Chapter 26
Biological Control of Invasive Plants in Protected Areas

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Abstract Classical weed biological control is widely used in natural areas. It is based on introduction of specialised natural enemies (herbivorous insects and fungal pathogens) from the weed's native range. It can be used safely if specialised natural enemies are selected and can be highly effective in suppressing weeds over large areas. Agents used in modern projects typically have genus or species level specificity and are safe when proper risk analysis and procedures are followed. Agents spread over large areas and can move into hard-to-reach areas. If correctly selected, agents are safe for use in areas too ecologically sensitive for chemical or mechanical control. Costs are independent of area to be treated because agents are self-reproducing, and results are self-sustaining. Biological control is most appropriate for use against widespread weeds, difficult to control with other methods that occur in critical habitats and damage biodiversity or ecosystem function. Finding suitable agents is easier against weeds distantly related to local native plants. Such targets reduce risk to native flora, facilitate agent screening, lower cost, and increase likelihood of success. Projects should be partnerships between biological control scientists and conservation biologists, and biological control activities should be done within a comprehensive restoration plan for the ecosystem. In some cases, suppression of the invasive weed may be sufficient, but sometimes additional actions, such as replanting native species or modifying ecosystem processes such as fire or flooding regimes may be essential.

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26.1 What Is Biological Control?

Biological is a tool that can be used in some cases to assist in the ecological restoration of areas affected by high density, damaging populations of invasive plants, reducing such plants to densities that pose less of a burden on native biodiversity and allowing ecosystems to avoid invasion-driven physical transformations (Van Driesche et al. 2008). This result is achieved by harnessing the power of herbivory or fungal infections to lower the fitness of targeted exotic plants, allowing the intrinsic competitive power of native plants to be more effective. If landscape-scale reduction of the target plant's density is acceptable (as in the case of weeds with no economic value), projects seek to lower plant fitness directly, resulting in smaller infestations, slower spread, and reduced weed biomass. All components of the target plant – seeds, foliage, stems, etc. – can then be attacked. In contrast, if plants have important social or economic uses (such as some introduced forestry species in South Africa), then the project's goals must be limited to the suppression of reproduction, selecting agents that attack only flowers or seeds. This can help limit the plant's spread from economic use sites into wild lands and can support manual clearance of existing stands in natural areas by minimizing regeneration. Weed biological control is an ecological restoration tactic whose risks are generally low (Pemberton 2000) and whose use is often effective. In South Africa, for example, 19 of 23 (83 %) projects were completely or partially successful (Hoffmann 1996; Clrruttwell McFadyen 1998) and in Hawaii 10 of 21 (50 %) targeted weeds were completely or partially suppressed (Markin et al. 1992; Gardner et al. 1995). Here we discuss the contribution of biological control of invasive plants to protection of natural areas, including legally protected preserves, and suggest steps for strategic integration of weed biological control into restoration ecology.

While the focus of this book is on controlling weeds in legally protected areas (parks, preserves), biological projects operate on a much wider scale. Unlike locally applied measures (chemical and mechanical control or replanting of native species), which are typically done inside preserves, biological control is applied to whole landscapes. As such, it is the case that preserves fall inside areas affected by biological control rather than the reverse. For example, control of *Euphorbia esula* (leafy spurge) in the northern prairie of North America was an areawide reduction of the pest over millions of ha, including a number of preserves as, for example, the Pine Butte Swamp Preserve of The Nature Conservancy in Montana (USA). Similarly, biological control of *Melaleuca quinquenervia* (melaleuca) over several hundred thousand ha of southern Florida (USA) overlapped with such preserves as the Everglades National Park (a World Heritage site) and the Big Cypress National Preserve. In the Northern Territories of Australia, biological control of *Mimosa pigra* (mimosa) certainly included areas within Kakadu National Park, while biological control of *Opunti*a spp. cacti and *Hakea sericea* (hakea) in South Africa affected Kruger National Park and various Protected Areas of the Cape Floral Region, respectively. Many other instances of overlap between legally protected areas and regions where biological control has reduced damaging invasive plants could be identified.

26.2 Advantages and Disadvantages of Biological Control Versus Other Methods

Invasive plants in natural areas may be locally suppressed by hand weeding, mechanical control (cutting, dredging, mechanical clearing of brush), and application of herbicides. Size of the area over which the weed must be controlled determines the practicality of these methods. Preserve managers may intend to suppress a weed in only a specific, often small, area (the preserve), but these patches are frequently part of a landscape-wide infestation. Weed reduction at the landscape level requires biological control, but doing so requires long-term commitment of resources and the expertise of specialised scientists. Consequently, biological control is not the first choice for weed control on a single preserve. In part this is because such projects cannot be initiated or carried out at the preserve level since they act over the whole landscape and require governmental approval, special skills, and years of effort before rewards are produced. The advantages of biological control, however, are especially important in cases where invasive plants are widespread and control is desired over large areas. Because biological control uses self-perpetuating living organisms, control spreads on its own after effective agents have been identified and established, until they reach their ecological limits. These features make biological control the only control method that is economically feasible for suppression of invasive plants over very large areas (millions of ha). Also, the method is free of both the disturbances characteristic of mechanical control and the pollution that may follow the widespread use of herbicides.

Disadvantages of biological control from the perspective of preserve stewards include the fact that the method is beyond their direct control to initiate against new target plants. Stewards can, however, participate in regional projects, releasing useful agents on their property after effective agents become available. The participation of The Nature Conservancy (TNC) in the control of *Euphorbia esula* on some prairie preserves in North America is an example of such participation. *Aphthona* beetles were introduced to preserves after being studied and proven effective at other sites (Cornett et al. 2006). In some cases, conservation groups may participate at earlier stages, as for example TNC participation in preparation for the release and evaluation of *Aphalara itadori*, a psyllid being studied in the United States and the U.K. (Shaw et al. 2009) for the control of *Fallopia japonica* (Japanese knotweed) along rivers (Gerber et al. 2008).

The most important disadvantage of biological control agents stems from their permanency. Once released, inappropriately selected agents can rarely be removed, although potentially they might themselves be amenable to suppression via biological control using insect parasitoids (e.g. Pemberton and Cordo 2001). The use of biological control against invasive plants requires a high level of certainty about the safety and desirability of each new herbivore before it is released. The track record of weed biological control insects released in the United States (including Hawaii) and the Caribbean since the 1960s provides strong evidence that modern agent selection processes provide this necessary level of certainly (Pemberton 2000). Finally, the ecological limits to spread for each newly released biological control agent must be predicted so that non-target flora in all potentially invaded regions can be considered. However, human-assisted accidental spread of agents may occasionally move agents to distant areas (Pratt and Center 2012), far beyond the area targeted for biological control. In this respect, biological control agents are no different than any other species, all being potentially subject to such chance events.

26.3 When Is Biological Control the Right Approach?

Biological control projects should not be under taken lightly as they require a long-term commitment of funds and scientific manpower to carry through to completion. Premature commitment to a project against a minor pest may consume resources better used against a more serious invader. Appropriate targets should be invasive non-native plants that are widespread (or potentially so) or are intransigent to other control methods in critical habitats and cause (or potentially cause) significant damage, usually to natural areas. Several factors further modify both feasibility and cost of projects. Securing agents with adequately narrow host ranges is more likely when targeted invasive plants are only distantly related to native plants. Targeting such species lowers the risk to native flora, expedites agent screening, is often less costly, and is more likely to succeed. *Melaleuca* and *Tamarix* spp. (saltcedar) in North America are both in subfamilies or tribes with few or no representatives in the native flora (saltcedar: subfamily, Baum 1967; Crins 1989, and melaleuca: tribe, Serbesoff-King 2003). In contrast, projects in North America directed against invasive thistles (*Carduus*, *Cirsium*, and *Silybum*) or knapweeds (*Centaurea*) are more complicated because there are many congeneric native species (e.g. for thistles; Schroder 1980), some of which may be endangered. While projects against targets with native congeners are not uncommon, screening of more species may be needed to find suitable agents.

While many projects begin with no known prospective agents, some have the advantage of being directed against species that have been controlled elsewhere, making them quicker and cheaper because potentially effective species are known. The principal cost in such cases is the screening of additional species from the flora of the new area. It is important to recognise that safety is contextual and a species safe to introduce in one country may be unsafe in another due to differences in the composition of resident plant communities.

Projects should be directed against plants that cause the most ecological damage to local ecosystems. Species that change the properties of the invaded community, such as increasing fire frequency or intensity (Brooks et al. 2004) or that have a structural form that allows them to overtop native species are likely to be highly damaging. Invasive floating aquatic plants, vines, and ground covers are likely to seriously damage native plant communities. For example, plants such as *Eichhornia crassipes* (water hyacinth) and *Salvinia molesta* (salvinia) severely alter the submersed aquatic communities that they blanket (Mitchell 1978; Thomas and Room 1986).

Availability of funding is also an important consideration in starting a classical biological control programme. Programmes without adequate, long-term funding (often 10 years or longer) and enough political support to see the work through the inevitable setbacks have little chance of success. It is also important to select species as targets for biological control that reflect local conservation priorities.

26.4 Mechanics of Biological Control

Steps and decision points involved in a classical weed biological control project are illustrated in Fig. 26.1. More detailed information on the actual mechanics involved is available in the literature (e.g. Harley and Forno 1992; Van Driesche et al. 2008). The flow chart in Fig. 26.1 is of a generalised nature and does not cover all aspects of a weed biological control project. Selection of a candidate agent often involves more than simple determination of host range. It may be warranted to attempt to predict the potential efficacy of the agent; however, how biological control agents actually perform in nature depends on a complex set of population processes that can't be tested in the laboratory, including the effects of parasitoids, predators, competitors, and climate. This step is often advocated, even though such predictions have never been proven to be possible. Any such predictions should therefore be validated after release to advance the science involved. Also, consideration must be given to the presence of pre-existing agents so that the plant parts attacked, the phenological timing of agent populations, and agent habitat preferences are all mutually complementary. Such considerations become important in projects requiring multiple agents. The order of introduction may also need to be considered, so that earlier introduced agents do not interfere with success of later introductions. Each project is unique so rules of thumb are often meaningless and projects must be adaptively managed. The overall purpose of this process is to balance the risk of introducing a novel herbivore or plant pathogen into the system against the risk imposed by the plant invader (i.e. the risk of doing nothing).

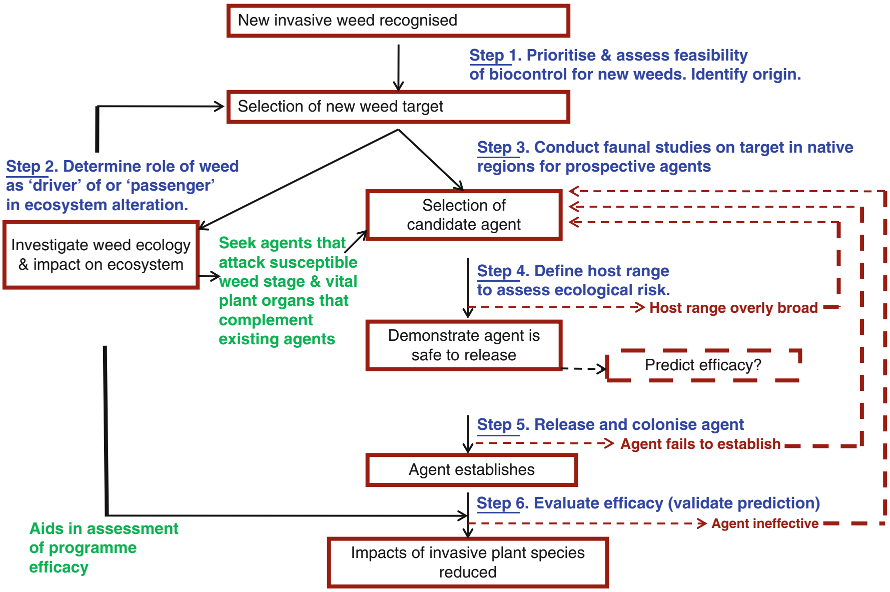

Fig. 26.1 The steps and decision points involved in biological control of an invasive plant species

26.5 Fitting Biological Control into a Holistic Approach to Ecological Restoration

Biological control of invasive plants in natural areas (including legally protected preserves) should be a partnership between biological control scientists and conservation biologists, and the biological control activities should be part of a larger, holistic programme for the restoration of the affected ecosystems or protected areas.

26.5.1 Partnerships and Goal Setting

Partnerships for such biological control efforts are critical because of the complicated nature of the problems being addressed, requiring the active participation of land managers and ecologists who are familiar with the systems in need of restoration. Skills needed for large scale ecological restoration projects are commonly spread over several agencies or universities, with distinct budgets and somewhat different perspectives on the problem. It is important to overcome this separation and form a team willing to work together to bring into play the full range of ideas and skills needed. Such partnerships also help avoid conflicts that might arise when actions are taken by single parties before all concerned have reached an accord.

Defining restoration goals and the role of biological control and other activities is a critical first step to be taken by the partners in a project. While biological control aims to lower the density of what is believed to be an invader damaging to the natural area, the goal per se is to restore the natural community, either in terms of biodiversity or ecological function. For example, *Lygodium microphyllum* (Old World climbing fern) has changed the fire cycle of invaded *Taxodium distichum* (cypress) forests by increasing fire frequency and intensity. Therefore, lowering these fire characteristics should be a goal of the project. Additionally, if certain native species have been displaced from a plant community, their return to pre-invasion levels is also a goal. Such native plants, however, may not recover spontaneously following the suppression of the responsible invasive species because propagules of desired plants may be lacking or other invasive weeds may increase quickly following biological control. For example, at some sites where the weevil *Rhinoncomimus latipes* reduced infestations of *Persicaria perfoliata* (mile-a-minute weed), the invasive plants *Microstegium vimineum* (Japanese stiltgrass) and *Rosa multiflora* (multiflora rose) increased in abundance (Lake 2011; Hough-Goldstein et al. 2012). In that case, planting native perennials, along with use of a pre-emergent herbicide to suppress the other invasive species, was used to restore such communities (Lake 2011). Similarly, Stephens et al. (2009) found that only non-native grasses showed consistent increases following reductions of *Centaurea diffusa* (diffuse knapweed) caused by the weevil *Larinus minuta*. Additional measures were, therefore, necessary to restore that community.

26.5.2 Determining Causality of Community Degradation

Invasive plants may be fundamentally responsible for ecosystem degradation or just symptoms of other processes. This has been described with the simile of 'drivers and passengers.' Also, if invasive plants are drivers, it is necessary to determine if they act alone or are facilitated by other factors. Because there is a long time lag between starting a biological control project and release of effective agents, surveys for prospective agents should start as soon as there is agreement that the targeted invasive plant is the driver of the observed habitat degradation. In some cases, biological control may not be advisable if the invader is merely a passenger, responding to some other disturbance (e.g. nutrification of an aquatic system stimulating growth of aquatic weeds).

Determining that an invasive plant is merely a passenger requires careful observation and knowledge of the community. For example, in the 1950s, several species of *Opuntia* cacti were seen as pests in pastures, including the two native species *O. stricta* and *O. triacantha* and one introduced species, *O. cochenillifera* (Pemberton and Liu 2007) on the Caribbean islands of Nevis and St. Kitts. However, these high density cactus stands were largely the consequence of pasture overgrazing, opening land to cactus invasion and reducing competition from other plants. However, the ability of the moth *Cactoblastis cactorum* to suppress *Opuntia* species was well known (Dodd 1940; DeBach 1974), and so it was released on Nevis, Montserrat, and Antigua (Simmonds and Bennett 1966), ignoring risks to native cacti. Given that the fundamental reason for damaging cacti levels was overgrazing by goats (Simmonds and Bennett 1966), the correct response would have been not biological control, but rather better livestock management. Another example in which the difference between a driver and passenger is not clear is that of *Alliaria petiolata* (garlic mustard), a European mustard that forms dense stands in deciduous forests of the north-eastern and north central United States, which are associated with low diversity of native forest herbaceous plants (Blossey et al. 2002). Based on those facts, biological control of *A. petiolata* seemed necessary to protect forest wildflower diversity. However, some have linked invasive earthworms (Maerz et al. 2009) and overgrazing by deer (Knight et al. 2009) as more fundamental drivers of change in these forests. However, it should be noted that things can change: species that were not originally drivers may become so once the community has been widely invaded if the invaders change fundamental aspects of the community. So, it is necessary to keep an open mind and continue to observe invaded systems in such cases.

That some invasive plants are drivers of change is well know from impacts in other locations and the invasion of new areas by such species should be seen as a cause for alarm and suggest the need for immediate plant suppression (Reichard and Hamilton 1997). Additionally, studies in the invaded area may reveal that invasive plants are drivers, having highly damaging effects either by suppressing native plants, e.g. *Miconia calvescens* (miconia) in Tahitian forests (Meyer and Florence 1996; Medeiros et al. 1997; Meyer 1998), or by changing a fundamental characteristic defining and creating a community, such loss of water depth due to soil accretion caused by *Melaleuca quinquenervia* in the Florida Everglades (Center et al. 2012). Such plants clearly merit being targeted for biological control.

Synergy among invaders is also possible in systems suffering from multiple invasions, such as the Florida Everglades (Simberloff and von Holle 1999). The ecological damage from a single invader, and hence the need for its biological control, may change due to its interactions with other invaders, requiring a community-based view to correctly assess risk and need for control. For example, a non-native plant group, such as the figs (*Ficus*), which require specific exotic pollinators, may persist at innocuous levels in the introduced range for decades but then rapidly become invasive after their pollinators invade (Nadel et al. 1992). Once able to reproduce, figs produce many fruits that are spread by exotic frugivorus birds (Kaufmann et al. 1991). Seed dispersal then greatly facilitates the spread of figs to new areas.

External factors may either enhance an important invader's impacts or make undoing its damage more difficult. For example, *Tamarix* spp. are invasive in riparian areas in deserts of the south-western United States. However, the invasion was strongly facilitated by altered river management, dammed rivers being more favourable for *Tamarix* and less favourable for reproduction of native cottonwoods and willows, thus reducing native competition to *Tamarix*. While biological control for the suppression of the species is necessary to restore invaded riparian communities (DeLoach et al. 2003), it may not be sufficient in some areas if native vegetation propagules are absent or conditions are unfavourable for their growth. In such cases, actions such as replanting of native species or resumption of natural flooding may be required (Shafroth et al. 2005). Similarly, eutrophication of waterways is well known to exacerbate infestation of invasive aquatic plants (Coetzee and Hill 2012).

26.5.3 Integrating Activities and Dealing with Complications

In some cases, biological control may need to be combined with other control tactics to suppress an invader. Also, invader suppression itself may need to be combined with other efforts to restore habitat conditions favourable for native species. If an invader is detected early, biological control may not be the right approach as it may be possible to simply eradicate small invader populations by chemical or mechanical means. However, if the infestation is spreading rapidly or is already widespread, biological control is likely to be needed. Biological, chemical, and mechanical controls, or replanting of native species are likely to be implemented by different restoration partners due to differences in expertise. In such cases, careful joint planning of the timing, placement, and degree of all such activities is critical to prevent delays or conflicts. In restoration projects, unforeseen complications are common and the restoration plan must adapt to new developments are they occur. Adaptive management relies on monitoring of the system as it changes in responses to control efforts, so that tactics and goals can be changed as needed if new insights are gained into how the system is currently functioning. While each case is different, complications (or indeed "surprises") are commonplace and should be anticipated, at least in general terms.

While releases of particular individual natural enemy species cannot be "undone" (and therefore must be carefully assessed for safety before release), in the aggregate over many projects and agents, post-release monitoring of outcomes is useful in determining if estimates of safety were well founded and, if not, monitoring data can be used to identify faulty assumptions or procedures that might need to be changed. More commonly, such monitoring is likely to validate predictions broadly but may detect small deviations from predictions that can be used to increase efficacy of pre-release safety procedures.

Invader replacement is one such complication. Suppression of a dominant invader by biological control agents frees space and resources for other plants. These plants may be native species or, in some cases, other invasive species formerly suppressed by the controlled invader. Invader replacement is particularly common in aquatic systems where anthropogenic nutrification has stimulated plant growth (Coetzee and Hill 2012). Rapid control of a problematic aquatic weed, particularly through the use of herbicides, often leads to an upsurge in the abundance of a different macrophyte or a massive algal bloom (Richard et al. 1984). Although spontaneous declines in invasive aquatic macrophytes are not well understood, nutrient depletion has been identified as one likely factor (Barko et al. 1994), and this relationship should be considered in planning the biological control of such species.

Unanticipated food web effects are another complication that may arise at some stages of a biological control project. When biological control agents establish and become common but do not by themselves lower the density of the pest plant they attack, then the biological control agent may supply a readily available food subsidy for resident predators. This is generally a temporary condition, subsiding as the target is controlled. One potential example is that of seed-head gall flies, *Urophora affinis* and *U. quadrifasciata*, introduced to control *Centaurea stoebe* subsp. *micranthos* (spotted knapweed). These flies became abundant and according to Pearson et al. (2000) failed to control *C. stoebe* subsp. *micranthos* (but see Story et al. 2008). During this period, galled seed-heads became a protein-rich food source for deer mice (*Peromyscus maniculatus*), enabling them to more readily survive winters and reproduce earlier in spring, leading to higher mouse populations in knapweed stands (Ortega et al. 2004). These authors further speculated that more mice would mean higher levels of Sin Nombre hantavirus, a human pathogen (Pearson and Callaway 2006). Although this example may seem cause for alarm, there is no evidence of increased incidence of this virus, and, in terms of the biological control project's goals, gall flies alone were never expected to entirely control the weed. Other agents have combined with them to reduce knapweed infestations (see Corn et al. 2006; Story et al. 2006, 2008). It does, however, serve as a reminder that such effects are possible when a new species is inserted into the trophic structure of a community.

Finally, society's view of the desirability of suppressing the target pest may change erratically over time as new facts emerge. For example, biological control of *Hydrilla verticillata* (hydrilla), a submerged aquatic weed, began in Florida in the 1980s. Two leaf-mining flies and two weevils were released as biological control agents (Center et al. 1997; Grodowitz et al. 1997; Wheeler and Center 2007), with partial suppression of plant density. Subsequently, a new herbicide called fluridone was developed that provided easier control of *H. verticillata* and biological control efforts stalled in the 1990s before the project could be completed. When *H. verticillata* became resistant to fluridone in the early 2000s (Michel et al. 2004), interest in biological control revived briefly, but stalled again when the endangered Florida snail kite (*Rostrhamus sociabilis plumbeus*), which normally feeds on the Florida apple snail (*Pomacea paludosa*), was discovered exploiting an exotic apple snail (*Pomacea insularum*) in mats of *H. verticillata*. However, this favourable view of *H. verticillata* reversed again when it was found that it supported growth of a toxic cyanobacterium (Wilde et al. 2005) that became linked to the deaths of thousands of American coots (*Fulica americana*) (Wilde et al. 2005) and possibly some bald eagles (*Haliaeetus leucocephalus*) and other predators via bioaccumulation of the toxin in food webs (Birrenkott et al. 2004; Fischer et al. 2006). These impacts on wildlife stimulated interest in the release of herbivorous fish to control *H. verticillata* in the south-eastern United States, but concerns arose that these fish might be sensitive to the toxin or might transfer it through the food web to piscivorous species (Wilde et al. 2005). Finally, to complicate matters further, in some locations manatees (*Trichechus manatus*) have been shown to benefit from *H. verticillata* infestations (Evans et al. 2008). While biological control of *H. verticillata* may or may not be resumed in the future, these events illustrate the rapidity with which societal views of the desirability of suppression of an invasive plant can change.

26.6 The Historical Record of Weed Biocontrol in Natural Areas

Biological weed control has targeted invasive species in many natural or semi-natural systems (Van Driesche et al. 2010) (Table 26.1) and these efforts have benefits to legally protected preserves within the affected regions or landscapes. Here we review many of these projects, to provide a sense of the magnitude of the benefits of biological control to ecosystem restoration. Projects are arranged by habitat to give an integrated sense of biological control's value in particular systems.

26.6.1 Fynbos Invaders

The South African fynbos supports 8,700 plant species, 68 % of which are endemic (Richardson et al. 1997; Holmes et al. 2000). Its infertile soils are readily invaded by nitrogen-fixing plants that raise soil fertility and depress native plant growth (Lamb and Klaussner 1988; Stock et al. 1995; Yelenik et al. 2004). Fynbos habitats have been invaded by various introduced woody plants in such genera as *Acacia*, *Pinus*, *Hakea*, and *Sesbania*. All of these but the pines have been targeted for biological control, with considerable success. Biological control, often in the form of seed reduction, together with manual clearance, has greatly reduced the threat of several invaders, including (i) *Acacia saligna*, controlled by the fungus, *Uromycladium tepperianum* (Wood and Morris 2007); (ii) *A. longifolia* (Dennill and Donnelly 1991); (iii) *A. pycnantha* (Moran et al. 2005); (iv) *A. cyclops*, which formed impenetrable stands in the lowland fynbos (Richardson et al. 1996) and threatened plant biodiversity in the Cape Peninsula (Higgins et al. 1999), was rendered less invasive by the seed weevil *Melanterius* cf. *servulus* (Impson et al. 2004) and the flower-galling midge *Dasineura dielsi* (Adair 2005; Impson et al. 2008, 2011); (v) *Hakea sericea*, controlled by five introduced insects and the pathogen *Colletotrichum gloeosporioides* in conjunction with manual removal (Gordon 1999; Esler et al. 2010; Gordon and Fourie 2011; Gordon, personal communication, 2012), and (vi) *Sesbania punicea* (Hoffmann and Moran 1998), a leguminous tree that formed dense bands 20–30 m wide along rivers until three

Table 26.1 Plants invasive in natural areas that have been targets of biological control, with notes on habitat/biome invaded, degree of project success, and location

Target species	Habitat/ Biome	Success	Notes/Location	References
Acacia cyclops (rooikrans)	F, Co	C	Agents effective at reducing seed production but integrated control needed to reduce tree densities	Impson et al. (2004, 2008, 2011) and Adair (2005)
Acacia longifolia (long-leaved wattle)	F	C	South Africa, especially the fynbos region: bud galler and seed feeder effectively reduce seed crop	Dennill and Donnelly (1991), Dennill et al. (1999), and Moran et al. (2005)
Acacia nilotica subsp. *indica* (prickly acacia)	GD	IP	Western Queensland, Australia: transforms natural grassland into woody savannah	Palmer et al. (2012)
Acacia pycnantha (golden wattle)	F	C	South Africa, especially the fynbos region: extensive damage from bud galler	Moran et al. (2005) and Impson et al. (2011)
Acacia saligna (Port Jackson willow)	F	C	South Africa, especially the fynbos region: seed beetles destroy to 90 % of seeds; gall-forming rust fungus highly effective	Moran et al. (2005), Wood and Morris (2007), and Impson et al. (2011)
Ageratina adenophora & A. riparia (mistflowers)	WR, O	P	Africa, Asia, Australia, USA, New Zealand, Papua New Guinea, Philippines, Tahiti, Hawaii: invades stream banks and moist areas. Aggressive competitor; controlled in Hawaii, New Zealand, and possibly South Africa by a smut fungus	Heystek et al. (2011), Cruttwell McFadyen (2012), and Schooler et al. (2012a)
Alliaria petiolata (garlic mustard)	FW	IP	Northeast and north central USA: agents under evaluation	Blossey et al. (2001b)
Alternanthera philoxeriodes (alligator weed)	A, WR	C, P	USA, Australia, New Zealand, and China: aquatic infestations controlled by a chrysomelid leaf beetle but not terrestrial populations	Coulson (1977), Julien (1981), Julien and Griffiths (1998), Sainty et al. (1998), Buckingham (2002), and Julien et al. (2012a)
Anredera cordifolia (Madeira vine)	WR, FW	IP	Coastal eastern Australia, South Africa, New Zealand, Sri Lanka, Hawaii: agent surveys underway	van der Westhuizen (2006, 2011), Cagnotti et al. (2007), and Palmer and Senaratne (2012)

Arundo donax (giant reed)	WR, GD	IP	South-western USA, especially Texas: adventive gall wasp found prior to release and armoured scale released	Tracy and DeLoach (1998), Racelis et al. (2009, 2010), and Goolsby et al. (2011)
Asparagus asparagoides (bridal creeper)	FW, Co	P, IP	Coastal areas of temperate Australia: little control to date	Turner et al. (2008b) and Morin and Scott (2012)
Azolla filiculoides (red water fern)	A	C	South Africa: weevil highly effective	McConnachie et al. (2004), Hill and McConnachie (2009), and Coetzee et al. (2011a)
Baccharis halimifolia (groundsel bush)	FW, Co	P	Australia: changes in climate and land use along with biological control have contributed to declines	Palmer and Sims-Chilton (2012)
Cabomba caroliniana (fanwort)	A	IP	Australia: agents under evaluation in quarantine	Schooler et al. (2012b)
Caesalpinia decapetala (Mauritius thorn)	FW	IP	New Zealand, Australia, USA, Kenya, Zimbabwe, South Africa. Transformer species increasing fire risk and causing trees to collapse in subtropical forests	Byrne et al. (2011)
Campuloclinium macrocephalum (pompom weed)	GD	IP	South Africa: disrupting grassland conservation efforts	McConnachie et al. (2011)
Cardiospermum grandiflorum (balloon vine)	Co	IP	Australia, the Cook Islands, Hawaii, New Zealand, South Africa	Simelane et al. (2011)
Centaurea diffusa & *C. stoebe* (diffuse and spotted knapweeds)	GD	IP	Western North America: plant densities not yet reduced at most locations, but declines observed in Oregon and California	Gutierrez et al. (2005), Pitcairn et al. (2005), and Smith (2007)
Cereus jamacaru (queen of the night cactus)	GD	C	South Africa: mealybug effective in most parts of country; cerambycid stem borer very effective when at high density	Paterson et al. (2011)
Cestrum laevigatum (inkberry)	FW, Co	IP	South Africa: forms dense stands in coastal forests and thickets	Fourie (2011)
Cestrum parqui (Chilean inkberry)	WR	IP	South Africa: along the Vaal river in the High Veld	Fourie (2011)

(continued)

Table 26.1 (continued)

Target species	Habitat/ Biome	Success	Notes/Location	References
Chromolaena odorata (Siam weed)	GD, FW	P, IP	Significant control in Papua New Guinea and East Timor of one biotype; some control of second biotype in South Africa	Day and Bofeng (2007), Zachariades et al. (2009, 2011b), Day and Cruttwell McFadyen (2012), and Day, personal communication, 2012
Chrysanthemoides monilifera subsp. *rotundata* (bitou bush), *C. m.* subsp. *monilifera* (boneseed)	Co	P	Bitoubush along coastline of New South Wales, Australia: flowering and seed production widely suppressed. Boneseed in SE Australia. No reductions in density of either subspecies	Holtkamp (2002), Edwards et al. (2009), and Adair et al. (2012)
Cryptostegia grandiflora (rubber vine)	GD, FW	P	Dry tropics of Australia: excellent control achieved where rust fungus established	Evans and Tomley (1994), Mo et al. (2000), Vogler and Lindsay (2002), and Palmer and Vogler (2012)
Cylindropuntia spp. (jumping cholla)	GD	P, IP	Reported as injuring, even causing death of South African wildlife; effective biological control of some species but not others	Paterson et al. (2011) and Holtkamp (2012)
Cytisus scoparius (Scotch broom)	FW	IP	Australia, New Zealand, USA: causes loss of native plant species in Australia. Unsuccessful to date despite long history of effort (>50 years)	Hosking et al. (2012)
Dioscorea bulbifera (air potato)	FW	IP	Florida, USA: first agent released 2011	Pemberton (2009)
Dolichandra unguis-cati (= *Macfadyena unguis-cati*) (cat's claw)	WR, FW, O	IP	Invasive in Australia, South Africa, India, Mauritius, China, Hawaii and Florida in the USA, New Caledonia, St. Helena, and New Zealand	Dhileepan et al. (2007a, b), King et al. (2011), and Dhileepan (2012)
Eichhornia crassipes (water hyacinth)	A	C, P, IP	Southern USA, Mexico, East and West Africa, India, and other warm regions: complete control in many tropical areas; partial control in cooler regions	Beshir and Bennett (1985), Center et al. (2002), Coetzee et al. (2009, 2011a), and Julien (2012a)

Euphorbia esula (leafy spurge)	GD	P	Northern prairies of North America: complete control in many areas	Cornett et al. (2006), Cline et al. (2008), and Samuel et al. (2008)
Euphorbia paralias (sea spurge)	Co	IP	Coastal southern Australia: project in early stages	Scott (2012)
Fallopia japonica (Japanese knotweed)	WR	IP	United States and United Kingdom: first agent approved for release in UK	Shaw et al. (2009)
Genista monspessulana (Cape broom)	GD, FW	IP	Australia, USA: no agents released but one found adventive in Australia. International collaboration on-going	Sheppard and Henry (2012)
Hakea sericea & H. gibbosa (silky & rock hakea)	F	C	South Africa: control achieved in combination with manual clearing	Gordon (1999), Esler et al. (2010), Gordon and Fourie (2011), and Gordon (2012)
Hydrilla verticillata (hydrilla)	A	P, IP	Southern USA: partial control in a few areas	Balciunas et al. (2002), Coetzee et al. (2011a), and Grodowitz, personal communication, 2012
Hypericum perforatum (St. Johnswort)	GD, FW	C, P	Western USA: complete control; Australia: partial control	Huffaker and Kennett (1959), McCaffrey et al. (1995), Briese (1997), and Briese and Cullen (2012)
Jatropha gossypiifolia (bellyache bush)	WR	IP	Dry tropics of Australia: one agent released but not established	Heard et al. (2012)
Lantana camara hybrid complex (lantana)	GD, FW, Co	P, IP	Hawaii, Africa, Asia, Oceania, northern and eastern Australia: limited success in a few areas despite >100 years history	Day and Zalucki (2009), Urban et al. (2011), and Day (2012a)
Leptospermum laevigatum (Australian myrtle)	F	IP	South Africa: negligible control by two biological control agents	Gordon (2011)
Leucaena leucocephala (lead tree)	FW, Co, O	IP	Hawaii, Taiwan, Fiji, Northern Australia, South America, Europe, India, SE Asia, USA: susceptible to psyllid infestations	Austin et al. (1996) and Olckers (2011a)
Lygodium microphyllum (Old World climbing fern)	WR, FW	IP	Southern Florida, USA: control is developing at release sites	Boughton and Pemberton (2009)

(continued)

Table 26.1 (continued)

Target species	Habitat/ Biome	Success	Notes/Location	References
Lythrum salicaria (purple loosestrife)	WR	P	Northern USA and adjacent areas of Canada: control in some areas	Blossey et al. (2001a), Landis et al. (2003), Denoth and Myers (2005), and Grevstad (2006)
Marrubium vulgare (horehound)	GD, FW, O	P, IP	Southern Australia: weed suppression and seed reduction noted. Also invasive in North and South America and New Zealand	Weiss and Sagliocco (2012)
Melaleuca quinquenervia (melaleuca)	WR	C	Southern Florida, USA; control very effective in combination with mechanical and chemical control of mature plants	Pratt et al. (2005), Center et al. (2007), Rayamajhi et al. (2007, 2008, 2009), and Tipping et al. (2008b, 2009)
Miconia calvescens (miconia)	WR, FW, Co	P, IP	Partial control in Tahiti; no control yet in Hawaii	Seixas et al. (2004), Badenes-Perez et al. (2007), and Meyer et al. (2008, 2009)
Mikania micrantha (hemp vine)	WR, FW, Co	P, IP	Australia, Asia, Pacific Islands. A rust reducing infestations in some areas	Day (2012b)
Mimosa pigra (mimosa)	WR	P	Africa, Australia, & Asia. Seed banks reduced by about 90 % in northern Australian wetlands	Heard and Paynter (2009) and Heard (2012)
Moraea spp. (Cape tulips)	GD, FW, O	IP	Australia. Two South Africa species under investigation (*M. flaccida* & *M. miniata*)	Scott and Morin (2012)
Myriophyllum aquaticum (parrot's feather)	A	P	South Africa, control is considered satisfactory although not complete	Coetzee et al. (2011b)
Myriophyllum spicatum (Eurasian watermilfoil)	A	P, IP	United States and Canada: some control attained using a native weevil; South Africa: in progress	Newman (2004) and Coetzee et al. (2011a)
Opuntia spp. (prickly pear cacti)	GD	C	Australia, South Africa. Australia estimates benefit:cost at 147:1	Dodd (1940), Paterson et al. (2011), and Hosking (2012)

Opuntia robusta (wheel cactus)	GD, FW	IP	Australia: little to no control by agents from other *Opuntia* species. Also invasive in the Americas, New Zealand, South Africa, and Mediterranean Europe	Baker (2012)
Paraserianthes lophantha (stinkbean)	F	P	South African fynbos: transformer species, substantial control by seed weevil	Dennill and Donnelly (1991), Dennill et al. (1999) and Impson et al. (2009, 2011)
Parkinsonia aculeata (parkinsonia)	WR, GD	IP	Australia: no population level impacts yet realised	van Klinken and Heard (2012)
Parthenium hysterophorus (parthenium weed)	GD	P	Primarily a pasture weed; Queensland, Australia, control achieved in some areas; Also invasive in many parts of Asia & Africa	Dhileepan and Cruttwell McFadyen (2012)
Pereskia aculeata (Barbados gooseberry)	GD	IP	South Africa: Overtops and kills native flora, sometime collapsing large trees; chyrsomelid beetle shows some promise	Paterson et al. (2011)
Persicaria perfoliata (mile-a-minute weed)	FW	P, IP	Eastern USA: Effective control at some locations	Hough-Goldstein et al. (2009, 2012)
Pistia stratiotes (water lettuce)	A	C, P	Papua New Guinea, Australia; several regions in Africa; and warm parts of North America. Complete control obtained in Queensland and some areas of South Africa	Harley et al. (1990), Dray and Center (1992), Ajuonu and Neuenschwander (2003), Mbati and Neuenschwander (2005), Neuenschwander et al. (2009), Coetzee et al. (2011a), and Day (2012c)
Phyla canescens (lippia)	WR	IP	Australia: Many prospective agents found but none yet released	Julien et al. (2012b)
Prosopis spp. (mesquite)	GD	P	Arid parts of Australia: control achieved in the Pilbara region; South Africa: seed beetles somewhat effective	van Klinken and Campbell (2009), Zachariades et al. (2011a), and van Klinken (2012)
Rubus spp. (blackberries)	FW, O	IP	Chile: reduction in size and competitiveness of plants; Australia, Hawaii, tropical Africa, West Indies, UK	Oehrens (1977), Oehrens and Gonzalez (1977), and Morin and Evans (2012)

(continued)

Table 26.1 (continued)

Target species	Habitat/ Biome	Success	Notes/Location	References
Salsola spp. (tumble weeds)	GD	IP	Western USA, especially California	Smith (2005) and Smith et al. (2009)
Salvinia molesta (giant salvinia)	A	C	Australia, Papua New Guinea, parts of the USA, and parts of Africa, especially the Congo basin. Australia estimates benefit: cost up to 53:1	Room et al. (1981), Thomas and Room (1986), Mbati and Neuenschwander (2005), Diop and Hill (2009), Julien et al. (2009), Coetzee et al. (2011b), and Julien (2012b)
Schinus terebinthifolius (Brazilian peppertree)		IP	Florida, USA: agents under evaluation; Hawaii	Cuda et al. (2009)
Senecio jacobaea (tansy ragwort)	GD	C	Western USA: highly successful in northern California and western Oregan	Pemberton and Turner (1990), McEvoy et al. (1991), Turner and McEvoy (1995), and Coombs et al. (1996, 2004)
Sesbania punicea (sesbania)	F	C	South Africa, especially the fynbos region: three agents maintain plant at non-problematic levels	Hoffmann and Moran (1991, 1998)
Solanum mauritianum (bugwood, tree tobacco)	FW	IP	South Africa: colonises native forest margins overtopping and shading out native species	Olckers (2011b)
Solanum viarum (tropical soda apple)	GD	IP	Southeastern USA: control achieved at release sites, agent spreading	Medal et al. (2008) and Medal and Cuda (2010)
Tamarix ramosissima (saltcedar)	WR, GD	P, IP	Western USA; control developing around release sites	Hudgeons et al. (2007), Carruthers et al. (2008), DeLoach et al. (2008), and Dudley and Bean (2012)
Tecoma stans (yellow bells)	WR	IP	South Africa and neighbouring countries: a 'transformer' species, invades watercourses and rocky sites in tropical/subtropical areas including high-rainfall to semi-arid areas; rust fungus recently released but establishment unconfirmed	Madire et al. (2011)

Triadica sebifera (Chinese tallow tree)	WR	IP	South-eastern USA: agents under evaluation; invades lake and pond margins; displaces native plants; reduces nesting habitat for birds	Wang et al. (2009)
Ulex europaeus (gorse)	WR, FW, Co	P, IP	Chile, Oregon (USA), Tasmania, Hawaii, New Zealand: some impact in Chile, Hawaii and Tasmania	Norambuena (1995), Norambuena and Piper (2000), Davies et al. (2007), Norambuena et al. (2007), Hill et al. (2008), and Ireson and Davies (2012)
Vincetoxicum nigrum & V. rossicum (swallow-worts)	WR, GD, FW	IP	North-eastern USA: surveys for agents in progress	Weed and Casagrande (2010)

Habitat/Biome symbols: *F* fynbos, *A* aquatic, *WR* wetland/riparian, *GD* grassland/desert, *FW* forest/woodlands, *Co* coastal, *O* other
Outcome symbols: *C* complete control, *P* partial control, *IP* in progress

beetles (*Trichapion lativentre*, *Rhyssomatus marginatus*, and *Neodiplogrammus quadrivittatus*) were introduced that destroyed its buds and seeds and bored in its stems (Hoffmann and Moran 1991), reducing its density >95 % (Hoffmann and Moran 1998) and returning rivers to pre-invasion conditions (Hoffmann 2011).

26.6.2 Floating Weeds

In warm regions, floating invasive plants may blanket water surfaces, e.g. *Eichhornia crassipes*, *Salvinia molesta*, *Azolla filiculoides* (red fern), *Pistia stratiotes* (water lettuce) (see Table 26.1), having profound effects on light penetration, changes in nutrients, oxygen, and pH (Toft et al. 2003) and affecting the whole aquatic community. Native benthic plants and associated invertebrates are strongly affected by these changes (Hansen et al. 1971). Biological control has been highly effective in some locations against *A. filiculoides* (Hill and McConnachie 2009; Coetzee et al. 2011a), *E. crassipes* (Center et al. 2002; Wilson et al. 2007; Coetzee et al. 2009, 2011a; Julien 2012a), *S. molesta* (Tipping et al 2008a; Julien et al. 2009), *P. stratiotes* (Neuenschwander et al. 2009; Coetzee et al. 2011a), and *Alternanthera philoxeriodes* (alligator weed; Buckingham 2002). Many water bodies have been relieved of burdening layers of these weeds by biological control and some, like Lake Victoria in East Africa, harbour globally important biota (here, cichlid fishes) (Anonymous 2000; Wilson et al. 2007).

26.6.3 Wetlands Invaders

Wetlands have been invaded by several non-aquatic plants, including the tree *Melaleuca quinquenervia*, the fern *Lygodium microphyllum*, the shrub *Mimosa pigra*, and the herbaceous perennials *Lythrum salicaria* (purple loosestrife) and *Fallopia japonica*. These plants have reduced native biodiversity through habitat change and competition with native plants. All five have been targeted with biological control and at least one (*M. quinquenervia*) has been successfully controlled, while *M. pigra*, *L. salicaria*, and *L. microphyllum* projects have had partial success or are in progress.

Melaleuca quinquenervia formed dense monocultures in Florida, displacing native vegetation (Rayamajhi et al. 2002) and reducing biodiversity of freshwater marshes by 60–80 % (Austin 1978). The weevil *Oxyops vitiosa*, the psyllid *Boreioglycaspis melaleucae*, and the cecidomyiid *Lophodiplosis trifida* suppressed seeding and seedling survival (Center et al. 2007, 2012; Rayamajhi et al. 2007; Tipping et al. 2009), killing 85 % of seedlings, saplings, and suppressed understory trees, leading to a fourfold increase in plant biodiversity (Rayamajhi et al. 2009). This tree is now largely under biological control after an effective integrated control programme in which biological control contributed restraints on seed production, seedling survival, and stump regrowth, while cutting or application of herbicides removed mature trees (Center et al. 2012).

Lygodium microphyllum from Australia smothers trees in Everglades hammocks, cypress swamps, and pine flatwoods in Florida (Pemberton and Ferriter 1998) and increases fire intensity by forming flammable skirts on tree trunks (Pemberton and Ferriter 1998). The pyralid moth *Neomusotima conspurcatalis* has established and now is defoliating the fern at some release sites, allowing regrowth of native plants (Boughton and Pemberton 2009). It has been slow to disperse but is now found several miles from release sites (Center, personal observation).

Mimosa pigra invaded tropical wetlands in Australia, Asia, and Africa, particularly along margins of wetlands, lakes, and channels, but also in open plains and swamps (Cook et al. 1996). In Australia, *M. pigra* converts several vegetation types into homogeneous shrublands with little biodiversity (Braithwaite et al. 1989), threatening vulnerable plant and animal species (Walden et al. 2004). Among two fungi and nine insects established, two species have shown the most impact to date: the sesiid borer *Carmenta mimosa* and the leaf-mining gracillariid *Neurostrota gunniella*, which together have reduced seed set and seedling regeneration, causing *M. pigra* stands to shrink at the edges (Heard and Paynter 2009). Seed banks are now 90 % below pre-biological control levels (Heard 2012).

Lythrum salicaria is a Eurasian perennial that has extensively invaded wetlands in North America, damaging plants, birds, amphibians, and insects (Blossey et al. 2001a; Maerz et al. 2005; Brown et al. 2006; Schooler et al. 2009). The leaf feeding beetles *Galerucella calmariensis* and *Galerucella pusilla*, the root-mining weevil *Hylobius transversovittatus*, and the flower-feeding weevil *Nanophyes marmoratus* were released (Blossey et al. 2001a) and caused defoliation at many sites (Blossey et al. 2001a; Landis et al. 2003; Denoth and Myers 2005; Grevstad 2006). In Michigan, *G. calmariensis* reduced plant height by 61–95 % (Landis et al. 2003) and in many sites where loosestrife has been suppressed, native species have increased (Landis et al. 2003).

26.6.4 Grassland and Desert Invaders

Grasslands and deserts have been invaded by many plant groups, including toxic forbs, woody shrubs, cacti, and grasses (the latter, often introduced for grazing). Toxic forbs have been repeatedly targeted for biological control because of their harm to grazing, e.g. *Centaurea diffusa*, *C. maculosa* and *C. solstitialis* (yellow startistle), *Euphorbia esula*, *Hypericum perforatum* (St. John's wort), *Salsola* spp., and *Senecio jacobaea* (tansy ragwort). These comprise some of the earliest weed biological control projects. Projects against *S. jacobaea* (McEvoy et al. 1991; Turner and McEvoy 1995; Coombs et al. 1996) and *H. perforatum* (Huffaker and Kennett 1959; McCaffrey et al. 1995) are considered complete successes, at least in some countries. In coastal prairies in Oregon, biological control of *S. jacobaea* led to a 40 % increase of the rare hairy-stemmed checkered-mallow (*Sidalcea hirtipes*; Gruber and Whytemare 1997). In natural California grasslands dominated by St. John's wort, biological control allowed native grasses such as *Danthonia californica* (California oatgrass) and *Elymus glaucus* (blue wild rye) to increase (Huffaker and Kennett 1959).

Projects against invasive shrubs in these habitats include ones against *Lantana camara* (lantana), *Prosopis* spp. (mesquite), and *Tamarix* spp. Of these, little has yet been achieved against *L. camara* (Day and Zalucki 2009; Urban et al. 2011), but the project against *Prosopis* spp. has been partially successful (van Klinken and Campbell 2009; Zachariades et al. 2011a) and saltcedar is currently being repeatedly defoliated by introduced chrysomelids in the south-western United States. Vegetative change from reduction of saltcedar, however, has yet to occur (Dudley and Bean 2012).

Invasive cacti have been controlled by biocontrol agents several times. Targeted species include *O. stricta* (prickly pear cactus), *Cylindropuntia fulgida* var. *fulgida* (jumping cholla), *Pereskia aculeata* (Barbados "gooseberry"), and *Cereus jamacaru* (queen of night cactus). While little has been achieved against *P. aculeata* (Paterson et al. 2011), *O. stricta* has been completely controlled by *Cactoblastis cactorum* in several locations (Dodd 1940; Paterson et al. 2011) and partial control has been achieved against jumping cholla (Paterson et al. 2011). While not specifically documented, dense stands of cacti such as those that once dominated large regions in South Africa, certainly caused declines in abundance of native species (Hoffmann 2011).

Among invasive plants, grasses may be particularly damaging to biodiversity because of their effects on fire cycles (Brooks and Pyke 2001). However, few grasses have been targets for biological control because of concerns for the economic value of introduced grasses and the assumption that grass-feeding insects were not sufficiently specialised for introduction. Currently some grasses (e.g. *Arundo donax* in the United States) are targets of biocontrol projects (Goolsby and Moran 2009; Goolsby et al 2011) and pathogenic fungi as well as insects have been of particular interest (e.g. Palmer et al. 2008).

26.6.5 Forest Invaders

Invasive plants in forest communities that have been targeted for biological control include (i) forbs: *Alliaria petiolata*, (ii) vines: *Anredera cordifolia* (Madeira vine), *Cryptostegia grandiflora* (rubber vine), *Dioscorea bulbifera* (air potato), *Dolichandra unguis-cati* (= *Macfadyena unguis-cati*, cats claw), and *Persicaria perfoliata*, (iii) shrubs: *Solanum mauritianum* (tree tobacco), and (iv) trees: *Caesalpinia decapetala* (Mauritius thorn), *M. calvescens* (Table 26.1). Of these, projects against *M. calvescens*, *C. grandiflora*, and *P. perfoliata* have had some success.

Miconia calvescens is a small, broad-leaved tree from the Americas that invaded natural forests on Pacific islands, including Hawaii and Tahiti and formed dense monocultures that suppressed native vegetation (Meyer and Florence 1996; Medeiros et al. 1997; Meyer 1998). The fungus *Colletotrichum gloeosporioides* forma specialis *miconiae* from Brazil (Killgore et al. 1999) was released in Tahiti and caused partial defoliation (up to 47 %) in mesic and wet forests below 1,400 m, which allowed substantial recovery of native vegetation (Meyer et al. 2008, 2009).

Cryptostegia grandiflora invaded forested areas along rivers in the dry tropics of Queensland, Australia, and later spread into adjacent grasslands and savannas (Tomley 1995). Dense stands killed eucalyptus trees and reduced native biodiversity, with infested areas being avoided by native birds (Bengsen and Pearson 2006) and lizards (Valentine et al. 2007). In drought-prone areas, *C. grandiflora* has been controlled by the rust *Maravalia cryptostegiae* (Evans and Tomley 1994; Vogler and Lindsay 2002) and the pyralid moth *Euclasta whalleyi* (Mo et al. 2000), allowing increased growth of local grasses (Palmer and Vogler 2012).

Persicaria perfoliata, a spiny annual vine of Asian origin, invades forest edges and disturbed open areas within forests in the mid-Atlantic region of the United States (Hough-Goldstein et al. 2008), degrading wildlife habitat and out-competing native plants, due to its early germination, rapid growth, and ability to climb over other plants (Wu et al. 2002). *Rhinoncomimus latipes* established at release sites (Hough-Goldstein et al. 2009, 2012) and reduced spring plant densities by 75 % within 2–3 years.

26.6.6 Coastal Invasive Plants

Plants of several forms have invaded a variety of coastal habitats, including mudflats, sand dunes, littoral grasslands, and forests. Species targeted for biological control have included *Chrysanthemoides monilifera* subsp. *rotundata* (bitou bush), *Asparagus asparagoides* (bridal creeper), *Acacia cyclops* (rooikrans), *Spartina* spp. (cordgrasses), and *Ulex europaeus* (gorse). Of these, populations of *C. monilifera*, *A. asparagoides*, and *A. cyclops* (discussed above under fynbos invaders) have been partially suppressed.

Chrysanthemoides monilifera subsp. *rotundata* invaded over 80 % of the coastline of New South Wales, Australia (Thomas and Leys 2002), where it dominated sand dunes, coastal grasslands, heath, woodlands, and rainforests and drastically altered these communities, becoming the dominant threat to 150 native plants in 24 plant communities (DEC 2006). Four introduced insect species established (Adair et al. 2012) and reduced flowering and seed production (Holtkamp 2002; Edwards et al. 2009), making a contribution toward suppression. Plant density, however, has yet to decline (Adair et al. 2012).

Asparagus asparagoides invaded coastal shrublands, woodlands, and forests in Australia (Morin et al. 2006a), where it smothered natural vegetation. In Western Australia, areas infested with this species had only half as many native plant species as nearby non-invaded areas (Turner et al. 2008a). It also threatened four endangered ecological communities in New South Wales – littoral rainforest, river-flat eucalypt forest on coastal floodplains, swamp-oak floodplain forest, and subtropical coastal floodplain forest (Downey 2006), as well as threatening many native plants, including the orchid *Pterostylis arenicola* (Sorensen and Jusaitis 1995) and the shrub *Pimelea spicata* (Willis et al. 2003). An introduced rust fungus *Puccinia myrsiphylli*, a leaf beetle *Crioceris* sp., and an undescribed Erythroneurini leafhopper have established. The leafhopper has had some effect, but the rust fungus caused significant reduction in *A. asparagoides* densities (Morin and Edwards 2006; Morin et al. 2006b; Turner et al. 2008b; Morin and Scott 2012).

26.7 Conclusions

The affection of people for novel plants ensures that plants will continue to be moved into new biogeographical regions where some will become invasive, sometimes in protected nature reserves. Given that prospect, use of biological control to dampen the impacts of the most damaging of these species in protected areas and landscapes generally is and will likely remain an important restoration tool. For example, without biological control the Everglades, a World Heritage Site, may have been abandoned to become a biologically impoverished *Melaleuca quinquenervia* swamp forest, many tropical rivers around the world would be burdened with over capping layers of floating exotic weeds, and fynbos habitats would be converted to woodlands of exotic trees. Both the benefits of classical weed biological control to native plants (Van Driesche et al. 2010) and the limited nature of the entailed risks (Pemberton 2000) are now better recognised. Improved communication between biological control scientists and conservation biologists (Van Driesche 2012) and emerging mutual trust should allow the use of biological control to help resolve some of the worst cases of invasive plants in natural areas, including in legally protected reserves.

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