesche, B. Blossey, M. Hoddle, S. Lyon and R. Reardon, (eds.), USDA Forest Service, pp. 41-64.

Colautti, R.I. & MacIsaac, H.J. (2004). A neutral terminology to define “invasive” species. Diversity and Distributions, **10**, 135-141.

Diatloff, G., Lee, A.N. and Anderson, T.M. (1979). A new approach for *Salvinia* control. Journal of Aquatic Plant Management, **17**, 24-27.

Doeleman, J.A. (1989). Biological control of *Salvinia molesta* in Sri Lanka: An assessment of costs and benefits. ACIAR Technical Report 12, Canberra.

Ensbey, R. (2000). Cabomba: identification and management of cabomba species. *in* Agnote - NSW Agriculture.

Ensbey, R. (2001). Alligator weed. *in* AGFACTS - NSW Agriculture.

Farrell, T.P. (1978). The spread and control of *Salvinia molesta* in Lake Moondara, Mt Isa, Queensland. In, Proceedings of the First Conference of the Council of Australian Weed Science Societies, April 1978, Burwood, Victoria, 179-188.

Forno, I.W. and Harley, K.L.S. (1979). The occurrence of *Salvinia molesta* in Brazil. Aquatic Botany, **6**, 279-286.

Forno, I.W., Sands, D.P. and Sexton, W. (1983). Distribution, biology and host specificity of *Cyrtobagous singularis* Hustrache (Coleoptera: Curculionidae) for the biological control of *Salvinia molesta*. Bulletin of Entomological Research, **73**, 85-95.

Greenwood, H., O'Dowd, D.J. and Lake, P.S. (2004). Willow (*Salix* x *rubens*) invasion of the riparian zone in south-eastern Australia: reduced abundance and altered composition of terrestrial arthropods. Diversity and Distributions, **10**, 485-492.

Howell, J. and Benson, D. (2000). Predicting potential impacts of environmental flows on weedy riparian vegetation of the Hawkesbury-Nepean River south-eastern Australia. Austral Ecology, **25**, 463-475.

Julien, M.H. (1981). Control of aquatic *Alternanthera philoxeroides* in Australia, another success for *Agasicles hygrophila*. In: *Proceedings of the 5th International Symposium on Biological Control of Weeds*, E.S. Delfosse (ed.) CSIRO, Melbourne, pp. 583-588.

Julien, M.H. (2001). Biological control of water hyacinth with arthropods: a review to 2000. In: *Proceedings of the second meeting of the IOBC Global Working for the Biological and Integrated Control of Water Hyacinth*, M.H. Julien, M.P. Hill, T.D. Center and D. Jianqing (eds.), Beijing, China, ACIAR Proceedings No. 102, pp. 8–20.

Julien, M.H. and Bourne, A.S. (1988). Alligator weed is spreading in Australia. Plant Protection Quarterly, **3**, 91-96.

Julien, M.H., Bourne, A.S., and Low, V.H.K. (1992). Growth of the weed *Alternanthera philoxeroides* (Martius) Grisebach, (alligator weed) in aquatic and terrestrial habitats in Australia. Plant Protection Quarterly, **7**, 102-108.

Julien, M.H., Center, T.D. and Tipping, P.W. (2002). Floating Fern (Salvinia). In: *Biological Control of Invasive Plants in the Eastern United States*, R. Van Driesche, B. Blossey, M. Hoddle, S. Lyon and R. Reardon, (eds.), USDA Forest Service, pp. 17-32.

Julien, M.H. and Griffiths, M.W. (eds) (1998). *Biological Control of Weeds. A World Catalogue of Agents and Their Target Weeds, 4th edition*. CABI Publishing, Wallingford, United Kingdom.

Julien, M.H., Griffiths, M.W. and Wright, A.D. (1999). *Biological Control of Water Hyacinth*. The weevils *Neochetina bruchi* and *N. eichhorniae*: biologies, host ranges, and rearing, releasing and monitoring techniques for biological control of *Eichhornia crassipes*. ACIAR Monograph No. 60, 87 p.

Julien, M.H., Skarratt, B. and Maywald, G.F. (1995). Potential geographical distribution of alligator weed and its biological control by *Agasicles hygrophila*. Journal of Aquatic Plant Management, **33**, 55-60.

Julien, M.H. and White, G. (1997). *Biological Control of Weeds: Theory and practical application*. ACIAR Monograph No. 49, 192pp.

Karrenberg, S., Edwards, P.J. and Kollmann, J. (2002). The life history of Salicaceae living in the active zone of floodplains. Freshwater Biology, **47**, 733-748.

Keane, R.M. and Crawley, M.J. (2002). Exotic plant invasions and the enemy release hypothesis. Trends in Ecology and Evolution, **17**, 164-170.

Kolar, C.S. and Lodge, D.M. (2001). Progress in invasion biology: predicting invaders. Trends in Ecology and Evolution, **16**, 199-204.

Landis, D.A., Sebolt, D.C., Haas, M.J. and Klepinger, M. (2003). Establishment and impact of *Galerucella calmariensis* L. (Coleoptera: Chrysomelidae) on *Lythrum salicaria* L. and associated plant communities in Michigan. Biological Control **28**, 78-91.

Lockwood, J.L., Cassey, P. and Blackburn, T. (2005). The role of propagule pressure in explaining species invasions. Trends in Ecology and Evolution, **20**, 223-228.

Mackey, A.P. and Swarbrick, J.T. (1997). The biology of Australian weeds 32. *Cabomba caroliniana* Gray. Plant Protection Quarterly, **12**, 154-165.

Mahoney, J.M. and Rood, S.B. (1998). Streamflow requirements for cottonwood seedling recruitment; an integrative model. Wetlands, **18**, 634-645.

Mal, T.K., Lovett-Doust, K.J., Lovett-Doust, L. and Mulligan, G.A. (1992). The biology of Canadian weeds. 100. *Lythrum salicaria*. Canadian Journal of Plant Science, **72**, 1305-1330.

Mitchell, D.S. and Tur, N.M. (1975). The rate of growth of *Salvinia molesta* in laboratory and natural conditions. Journal of Applied Ecology, **12**, 213–225.

Mooney, H.A. (2005). Invasive alien species: The nature of the problem. In: *Invasive Alien Species*, H.A. Mooney, R.N. Mack, J.A. McNeely, L.E. Neville, P.J. Sheia, and J.K. Waage (eds.), Island Press, Washington, DC. pp. 1-15.

Myers, J. and Bazely, D. (2003). *Ecology and Control of Introduced Plants*. Cambridge University Press, Cambridge.

Orgaard, M. (1991). The Genus Cabomba (Cabombaceae) - A Taxonomic Study. Nordic Journal of Botany, **11**, 179-203.

Parker, I. M., D. Simberloff, D., Lonsdale, W.M., Goodell, K., Wonham, M., Kareiva, P.M., Williamson, M.H. Von Holle, B., Moyle, P.B., Byers, J.E. and Goldwasser, L. (1999). Impact: Toward a Framework for Understanding the Ecological Effects of Invaders. Biological Invasions, **1**, 3-19.

Pidgeon, R.W.J. and Cairns, S.C. (1981). Decomposition and colonisation by invertebrates of native and exotic leaf material in a small stream in New England (Australia). Hydrobiologia, **77**, 113-127.

Paynter, Q. and Flanagan, G.J. (2004). Integrating herbicide and mechanical control treatments with fire and biological control to manage an invasive wetland shrub, *Mimosa pigra*. Journal of Applied Ecology, **41**, 615-629.

Room, P.M. (1990). Ecology of a simple plant-herbivoure system: Biological control of *Salvinia.* Trends in Ecology and Evolution, **5**, 74-79.

Room, P.M. (1986). Equations relating to the growth and uptake of nitrogen by *Salvinia molesta* to temperature and the availability of nitrogen. Aquatic Botany, **24**, 43-59.

Room, P.M. (1983). Falling-apart as a lifestyle – the rhizome architecture and population growth of *Salvinia molesta*. Journal of Ecology, **17**, 349-365.

Room, P.M., Forno, I.W. and Taylor, M.J. (1984). Establishment in Australia of two insects for biological control of the floating weed *Salvinia molesta* Bulletin of Entomological Research, **74**, 505-516.

Room, P.M., Harley, K.L.S., Forno, I.W. and Sands, D.P.A. (1981). Successful biological control of the floating weed salvinia. Nature, London, **294**, 78-80.

Schooler, S.S. (1998). *Biological Control of Purple Loosestrife, Lythrum salicaria, By Two Chrysomelid Beetles, Galerucella pusilla and G. calmariensis.* MSc Thesis, Oregon State University, Corvallis, OR.

Schooler, S.S. (2003). *Negative Effect of Purple Loosestrife and Reed Canary Grass on the Diversity of Wetland Plant and Moth Communities.* PhD Thesis, Oregon State University, Corvallis, OR.

Schulze, D.J. and Walker, K.F. (1997). Riparian eucalypts and willows and their significance for aquatic invertebrates in the River Murray, South Australia. Regulated Rivers: Research and Management, **13**, 557-577.

Spencer, N.R. and Coulson, J.R. (1976). The biological control of alligator weed, *Alternanthera philoxeroides*, in the United States of America. Aquatic Botany, **2**, 177-190.

Storrs, M.J. and Julien, M.H. (1996). *A handbook for the integrated control of Salvinia molesta in Kakadu National Park.* Northern Landscapes Occasional Papers No. 1. Australian Nature Conservation Agency, Darwin, Australia.

Thomas, P.A. and Room, P.M. (1986). Taxonomy and control of *Salvinia molesta*. Nature, **320**, 581-584.

Thompson, D.Q. (1991). History of purple loosestrife (*Lythrum salicaria* L.) biological control efforts. Natural Areas Journal, **11**, 148-150.

Thompson, D.Q., Stuckey, R.L., and Thompson, E.B. (1987). Spread, impact and control of purple loosestrife (*Lythrum salicaria*) in North American wetlands. U.S. Fish and Wildlife Research, Washington DC.

Tucker, T.A., Langeland, K.A., and Corbin, F.T. (1994). Absorption and translocation of 14C-imazapyr and 14C-glyphosate in alligator weed, *Alternanthera philoxeroides*. Weed Technology, **8**, 32-36.

Watkinson, A.R. and Powell, J.C. (1993). Seedling recruitment and the maintenance of clonal diversity in plant populations: a computer simulation of *Ranunculus repens*. Journal of Ecology, **81**, 707-717.

Walton, C. (2005). *Reclaiming Lost provinces: A century of weed biological control in Queensland*. Department of Natural Resources and Mines, Brisbane, QLD, 104 pp.

Wilcox, D.A. (1989). Migration and control of purple loosestrife (*Lythrum salicaria* L.) along highway corridors. Environmental Management, **13**, 365-370.

Williamson, M. (1996). *Biological Invasions*. Chapman and Hall, London, England.

Williamson, M.H., and Fitter, A. (1996). The characters of successful invaders. Biological Conservation, **78**, 163-170.