Diversity and Distributions, (Diversity Distrib.) (2011) **17**, 1030–1046

A Journal of Conservation Biogeography

Diversity and Distributions

SPECIAL ISSUE: HUMAN-MEDIATED INTRODUCTIONS OF AUSTRALIAN ACACIAS—A GLOBAL EXPERIMENT IN BIOGEOGRAPHY

BIODIVERSITY REVIEW

Risk assessment, eradication, and biological control: global efforts to limit Australian acacia invasions

John R. U. Wilson[1,2*], Carla Gairifo[2,3], Michelle R. Gibson[2], Margarita Arianoutsou[4], Baki B. Bakar[5], Stéphane Baret[6], Laura Celesti-Grapow[7], Joseph M. DiTomaso[8], Jean-Marc Dufour-Dror[9], Christoph Kueffer[10], Christian A. Kull[11], John H. Hoffmann[12], Fiona A. C. Impson[12], Lloyd L. Loope[13], Elizabete Marchante[14], Hélia Marchante[15], Joslin L. Moore[16,17], Daniel J. Murphy[18], Jacques Tassin[19], Arne Witt[20], Rafael D. Zenni[21] and David M. Richardson[2]

[1]South African National Biodiversity Institute, Kirstenbosch National Botanical Gardens, Claremont 7735, South Africa, [2]Centre for Invasion Biology, Department of Botany and Zoology, Stellenbosch University, Private Bag X1, Matieland 7602, South Africa, [3]Departamento de Engenharia Florestal, Instituto Superior de Agronomia da Universidade Técnica de Lisboa, Portugal,[4]Department of Ecology and Systematics, Faculty of Biology, University of Athens, Athens 15784, Greece, [5]Institute of Biological Sciences, University of Malaya, 50603 Kuala Lumpur, Malaysia, [6]Parc national de La Réunion, 112 rue Ste-Marie, 97400 St-Denis, Réunion, France, [7]Department of Environmental Biology, Univ Roma La Sapienza, I-00185 Rome, Italy, [8]Department of Plant Sciences, Mail Stop 4, One Shields Ave., University of California, Davis, CA 95616, USA, [9]Environmental Policy Center, Jerusalem Institute for Israel Studies, Jerusalem, Israel, [10]Institute of Integrative Biology, Plant Ecology Group, ETH Zürich, CH-8092 Zürich, Switzerland, [11]School of Geography and Environmental Science, Monash University, Melbourne, Vic. 3800 Australia, [12]Department of Zoology, University of Cape Town, Rondebosch 7700, South Africa, [13]U.S. Geological Survey, Pacific Island Ecosystems Research Center, PO Box 246, Makawao, HI 96768, USA, [14]Department of Life Sciences, Centre for Functional Ecology, University of Coimbra, Apartado 3046, 3001-401 Coimbra, Portugal, [15]Department of Environment, Centre for Studies of Natural Resources, Environment and Society, Escola Superior Agrária de Coimbra, 3040-316 Coimbra, Portugal, [16]Australian Centre of Excellence for Risk Analysis, University of Melbourne, Parkville, Vic. 2010, Australia, [17]Australian Research Centre for Urban Ecology, Royal Botanic Gardens, Melbourne, c/o School of Botany, University of Melbourne, Vic. 3010, Australia, [18]National Herbarium of Victoria, Royal Botanic Gardens Melbourne, Birdwood Avenue, South Yarra, Vic. 3141, Australia, [19]UPR 105, Cirad, Campus International de Baillarguet, 34398 Montpellier Cedex 5, France, [20]CABI Africa, PO Box 633-00621, Nairobi, Kenya, [21]Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, TN, USA

*Correspondence: John Wilson, Centre for Invasion Biology, Department of Botany & Zoology, Stellenbosch University, Private Bag X1, Matieland 7602, South Africa.
E-mail: john.wilson2@gmail.com

ABSTRACT

Aim Many Australian *Acacia* species have been planted around the world, some are highly valued, some are invasive, and some are both highly valued and invasive. We review global efforts to minimize the risk and limit the impact of invasions in this widely used plant group.

Location Global.

Methods Using information from literature sources, knowledge and experience of the authors, and the responses from a questionnaire sent to experts around the world, we reviewed: (1) a generalized life cycle of Australian acacias and how to control each life stage, (2) different management approaches and (3) what is required to help limit or prevent invasions.

Results Relatively few Australian acacias have been introduced in large numbers, but all species with a long and extensive history of planting have become invasive somewhere. Australian acacias, as a group, have a high risk of becoming invasive and causing significant impacts as determined by existing assessment schemes. Moreover, in most situations, long-lived seed banks mean it is very difficult to control established infestations. Control has focused almost exclusively on widespread invaders, and eradication has rarely been attempted. Classical biological control is being used in South Africa with increasing success.

Main conclusions A greater emphasis on pro-active rather than reactive management is required given the difficulties managing established invasions of Australian acacias. Adverse effects of proposed new introductions can be minimized by conducting detailed risk assessments in advance, planning for on-going monitoring and management, and ensuring resources are in place for long-term mitigation. Benign alternatives (e.g. sterile hybrids) could be developed to replace existing utilized taxa. Eradication should be set as a management goal more often to reduce the invasion debt. Introducing classical biological control agents that have a successful track-record in South Africa to other regions and identifying new agents (notably vegetative feeders) can help mitigate existing widespread invasions. Trans-boundary sharing of information will assist efforts to limit future invasions, in particular, management strategies need to be better evaluated, monitored, published and publicised so that global best-practice procedures can be developed.

Keywords

Australian weed risk assessment protocol, biological invasions, classical biological control, eradication, invasion debt, *Racosperma*.

DOI:10.1111/j.1472-4642.2011.00815.x
http://wileyonlinelibrary.com/journal/ddi

INTRODUCTION

It is still unclear to what extent biological invasions, and any impacts they might cause, can be predicted (e.g. Williamson, 1999). This is in part because of the many ways in which an invasion could arise (Blackburn *et al.*, 2011). This creates a conflict in natural resource management – what should be done to minimize the costs of biological invasions without unduly limiting the potential benefits derived from introduced organisms. Progress towards predicting future invaders has been made by focussing on particular taxonomic or functional groups (Paynter *et al.*, 2003; Richardson & Rejmánek, 2004; Harris *et al.*, 2007). Similarly, although invasions are context dependent, valuable insights into management have been gained from different parts of the world (Richardson *et al.*, 2008; Simberloff *et al.*, 2010). For best practices to be developed and implemented, the successes and failures of different management actions need to be documented and shared, see Richardson *et al.* (2008) and Simberloff *et al.*(2010) for relevant insights regarding *Pinus* species. Developing general management guidelines for a particular taxonomic group across different biogeographical regions can potentially save resources by focussing attention on the most critical aspects required for successful management.

Australian acacias (also termed 'wattles'; here defined as the 1012 species formerly placed in *Acacia* subgenus *Phyllodineae* DC. that are native to Australia; see discussion in Richardson *et al.*, 2011; Murphy *et al.*, 2010; Miller *et al.*, 2011) are a group of leguminous woody plants that include some of the most important plant invaders globally (Richardson & Rejmánek, 2011). They are a model group for studying the ecology, management and biogeography of plant invasions (Richardson *et al.*, 2011). In particular, this group presents an interesting conundrum because invasive species have a range of impacts [e.g. on water use, fire regimes, soil nitrogen levels, and directly on biodiversity through competition for space (Richardson & van Wilgen, 2004; Marchante *et al.*, 2008; Gaertner *et al.*, 2009; Le Maitre *et al.*, 2011)], but also a wide variety of commercial, subsistence and ornamental uses (Rinaudo & Cunningham, 2008; Griffin *et al.*, 2011; Kull *et al.*, 2011; van Wilgen *et al.*, 2011). The group also provides an excellent model to look for generalities across regions; for example, although *Acacia salicina* can invade arid areas [e.g. the Negev Desert in Israel (Dufour-Dror, 2010)], the same species is recommended for restoring degraded arid ecosystems in southern Tunisia (Jeddi *et al.*, 2009). Can the potential benefits of *A. salicina* for Tunisia be achieved safely given the known problems the species causes in Israel? Similarly *Acacia saligna* has been used in Ethiopia to rehabilitate overgrazed and eroded lands (Reubens *et al.*, 2011), and plants are sold in the arid regions of the south-western USA, despite the well-documented, widespread and transformative nature of *A. saligna* invasions in South Africa, Israel and other regions.

About one-third of Australian acacias have been introduced to regions outside Australia, but only 23 have become invasive (Richardson & Rejmánek, 2011; Richardson *et al.*, 2011). Interestingly, the known invaders are also the most heavily traded taxa (Fig. 1a, Griffin *et al.*, 2011). For those Australian acacias that are invasive, there are also substantial differences between the extents of invasions in different regions; for example, eleven Australian acacias in South Africa are amongst the most widespread invaders in the region, while other species are invasive (or only naturalized) at only a few sites (Fig. 1b). Again there is a strong correlation between history of usage and the extent of invasion (van Wilgen *et al.*, 2011). This is similar to *Pinus* (Procheş *et al.*, 2011), where the amount of planting is closely correlated with extent of invasions, at least in South Africa.

Two plausible explanations can account for these correlations. First, those species that are most likely to become invasive are also those most suited for utilization, and so are those that humans have spread most widely. Alternatively, invasiveness in this group is mediated by propagule pressure, and so the extent of invasions is the result of greater introduction and dissemination efforts. Even if the former were true, and managers needed to concentrate only on existing known invaders, Australian acacia invasions are still likely to increase in global extent over the next few decades (Richardson *et al.*, 2011). Within invaded ranges, species are spreading further; some species that are known to be widespread invaders have not yet been introduced in significant numbers to all suitable regions around the world, and we expect that some species currently at low densities will become widespread invaders in future simply because of population growth and spread. In short, there is a significant invasion debt in the group (*sensu* Seabloom *et al.* 2006). Reducing this debt will require management strategies that focus on multiple fronts and at all stages of the introduction-naturalization-invasion continuum.

The aim of this paper is to review the options available to limit invasions and manage the risks of undesirable impacts, in essence looking for pro-active rather than reactive management strategies. How the proposals presented here should be implemented given regional and local contexts (in particular where there is high commercial, subsistence, or cultural value) is a major and pressing research and social challenge, but not one that we discuss in this paper. The focus on identifying generic management approaches to minimize invasion risk should be viewed as one aspect of the broader discussion on Australian acacia introductions and read with reference to other papers in this special issue of *Diversity and Distributions* that describe the beneficial and cultural aspects of wattle introductions (Carruthers *et al.*, 2011; Griffin *et al.*, 2011; Kull *et al.*, 2011).

Our approach was threefold. First, we developed a generalized life cycle of Australian acacias and considered control options at each life stage. We then reviewed general management approaches to reducing the invasion risk of Australian acacias. Finally, we determined tools necessary to achieve pro-active control. The information is based on primary and secondary literature sources, knowledge and experience of the authors, deliberations at the workshop from

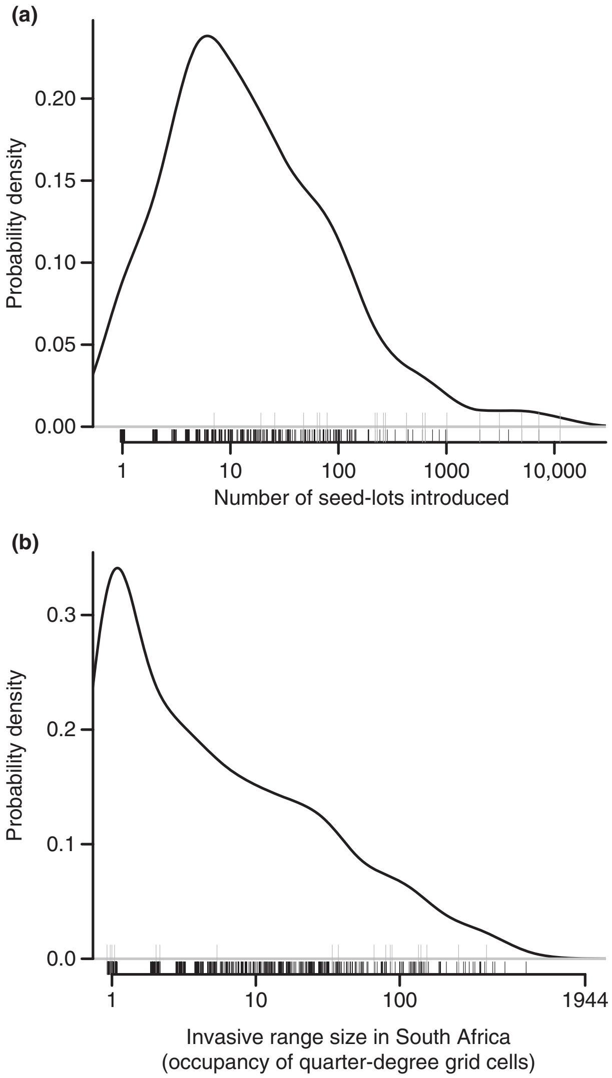

Figure 1 Invasive Australian acacias tend to be (a) those wattles that were introduced in the largest numbers and (b) more widespread than other plant invaders. (a) The number of seed lots exported by the Australian Tree Seed Centre from Australia for ~300 species of wattle between 1980–2010 (Griffin *et al.*, 2011). The fitted line is a probability density converting the discrete numbers into a smooth distribution that can be interpreted as the likelihood of observing any particular value given a random selection (function density{stats} in the programme R). Species that are unequivocally invasive (*sensu* Pyšek *et al.*, 2004) are the longer, dark-grey ticks. Values are integers, but with some error added to limit overplotting. (b) Frequency distribution of range size for all invasive plants in South Africa at the quarter-degree grid cell scale. Invasive wattles are shown as longer, dark-grey ticks. Data are from the South African Plant Invaders Atlas (2009). These data include historical records that have not been confirmed on resurvey (e.g. *A. fimbriata* and *A. cultriformis*), but do not include more recent confirmed records of localized naturalization (e.g. *A. ulicifolia*). Continental South Africa covers 1944 quarter-degree grid cells. Invasive Australian acacias are not a random sample of all species found in SAPIA ($P < 0.001$ from a Kolmogorov–Smirnov two-sample test), they tend to have either very narrow or very wide distributions.

which this special issue of *Diversity and Distributions* grew (Richardson *et al.*, 2011), and results from a questionnaire sent to experts around the world (see Appendices S1–S3 in Supporting Information).

CONTROLLING AUSTRALIAN ACACIAS AT DIFFERENT POINTS IN THEIR LIFE CYCLE

All Australian acacias have the same basic life cycle (Fig. 2), and as such potential management interventions apply across the group. Here we review control at each stage of a generalized life cycle starting with seedlings.

Australian acacia seedlings are capable of rapid growth under a variety of environmental conditions and have the ability to out-compete many other species for resources (Morris *et al.*, 2011). However, a high percentage of seedlings do not survive to saplings, so interventions should target successful seedlings to avoid wasting resources. This can include direct control through hand pulling, grazing, foliar herbicides, ploughing, or limiting recruitment opportunities by changing land management (e.g. grazing or fire regimes).

Reproductive maturity is reached after 1–5 years. Adult plants can be killed (e.g. through debarking, fire or mechanical control), but as many species resprout or coppice (and some species have a strong ability to sucker), herbicide treatment of stumps and additional follow-up to treat regrowth are often necessary. Classical biocontrol agents can be introduced to target vegetative growth (Box 1).

Australian acacias have showy, long-lasting floral displays whose rewards are available to a variety of generalist insect pollinators (Gibson *et al.*, 2011). In consequence throughout their introduced ranges, trees produce large quantities of viable seeds [up to 15,000 seeds m^{-2} per annum (Richardson & Kluge, 2008)]. Reducing seed production (e.g. through the use of biological control agents, Box 1) can limit both spread rates and the build-up of seed banks. More localized options can also reduce seed set, e.g. harvesting of flowers, hormonal control to reduce seed set or flowering, use of sterile cultivars, or harvesting before reproductive maturity.

Natural seed dispersal can be through ants or birds, and also running water, but probably the majority of seeds simply disperse via gravity. Seed predation by rodents and alydid bugs can significantly reduce the numbers of seeds prior to incorporation into seed bank (David & Jarvis, 1985; Holmes & Rebelo, 1988; Holmes, 1990a,b; Richardson & Kluge, 2008). Seed dispersal can be reduced by restricting the movement of soil that contains seed, and controlling plants or preventing plantings close to dispersal routes (e.g. roads, water-ways, and areas where there is significant water run-off).

Once seeds have dispersed, they are incorporated into the seed bank mostly in the top 10 cm of soil. These seeds are potentially long-lived and might remain dormant for many decades, germinating *en masse* following disturbance events (e.g. fire). Therefore, while management can be effective in clearing adult plants, reducing existing soil seed banks is essential to prevent widespread reinvasion after clearing (Box 2).

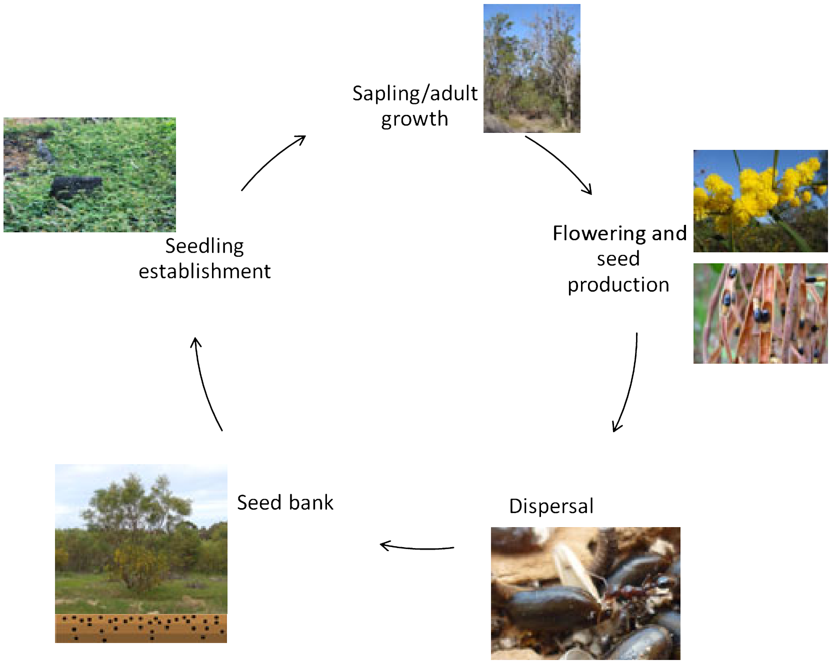

Figure 2 A generalized life cycle of an Australian acacia (based on information in Gibson *et al.*, 2011). Although there is substantial phenological variation in adult plant size and structure, Australian acacias share many traits. The most challenging for management (both in terms of controlling and as a threat to the sustainability of control) is their tendency to form large and persistent seed banks (Box 2).

APPROACHES TO REDUCING THE INVASION RISK OF AUSTRALIAN ACACIAS

Given the difficulties of controlling established invasions, because of seed banks in particular, priority should be given to preventative measures.

Controlling introductions – pre-border risk assessment

Historically, Australian acacias were typically introduced for (agro)-forestry or soil stabilization, although there are several notable ornamental species and many species have a variety of uses (Kull *et al.*, 2011). Australian acacias were mostly introduced deliberately as seed (Poynton, 2009; Griffin *et al.*, 2011). Given the size of the seeds (~5 mm), the likelihood of accidental introductions in contaminated soil is lower than for many other plants (although contaminated soil used as ship ballast might explain some of the historical trans-oceanic dispersal, and the movement of soil during construction or road maintenance can be a significant dispersal route within a region). Similarly, seed lots are generally unlikely to have been contaminated as seed can be relatively easily gathered from adult plants. There have been, however, some errors of identification during introduction and planting, for example, *Acacia mearnsii*, *Acacia dealbata* and *Acacia decurrens* were confused in early colonial Madagascar (Kull *et al.*, 2007). Furthermore, the main reasons for introduction (i.e. commercial or environmental applications) are such that most growers will not want to consider prohibited species – the possible exception being specialist horticulturalists. Widespread invasive species like *Acacia baileyana*, *A. dealbata*, *Acacia melanoxylon* and *A. saligna* are available for purchase online, apparently without restriction. While awareness campaigns can assist in reducing the introduction risk posed by the rapidly expanding online plant trade (Derraik & Phillips, 2010), quarantine inspections may be the only method of controlling some ornamental introductions.

Which species will be traded in future? Most species suitable for commercial forestry, i.e. those that can grow in excess of five metres, have already been identified (Griffin *et al.*, 2011), although existing forestry species might be used more extensively in the wet tropics. However, various new species are being assessed for horticulture, in particular for the cut flower and/or foliage industry (Ratnayake & Joyce, 2010), for ornamentals and street trees (Ducatillion & Blanc-Chabaud, 2010), for agroforestry (Rinaudo & Cunningham, 2008), for feed-stock and even for human nutrition (Rinaudo *et al.*, 2002). Pressure to introduce a range of Australian acacias for erosion control and rehabilitation, as well as for bio-fuels and carbon-capture, is likely to increase (Kueffer & Vos, 2004), particularly as global change will tend to lead to more habitats being overused and degraded with very nutrient-poor soils, dry climate, and high risk of wildfire (situations suitable for many Australian acacia species).

These new introductions of acacias will need to be managed carefully if widespread invasions and negative impacts are to be avoided. This could be achieved primarily through the use of import permits based on a risk-assessment framework and cost-benefit analysis. To determine the level of risk-assessment required, we used the Australian Weed Risk Assessment

Box 1 Classical biological control of Australian acacias

To date, South Africa is the only country to have deliberately released biological control agents (nine insect herbivore species and a fungus, see the table below). The only other countries involved in biological control of wattles are Portugal, where host-specificity testing for *Trichilogaster acaciealongifoliae* on *Acacia longifolia* has been carried out (Marchante *et al.*, 2011), and New Zealand, where several biological control agents are presumed to have been accidentally introduced and have since spread naturally (Hill *et al.*, 2000; Impson *et al.*, 2009).

Biological control in South Africa has largely been limited to agents that do not damage the vegetative parts of their host plant (i.e. only attack the flower buds, flowers or seed pods) so as to minimize the impact on commercial production of wattles (Dennill & Donnelly, 1991). While most of the agents released have been shown to have no direct or indirect effect on the growth or survival of host plants (Hoffmann *et al.*, 2002; Dorchin *et al.*, 2006; Moseley *et al.*, 2009), the large reductions in seed production reduce the costs of follow-up management and spread rates. Agents that have more direct impacts include the rust-fungus *Uromycladium tepperianum*, which reduces the life expectancy of adult *A. saligna* plants (Morris, 1997; Wood & Morris, 2007), and *T. acaciealongifoliae* which, by inducing galls on *A. longifolia* flower-buds, directly prevents seed set and also increases plant mortality by creating a resource sink that reduces the ability of plants to survive stress (Dennill, 1988).

The programme against *A. longifolia* provides an excellent indicator of what might be expected from biological control. This has been the longest running programme against an Australian acacia in South Africa, commencing in 1982 (Dennill & Donnelly, 1991; Dennill *et al.*, 1999), and has resulted in two agents being released (see the table below). The direct and indirect damage caused by *Trichilogaster acaciaelongifoliae*, along with the seed-feeding weevil, *Melanterius ventralis*, natural fire cycles and manual control operations, has resulted in a decline in the status of *A. longifolia* as a weed to the extent that the plant is no longer considered to be problematic in most areas of its invasive range in South Africa (F.A.C. Impson, personal communication).

The long history of the biological control programme on wattles in South Africa provides a valuable resource for other countries considering biological control options for this group of plants. The experience to date is particularly relevant for countries where Australian acacias are recognized as being problematic but where the plants also provide social and commercial benefits (Impson *et al.*, 2009). For those countries assessing new importations of Australian acacias, strong consideration should be given to the simultaneous introduction of seed-reducing agents (thereby minimizing spread from a designated area, the development of large seed banks and limiting postharvest weedy regeneration).

There are still many potential biological control agents that have not been fully assessed, including several reproductive feeders (Kolesik *et al.*, 2010; Impson *et al.* 2011), but also, given the previous limitations on agent selection in South Africa, many root and stem borers (Impson *et al.* 2011). The larvae of several genera of Cerambycid beetles and several species of both the goat and ghost moths (Cossidae and Hepialidae) are commonly associated with many of the Australian acacias (McKeown, 1947; Common, 1990). While these insect groups tend to be somewhat polyphagous, this type of agent has been successfully used against other invasive woody legumes, e.g. a stem-boring Sesiidae moth has contributed to the successful control of *Mimosa pigra*, in northern Australia (Paynter, 2005).

Table Box 1 Biological control agents released in South Africa. Agents that cause reductions in growth rates of the host plant are shown with *, all other agents reduce rates of reproduction.

Acacia species	Agents	Year of introduction
A. baileyana	*Melanterius maculatus* (Coleoptera: Curculionidae)	2006
A. cyclops	*Melanterius servulus* (Coleoptera: Curculionidae)	1994
	Dasineura dielsi (Diptera: Cecidomyiidae)	2001
A. dealbata	*Melanterius maculatus* (Coleoptera: Curculionidae)	1998
A. decurrens	*Melanterius maculatus* (Coleoptera: Curculionidae)	1998
A. longifolia	**Trichilogaster acaciaelongifoliae* (Hymenoptera: Pteromalidae)	1983
	Melanterius ventralis (Coleoptera: Curculionidae)	1985
A. mearnsii	*Melanterius maculatus* (Coleoptera: Curculionidae	1995
	Dasineura rubiformis (Diptera: Cecidomyiidae)	2002
A. melanoxylon	*Melanterius acaciae* (Coleoptera: Curculionidae)	1986
A. podalyriifolia	*Melanterius maculatus* (Coleoptera: Curculionidae)	2009
A. pycnantha	**Trichilogaster signiventris* (Hymenoptera: Pteromalidae)	1992
	Melanterius maculatus (Coleoptera: Curculionidae)	2005
A. saligna	**Uromycladium tepperianum* (Uredinales: Pileolariaceae)	1987
	Melanterius compactus (Coleoptera: Curculionidae)	2001

Protocol (Pheloung *et al.*, 1999; Gordon *et al.*, 2010). For each question, we estimated which answers could apply to any member of the group (e.g. no taxa are aquatic, but while most taxa do not have spines or thorns, some do) using the literature and knowledge of the authors. This provided lower and upper bounds on the potential risk-assessment score. Assuming that the proposed introduction was to a climatically suitable location, the overall range scored was 1–40 (Appendix S4), where scores of 1–6 indicate that further evaluation is required [e.g. obtaining more data or undertaking further investigations such as field trials (Pheloung *et al.*, 1999)] and scores > 6 indicate an introduction should be rejected. In summary, the general attributes of wattles (N-fixing, animal-dispersed seeds, long-lived seed bank) mean that prior to introduction, all taxa

Box 2 Controlling seed banks of Australian acacias

Most invasive wattles produce copious numbers of seeds that accumulate as extensive and abundant seed banks (Gibson *et al.*, 2011). Although the longevity of wattle seeds has not been fully investigated, anecdotal reports from Australia suggest that seed can survive for more than a century (based on observations of historically cleared land that recently has been allowed to burn). Even if it is more typical for seeds to persist for decades, it is clear that invasions can regenerate long after clearing. As such, seed banks represent a fundamental challenge to management in this group (Holmes *et al.*, 1987; Holmes, 1989a,b; Richardson & Kluge, 2008). There are several methods to reduce the size of existing seed banks:

- *Fires* scorch and destroy seeds in the surface layers and stimulate others to germinate *en masse*, a combination of which substantially reduces seed banks (Pieterse & Cairns, 1986, 1988). A thorough follow-up or repeat burn is needed to treat emerging seedlings.
- *Disturbance* – clearing operations and other forms of soil disturbance promote high levels of germination, and this effect can be used intentionally, e.g. raking, to deplete the seed bank. Again this will result in a large number of seedlings.
- *Litter removal* can be effective in removing seeds before they are incorporated in the seed bank (Marchante *et al.*, 2010b), one method being to vacuum around adult plants.
- *Earth covering* – seeds germinating more than 10 cm below the soil surface have a reduced chance of reaching the surface, and so covering an invaded site with 20 cm of earth can prevent recruitment.
- *Soil inversion or removal* of the top 20 cm of soil can significantly reduce seed survival and numbers (Cohen *et al.*, 2008).
- *Solarization* – areas exposed to sunlight are covered in plastic, and the resulting increase in soil temperature induces germination and kills seedlings. This was found to deplete *A. saligna* seed banks in experimental plots in Israel (Cohen *et al.*, 2008), but field trials were less successful because the plastic sheets were easily removed or damaged (e.g. by wild animals).

In summary, while there are several methods that can reduce existing soil seed banks, they tend to be highly destructive, resource intensive, only practical over small areas, unsuitable for use in natural areas, and may require extensive follow-up management to deal with germinating seedlings.

should be examined in more detail than is required for an initial screening using the Australian Weed Risk Assessment Protocol (Appendix S4). When we attempted other such schemes (e.g. Brunel *et al.*, 2010), similar results were obtained. Indeed we know of only one species where a completed risk assessment suggested there was a low invasion risk [*A. stenophylla* in Hawaii (PIER, 2003)].

The limitations of existing risk assessments have stimulated research on predictive correlates of invasiveness (Castro-Díez *et al.*, 2011; Hui *et al.*, 2011). As a preliminary effort to combine these studies with data on the history of invasion and naturalization [including information from within Australia (Maslin & McDonald, 2004)], we developed four categories of invasion risk (Table 1). The only species classified as low risk are those with a long history of introduction but no history of invasiveness. One major problem with this scheme is the degree to which improved information changes ratings; for example, while there are some reports of naturalization and spread of *A. leptocarpa* in Tanzania (Haysom & Murphy, 2003), the extent to which this species is actually invasive needs to be determined. Similarly the low invasion risk predicted for *A. stenophylla* in Hawaii (PIER, 2003) appears to contradict other anecdotal information online (e.g. http://forums.gardenweb.com/forums/load/azgard/msg0500014111061.html). While this approach might become useful, it clearly requires more detailed research and consideration (Box 3).

Managing naturalization

There is often a time delay before an introduced taxon is observed to invade; for example, in Israel it took about 50 years for *A. saligna* and 75 years for *Acacia cyclops* to be regarded as invasive (Dufour-Dror, 2010), while it was 10 and 20 years after *A. mearnsii* and *A. mangium* had been widely planted on La Réunion and Mayotte respectively before they became invasive (Tassin & Balent, 2004). These lag phases are comparable to those seen with other woody invaders [e.g. 15 years on average on a tropical island (Daehler, 2009), but well over a century on average for South Australia (Caley *et al.*, 2008)]. Any delay between deriving benefits and observing impacts is particularly problematic for proactive management, but equally offers opportunities for preventative measures to be instigated in time (i.e. before exponential population growth occurs).

If management resources are to be allocated efficiently, we need to understand why certain species establish and spread, and which factors prevent or limit establishment and spread. There are several reasons to explain a lack or delay in naturalization. It might be partially explained by a lack of suitable sites (e.g. climatic or edaphic) (Wilson *et al.*, 2007). Efforts have been made to rehabilitate degraded sites in northern Greece by planting *A. saligna*, but saplings could not cope with low winter temperatures and frequent frost. Similarly, despite repeated large-scale efforts to use *Acacia pycnantha* for drift-sand control in South Africa, *A. pycnantha* has only established and spread at a few sites on more rocky ground (Poynton, 2009). Wattles might also require a particular fire or flooding regime to launch widespread invasions (Witkowski, 1994; Roura-Pascual *et al.*, 2009). The presence of a particular fire cycle might partially explain the success of *A. saligna* invasions in Mediterranean ecosystems, but the relatively slow spread seen in Ethiopia. One other oft-cited reason for invasions to fail – a lack of mutualists

Table 1 Preliminary risk assessment categorization for Australian acacias.

Invasive potential	Criteria	Recommendations	*Acacia* species
Extremely high	Evidence for invasion outside of Australia	No new utilization without co-introduction of seed reducing agents, and long-term planning to deal with any invasions (e.g. through insurance), or the use of sterile varieties Identify and remove populations of these in climatically suitable countries where they are not yet widespread	*A. auriculiformis*, *A. baileyana*, *A. crassicarpa*, *A. cyclops*, *A. dealbata*, *A. decurrens*, *A. elata*, *A. holosericea*, *A. implexa*, *A. iteaphylla*, *A. longifolia*, *A. mangium*, *A. mearnsii*, *A. melanoxylon*, *A. paradoxa*, *A. podalyriifolia*, *A. pycnantha*, *A. retinodes*, *A. salicina*, *A. saligna*, *A. stricta*, *A. verticillata*, *A. victoriae* (Richardson & Rejmánek, 2011)
High	Evidence of naturalization (or invasion in Australia outside native range) OR Predicted as invasive based on native range properties, or with similar characteristics as known invaders	All species subject to a detailed full risk assessment specifying mitigation methods before importation or usage Seek alternatives Any small areas without a clear value to somebody to be removed as a precaution	*A. floribunda*, *A. iteaphylla*, *A. pravissima*, *A. prominens*, *A. sophorae*… (Groves *et al.*, 2005) OR *A. binervata*, *A. floribunda*, *A. howittii*, *A. terminalis*… (Castro-Díez *et al.*, 2011; Hui *et al.*, 2011)
Moderate	Species not covered by rest of the criteria	All species subject to a detailed full risk assessment specifying mitigation methods before importation or usage Any small areas without a clear value to somebody to be removed as a precaution	Remaining species unclassified elsewhere
Low	No instances of naturalization despite widespread planting over 50 years outside of native range	Monitor any new areas for spread Old plantings to be reassessed to confirm lack of spread Comparison between climatically suitable areas to be made	*A. pendula* (?)

The table provides examples of species in the different categories, however, these categories are not currently linked to a quantitative risk class, and the criteria are at present potentially overlapping. The classification presented here is based on a situation that is close to optimal for the species under consideration. Regional and local contexts (ecological, social, economic, etc.) can dramatically reduce the invasive potential.

(Richardson *et al.*, 2000a) – is less likely to apply. Wattles, or at least the invasive ones, have readily formed new associations with rhizobia, pollinators and seed dispersers (Rodríguez-Echeverría, 2010; Gibson *et al.*, 2011; Rodríguez-Echeverría *et al.*, 2011). Many Australian acacias produce large quantities of viable seed in their introduced range despite evidence of reproductive self-incompatibility within the group (Gibson *et al.*, 2011). Perhaps the greatest factor influencing Australian acacia invasions is pressure from humans. Constant harvesting, grazing and land cultivation can limit rates of spread. Providing such uses continue, naturalization will be managed at a level where impacts remain low (Kull *et al.*, 2011), but it is debatable whether utilization can prevent an invasion.

Even if there is no current evidence of an invasion at a site, the situation can clearly change, and so continued monitoring and potentially some management is required. Between 1974 and 1984 over 40 species of wattle were imported to various countries in West Africa for use in arid areas denuded of woody vegetation (Cossalter, 1986). *Acacia colei*, *A. torulosa* and *A. tumida* were introduced to Niger and have since been planted extensively. While these species appear to rarely self-sow, they still represent an invasion risk that should be monitored (Box 3).

Early detection procedures will be particularly important near biodiversity conservation sites (Kull *et al.*, 2008), and areas surrounding planting sites can be (and often are) regularly monitored for signs of recruitment (Mochiutti *et al.*, 2007), although this is not always the case [e.g. *A. auriculiformis* and *A. mangium* plantations in Malaysia (B.B. Bakar & M. Ohsawa, unpublished data)]. Arboreta and botanic gardens could be similarly aware and pro-active if resources permit; for example, all trees of *A. binervata* at Tokai Arboretum in South Africa were reportedly removed for fear that the species might spread and become a weed (Poynton, 2009). Herbaria records of known invaders could also be followed up as a method of timeously detecting naturalizing populations.

Australia will likely see the largest number of new wattle species naturalizing because of the difficulty of preventing plant movements within a country and other factors, such as an enthusiastic native plant horticulture movement (Reid &

Box 3 Key areas of short and long-term research and intervention recommended to understand and manage potential invasiveness of introduced Australian acacias

Biological and ecological traits

- Further understand seed-bank dynamics (e.g. seed longevity, seed rain, seed predation and decay and seed-bank extension).
- Determine dispersal and recruitment limitations, e.g. animal dispersal and the likelihood that flooding would precipitate mass recruitment in dry areas.
- Identify correlates or predictors of invasiveness.

Biogeography of invasive Australian acacias

- Compare species across regions that differ in invasion success (e.g. cf. *A. saligna* in Cyprus, Israel and South Africa with *A. saligna* in Ethiopia).
- Compare interspecific invasion success in the context of human usage (e.g. *A. mangium* is less invasive than *A. auriculiformis* in Malaysia).
- Identify areas of current climate suitability and investigate the effect of projected climate change on future potential distributions.
- Evaluate limitations in invasiveness for species widely planted but with no record of naturalization.

Control techniques and strategies

- Search for and assess new biological control agents, particularly agents that damage vegetative parts of the plant.
- Introduce existing biological control agents to other countries after suitable risk protocols have been followed.
- Assess and optimize different methods for depleting seed banks.
- Improve methods of detecting invasions including small incipient infestations or remnant plants in an eradication programme (Goodwin *et al.*, 2010).
- Develop techniques for restoring and rehabilitating areas where populations have been eliminated.
- Work with global arboreta and herbaria to identify potential foci of new invasions.

Responsible utilization

- Encourage growers to monitor and manage spread from plantation borders.
- Develop and promote non-invasive alternatives to Australian acacias for forestry and for rehabilitation of degraded dry habitats (Kueffer & Vos, 2004).
- Encourage commercial utilization to insure against the risk of invasion.
- Include an assessment of invasive potential in experimental planting trials, and if a species is not to be used further, trials to be removed on completion (Griffin, 1990).
- Support ongoing research into managing sterility and promote the usage of sterile crops throughout the forestry industry (no sterile cultivars are yet commercialized).
- Implement a process for decommissioning land after Australian acacia production.
- Introduce biological control agents that reduce reproductive output to minimize the invasive potential of plantations.
- Develop mitigation protocols (e.g. for potential biofuel production see DiTomaso *et al.*, 2010).

Public awareness

- Increase public awareness of potential invasive impacts through out-reach initiatives (Marchante *et al.*, 2010a).
- Develop relevant information on invasiveness for producers and online acacia buyers and sellers.

Effective legislation

- Produce easily accessible information on potential and predicted risk of Australian acacia invasions.

Global and regional co-operation

- Discourage the widespread use of known invasive species in countries where species have not yet been introduced.
- Develop and disseminate an internationally applicable best management practice guideline for preventing and managing Australian acacia invasions. Guidelines should be web based, linked to existing data portals on biological invasions, and target managers, scientists, and stakeholders (e.g. http://www.cabi.org/isc).
- Develop methodology and tools to incorporate views and concerns of local stakeholders into management planning.

Appropriate time-scales for funding.

- Advocate for longer-term funding (governmental and non-governmental) so management can operate over the time-scales required to successfully control Australian acacia invasions (i.e. > 5 years).

Murphy, 2008). As such, Australia provides a test of which species are likely to naturalize and how. The main difficulty will be to separate human-mediated introductions from natural range expansion (Wilson *et al.*, 2009).

Eradication

Finding and treating populations before they become widespread is often the only point at which eradication is

Table 2 Factors affecting eradication efforts [adapted from Panetta (2009)] and the relevance of these factors for Australian acacias.

Factors that impede plant eradication	Relevance to Australian acacias	Importance
Number and spatial distributions of invasive populations	Distribution usually clumped around arboreta, forestry plantations, etc., although some species can form extensive continuous stands	Low–medium
Accessibility of invasive populations	Some species are bird-dispersed and can spread to inaccessible areas, but invasions tend to be more accessible than some other invasive plants (e.g. wind-dispersed species)	Low–medium
Lack of conspicuousness within the matrix of invaded vegetation	Wattles are trees or shrubs with large showy floral displays, and distinctive morphology, and so are more conspicuous than many grasses and herbs [although it may be difficult to separate some invasive species (e.g. *A. implexa* from *A. melanoxylon*)]	Low
Detectability prior to reproduction	As above, however, many species can reproduce when significantly smaller than 1 m tall, and within 2 years (Gibson *et al.*, 2011)	Low–medium
Reproduction through vegetative fragmentation	While many species sucker and resprout, species cannot reproduce through fragmentation	Zero
Minimum length of the pre-reproductive period	All species will require at least 1 year before seed production occurs; thereafter seed production is generally annual	Low
Maximum longevity of seeds or vegetative propagules	Wattles often develop a persistent seed bank that lasts decades	High
Number of treatments required to control the largest plants	Mechanical clearing and treatment of stumps with herbicide is usually highly effective; however, some species sucker and resprout	Zero–low
Percentage of the invaded area requiring control procedures more expensive than standard methods	Plants can grow on very steep slopes or in riverine systems; however, this concern is more a function of the invaded landscape	Site dependent
Potential for managing propagule dispersal	Seeds are visible to the naked eye, and very basic decontamination procedures will limit spread during control, but the movement of contaminated soil or seeds by birds can be problematic	Low–Medium

achievable (Mack & Lonsdale, 2002). While we know of no cases where an invasive wattle population has been eradicated, there are several instances where efforts are ongoing. *Acacia paradoxa* is being targeted in South Africa (Zenni *et al.*, 2009), and the possibility of eradicating *A. saligna* from Cyprus is under investigation by the Cyprus Forest department [P. Delipetrou & C. Christodoulou, pers. comm.); see also Appendix S3]. However, applying a recent scheme for assessing the limitations to eradication (Panetta, 2009) to Australian acacias, it is clear that the major limitation to the eradication of an invasive Australian acacia population would be the presence of a long-lived seed bank (Table 2). As such, management programmes need to be careful not to declare an eradication until the seed bank is demonstrably exhausted; for example, several wattles have been designated as eradication targets on the island of Maui, Hawaii, and while *A. retinodes* is the most promising contender, reports of its eradication are premature (Kraus & Duffy, 2010). As a precaution, species with currently limited distributions particularly those that are known as invaders and where there is no conflict of interest could be removed, e.g. from disbanded forestry plantations and trials.

Eradication is feasible only for small infestations, but the point at which eradication should be discarded in favour of other options, including containment, is a matter of ongoing research (Panetta, 2007; Moore *et al.*, 2011). Containment is a matter of understanding and controlling dispersal routes, in particular preventing the dispersal of seeds beyond production areas by, for example, contaminated soil. More broadly, it also includes efforts to reduce seed production (Box 2). There have been few explicit efforts to contain the spread of Australian acacias (Appendix S3), though clearly preventing new invasions is an important strategy (van Wilgen *et al.*, 2011). By this stage, however, effort will also have to be spent on managing the impact of the invasion.

Managing impacts

Many Australian acacias, when invasive, have the potential to rapidly transform ecosystems by fixing nitrogen, changing fire regimes and altering community dynamics (Le Maitre *et al.*, 2011). If the range of traits seen in the group is applied to a recent scheme to assess the impacts of plant invasions (Magee *et al.*, 2010), all taxa are found to have aggressive invasiveness-impact with the vast majority deemed extreme invasiveness-impact (Table 3). Indeed, several wattle species are considered transformers *sensu* Richardson *et al.* (2000b), e.g. *A. longifolia* in Portuguese dune ecosystems (Marchante *et al.*, 2008). Restoration efforts must take the impact on ecosystems and ecosystem services into account to increase chances of success (Le Maitre *et al.*, 2011).

Once species are widespread, costly exercises are required if an area is to be cleared. Maintenance management using mechanical clearing and herbicide application to cut stumps can be very effective in killing older saplings and adult trees

Table 3 Assessment of the range in impact scores for Australian acacias (Magee *et al.*, 2010).

	Trait	Notes	Score (0 or 1)
Life history	Strongly clonal	Some resprout via suckering and coppicing, but this is not a major method for spreading (Reid & Murphy, 2008)	0
	Large propagule crop	Often prolific seeder (Richardson & Kluge, 2008; Marchante *et al.*, 2010b)	0–1
	Small seeds/fruits	Seeds usually < 5 mm in longest dimension (but not by much)	0–1
	Wind dispersal	Very limited	0
	Animal dispersal	Often (O'Dowd & Gill, 1986; Gibson *et al.*, 2011)	1
	Water dispersal	Can see significant spread by water (Milton & Hall, 1981)	1
	Specialized dispersal	No	0
	Dispersal over time	Persistent seed bank and long seed life, also often staggered germination, and staggered dispersal from inflorescence (e.g. Holmes, 1989a)	1
	Plasticity	Can have some morphological, phenotypic and genetic variability	0–1
Ecological amplitude	Drought tolerant	Often	1
	Wide moisture regime	Often	1
	Flooding/saturation tolerant	Growing in wet conditions, or adapted to intermittent flooding	0
	Wide nutrient or soil texture ranges	Variable	1
	Wide light regime	Often shade tolerant	Mostly 1
	Alkaline or saline tolerant	Occasionally	Mostly 0
	Grazing tolerant or increaser	Resilient to direct grazing impacts, increases because of low palatability, to toxicity, or release from competition	Mostly 0
	Increases postfire	Often fire adapted (Pieterse & Cairns, 1986)	Mostly 1
Ecosystem alteration	Alters hydrology	Yes (Le Maitre, 2004)	1
	Alters nutrient cycling	Nitrogen-fixers (Brockwell *et al.*, 2005; Rodríguez-Echeverría *et al.*, 2011)	1
	Alters fire regime	Yes. (van Wilgen & Richardson, 1985) Can increase the fire load and intensity and act as fire breaks	1
	Alters soil stability	Many species used for dune stabilization	1
	Excretes salts or toxins	Rarely	0
	Forms monocultures or near-monocultures	Yes (Le Maitre *et al.*, 2011)	1
	Invades in absence of human disturbance	For most invasions observed this is clearly the case	Mostly 1

The group scores a range of 47–74%. As scores of 36–50% indicate aggressive invasiveness-impact and scores of 51–100 extreme invasiveness-impact, the vast majority of Australian acacias are expected to have extreme invasiveness-impact. Information is based largely on the Flora of Australia (2011) unless specified.

and preventing coppicing or suckering (of most species), but only if the proper procedures are followed. Moreover, control will almost always require extensive follow-up efforts to deal with germination from the seed banks (Box 2). Integrated management of widespread invaders can, and arguably should, consider classical biological control (van Wilgen *et al.*, 2011; Box 1). However, by the time the invasion has reached this stage, it is already often too late to prevent substantial, potentially irreversible, impacts on the ecosystem.

TOOLS NECESSARY TO ACHIEVE PRO-ACTIVE CONTROL

We have discussed several ways in which Australian acacias can be managed to minimize the risk of invasion. For these to be effective, more research on particular areas and better facilitating mechanisms are required, and conflicts of interest with those for whom wattles are valuable have to be discussed and resolved (Box 3).

Responsible utilization

There are several existing initiatives designed to secure the beneficial properties of Australian acacias while at the same time reducing the invasion risk (Box 3); for example, biological controllers have worked with foresters to try to reduce seed production without compromising commercial production of wattles (Box 1), and efforts to breed sterile triploids for use in forestry are underway in Vietnam and South Africa (Griffin *et al.*, 2011).

The benefits derived from a species often change through time, and if a given natural resource industry closes, then not only is there no agency responsible for removing any naturalizing plants but there is likely to be a large disturbed area for an invasion to progress from. *Acacia mearnsii* was introduced to La Réunion Island in 1878 for tannin production and later used as a soil improver and fuel-wood as part of the pelargonium oil industry. The collapse of the industry during the early 1960s meant *A. mearnsii* could quickly proliferate in abandoned fields and spread into natural vegetation. It is now the most widespread invasive woody plant on that island covering more than 5500 ha (Tassin *et al.*, 2006). We recommend that risk-reducing measures be in place from the start of a venture and that a long-term mechanism is instigated for managing plantation abandonment (Box 3).

Public awareness

The management of biological invasions in general requires a level of public awareness and support, but this is particularly important if there are conflicts of interest (De Poorter, 2001; Andreu *et al.*, 2009). Wattles have many beneficial uses (Griffin *et al.*, 2011) and are often a precious resource central to many rural livelihoods providing fuelwood, food, fodder, and shelter (Kull *et al.*, 2011). Marchante *et al.* (2010a) outline some recent efforts in Portugal to raise public awareness and engage the public including: updated and informative web pages where online enquiries can be responded to; field-work projects tied to information sessions; training courses for educators and biodiversity professionals; distribution of printed material (e.g. species profiles, field guides, postcards, bookmarks); and stands at science, nature, and gardening forums and fairs. Such efforts require substantial and sustained investment, with a contact for enquiries (Box 3).

Effective legislation

Few countries have regulations aimed at limiting Australian acacia invasions (Appendix S3). Portuguese regulators, based on legislation from 1999, listed several introduced species that are prohibited from usage and exploitation and which require a national plan for control. New imports of unlisted species also required risk assessment. This legislation is under review, but it has not yet been widely enforced, nor have the control plans been implemented. There is a similar situation in South Africa. While the implementation of current legislation has been partially effective in reducing the unregulated sale and spread of wattles, it has little effect on how invasions are dealt with on private property and has not been widely enforced (van Wilgen *et al.*, 2011). We know of only one case in South Africa where a property owner was fined for not clearing their land of invasive alien vegetation. Commercialization of *A. mearnsii* has been forbidden on La Réunion Island since 2006, and legislation is in development in both Cyprus and Spain (Appendix S3). For most other countries where there are regulations, there are insufficient resources or capacity to implement them, and they can often be bypassed. The forestry legislation in Madagascar is probably typical – it does not distinguish between native and introduced species and, while there are phytosanitary restrictions in place for crop pests, there are no policies designed to reduce the risk of introducing invasive plants (Kull *et al.*, 2007).

In practice, a combination of voluntary agreement and enforced legislation might be required to achieve successful management goals (Simberloff, 2009). This will require resources, support, enthusiasm, and clearly defined lines of responsibility, i.e. will depend on a level of governance that is absent from many regions. Moreover, since administrative and biogeographical boundaries often do not align, pro-active management will only be practical if international experiences and support are shared.

Global and regional co-operation

Trans-boundary co-operation regarding the management of invasive alien species that has been successful in a few cases in Asia and Oceania is still limited in Europe (Hulme *et al.*, 2009), and is essentially non-existent in Africa. We recommend the establishment of a unified platform, e.g. a website, which hosts information about Australian acacia introductions, what benefits they provide, their invasive status, the importance of preventing further spread, a description and assessment of control options, and examples of how they can be used responsibly. The platform would enable risk assessments to be shared and the efficacy of control methods compared. This global platform could also be responsible for the development and maintenance of an alert list that summarizes current knowledge about the invasive potential of Australian acacia species. This could be achieved by better integrating available information into existing databases, e.g. CABI's Invasive Species Compendium (http://www.cabi.org/isc).

Appropriate time-scales for funding

In many regions, funding for specific control operations is of short duration (2–3 years at most), supporting initial control but not the essential follow-ups. This is particularly important for wattle invasions where the long-lived persistent seed banks are a major limitation to successful management (Box 2, Table 3). Indeed the time taken for eradication, unless new methods are developed, is effectively equal to the seed dormancy time period, i.e. decades for many species. Funding for the control of an invasive wattle needs to be assured in the medium- to long-term; otherwise most of the management efforts described here will be ineffective.

CONCLUSIONS

The problems caused by invasive Australian acacias are already substantial, but will increase in magnitude and diversity. There are many climatically suitable regions around the world where Australian acacias have not yet been introduced (Richardson *et al.*, 2011), but as countries and individuals seek to replicate the diverse beneficial uses of species demonstrated in other parts of the world (Kull *et al.*, 2011; van Wilgen *et al.*, 2011), and develop wattles to fulfil new needs (Griffin *et al.*, 2011), we expect wattles to continue to be distributed. As shown in the papers in this special issue of *Diversity and Distributions*, we have a reasonable understanding of the determinants and impacts of invasiveness in this group and the considerable difficulties managing established invasions. There are few documented examples where cultivation has not led eventually to an invasion, and so the prior expectation is that large-scale introductions to climatically suitable areas will lead to an invasion with the potential to transform ecosystems. Therefore, a precautionary approach should be taken, and more should be done pro-actively to limit invasions in terms of both management and research (Puth & Post, 2005).

In conclusion, our recommendations are as follows: (1) All new introductions should be contingent upon full and detailed risk assessments and cost-benefit analyses, and while perceived benefits might override ecological concerns in some cases, we would recommend the development and adoption of sustainable mitigation methods. (2) Eradication should be set as a management goal more often, particularly if there are no conflicts of interest and species have limited invasive distributions, or pre-emptively if a species is known to be invasive elsewhere. (3) Commercial plantings should carry the costs for the increased risk of invasions, production should focus on sterile cultivars, and responsible utilization practices should be developed and implemented. (4) Biological control remains the most cost effective, sustainable and reliable option for widespread invasive Australian acacias (Dennill *et al.*, 1999), and efforts should be made to implement it more widely. (5) Effective management of Australian acacia invasions will require better global co-operation, dissemination of information and experience, and an increase in public awareness. While the risks and appropriate responses will vary depending on the context, we believe following these general recommendations will help reduce the costs of Australian acacia invasions.

Whether our group-specific global approach was useful in clarifying what needs to be done and where efforts should be focussed, or has produced over-general impractical recommendations with no-one to implement them and that could exclude local approaches to particular socioeconomical and ecological contexts, remains to be seen. However, the management tools and approaches described here can be applied to other functionally similar groups. Five of Australia's 20 Weeds of National Significance are woody legumes: *Acacia nilotica*; *Prosopis* spp., *M. pigra*, *Parkinsonia aculeata* and *Ulex europaeus*. Mirroring efforts to control Australian acacias elsewhere, Australia has tried to prevent the introduction of woody legumes (Paynter *et al.*, 2003), eradicate them [e.g. *A. angustissima* and *A. karroo* (Csurhes & Navie 2009)], and has introduced several biological control agents to reduce their populations (Palmer *et al.*, 2010). Can the group-specific approach taken here be extended to other types of invasion to provide useful generalizations [e.g. how does the risk profile vary for different vines (Harris *et al.*, 2007)]? We believe that the replicated biogeographical experiment of Australian acacia introductions serves both as a model for how to manage a widely utilized group of potentially invasive species and offers an opportunity to learn whether management generalizations are worth making.

ACKNOWLEDGEMENTS

This paper greatly benefited from discussions during the workshop on 'Human-mediated introductions of Australian acacias – a global experiment in biogeography' held in Stellenbosch, South Africa, in October 2010. Funding for the workshop was provided by the Oppenheimer Memorial Trust and Stellenbosch University. Additional support was provided by the DST-NRF Centre of Excellence for Invasion Biology and the Working for Water programme through their collaborative project on 'Research for Integrated Management of Invasive Alien Species', and from the Working for Water/SANBI Early Detection and Rapid Response Programme. Pilar Castro-Díez, Charalambos Christodoulou, Pinelopi Delipetrou, Chris Harwood, Rod Griffin, Pierre Ehret, Stephen Midgley, Tony Rinaudo and Joe Miller provided useful discussion, comments and/or completed the questionnaire. The following authors are grateful for support with workshop participation: HM for FCT-MCTES, grant SFRH/BD/24987/2005; JM for ACERA; CG for FCT-MCTES, grant SFRH/BPD/29878/2006. EM was supported by FCT-MCTES, grant SFRH/BPD/63211/2009.

REFERENCES

Andreu, J., Vilà, M. & Hulme, P.E. (2009) An assessment of stakeholder perceptions and management of noxious alien plants in Spain. *Environmental Management*, **43**, 1244–1255.

Blackburn, T.M., Pyšek, P., Bacher, S., Carlton, J.T., Duncan, R.P., Jarošík, V., Wilson, J.R.U. & Richardson, D.M. (2011) A proposed unified framework for biological invasions. *Trends in Ecology and Evolution*, **26**, 333.

Brockwell, J., Searle, S.D., Jeavons, A.C. & Waayers, M. (2005) *Nitrogen fixation in acacias: an untapped resource for sustainable plantations, farm forestry and land reclamation*. ACIAR Monograph No. 115. Australian Centre for International Agricultural Research, Canberra.

Brunel, S., Branquart, E., Fried, G., van Valkenburg, J., Brundu, G., Starfinger, U., Buholzer, S., Uludag, A., Joseffson, M. & Baker, R. (2010) The EPPO prioritization process for invasive alien plants. *EPPO Bulletin*, **40**, 407–422.

Caley, P., Groves, R.H. & Barker, R. (2008) Estimating the invasion success of introduced plants. *Diversity and Distributions*, **14**, 196–203.

Carruthers, J., Robin, L., Hattingh, J., Kull, C., Rangan, H. & van Wilgen, B.W. (2011) A native at home and abroad: the history, politics, ethics and aesthetics of acacias. *Diversity and Distributions*, **17**, 810–821.

Castro-Díez, P., Godoy, O., Saldaña, A. & Richardson, D.M. (2011) Predicting invasiveness of Australian acacias on the basis of their native climatic affinities, life-history traits and human use. *Diversity and Distributions*, **17**, 934–945.

Cohen, O., Riov, J., Katan, J., Gamliel, A. & Bar, P. (2008) Reducing persistent seed banks of invasive plants by soil solarization – the case of *Acacia saligna*. *Weed Science*, **56**, 860–865.

Common, I.F.B. (1990) *Moths of Australia*. Melbourne University Press, Melbourne.

Cossalter, C. (1986) Introduction of Australian acacias into dry tropical West Africa. *Forest Ecology and Management*, **16**, 367–389.

Csurhes, S. & Navie, S. (2009) *Weed risk assessment: White ball acacia, Acaciella angustissima, Queensland Government Report number PR09-4478, Biosecurity Queensland*. Queensland Primary Industries and Fisheries, Brisbane, Australia, 23.

Daehler, C.C. (2009) Short lag times for invasive tropical plants: evidence from experimental plantings in Hawai'i. *PLoS ONE*, **4**, e4462, doi: 4410.1371/journal.pone.0004462.

David, J.H.M. & Jarvis, J.U.M. (1985) Population fluctuations, reproduction and survival in the striped fieldmouse *Rhabdomys pumilo* on the Cape Flats, South Africa. *Journal of Zoology*, **207**, 251–276.

De Poorter, M. (2001) Perception and ''human nature'' as factors in invasive alien species issues: a workshop wrap-up on problems and solutions. *The great reshuffling: human dimensions of invasive alien species* (ed. by J.A. McNeely), pp. 209–213. IUCN, Cambridge, UK.

Dennill, G.B. (1988) Why a gall former can be a good biocontrol agent—the gall wasp *Trichilogaster acaciaelongifoliae* and the weed *Acacia longifolia*. *Ecological Entomology*, **13**, 1–9.

Dennill, G.B. & Donnelly, D. (1991) Biological control of *Acacia longifolia* and related weed species (*Fabaceae*) in South Africa. *Agriculture Ecosystems and Environment*, **37**, 115–135.

Dennill, G.B., Donnelly, D., Stewart, K. & Impson, F.A.C. (1999) Insect agents used for the biological control of Australian *Acacia* species and *Paraserianthes lophantha* (Willd.) Nielsen (*Fabaceae*) in South Africa. *African Entomology*, **1**, 45–54.

Derraik, J.G.B. & Phillips, S. (2010) Online trade poses a threat to biosecurity in New Zealand. *Biological Invasions*, **12**, 1477–1480.

DiTomaso, J.M., Reaser, J.K., Dionigi, C.P., Doering, O.C., Chilton, E., Schardt, J.D. & Barney, J.N. (2010) Biofuel vs bioinvasion: seeding policy priorities. *Environmental Science and Technology*, **44**, 6906–6910.

Dorchin, N., Cramer, M.D. & Hoffmann, J.H. (2006) Photosynthesis and sink activity of wasp-induced galls in *Acacia pycnantha*. *Ecology*, **87**, 1781–1791.

Ducatillion, C. & Blanc-Chabaud, L. (2010) *L'art d'acclimater les plantes exotiques: le jardin de la villa Thuret, Paris*. Éditeur Quae, Paris.

Dufour-Dror, J.M. (2010) *Alien invasive plants in Israel*. Ahva Publishing, Jerusalem.

Flora of Australia Online (2011) Available at: http://www.environment.gov.au/biodiversity/abrs/online-resources/flora/main/index.html (accessed 27 January 2011). Australian Biological Resources Study, Canberra.

Gaertner, M., Den Breeyen, A., Hui, C. & Richardson, D.M. (2009) Impacts of alien plant invasions on species richness in Mediterranean-type ecosystems: a meta-analysis. *Progress in Physical Geography*, **33**, 319–338.

Gibson, M.R., Richardson, D.M., Marchante, E., Marchante, H., Rodger, J., Stone, G., Wandrag, E., Fuentes-Ramirez, A., Harris, C., Johnson, S.D., Murphy, D., Pauchard, A., Pauw, A. & Prescott, M.N. (2011) Reproductive biology of Australian acacias: important mediator of invasiveness? *Diversity and Distributions*, **17**, 911–933.

Goodwin, K.M., Engel, R.E. & Weaver, D.K. (2010) Trained dogs outperform human surveyors in the detection of rare spotted knapweed (*Centaurea stoebe*). *Invasive Plant Science and Management*, **3**, 113–121.

Gordon, D.R., Mitterdorfer, B., Pheloung, P.C., Ansari, S., Buddenhagen, C., Chimera, C., Daehler, C.C., Dawson, W., Denslow, J.S., LaRosa, A., Nishidal, T., Onderdonk, D.A., Panetta, F.D., Pyšek, P., Randall, R.P., Richardson, D.M., Tshidada, N.J., Virtue, J.G. & Williams, P.A. (2010) Guidance for addressing the Australian Weed Risk Assessment questions. *Plant Protection Quarterly*, **25**, 56–74.

Griffin, A.R. (1990) Effects of Inbreeding on growth of forest trees and implications for management of seed supplies for plantation programs. *Reproductive ecology of tropical forest plants* (ed. by K.S. Bawa and M. Hadley), pp. 362–382. Man & Biosphere Series. UNESCO, Paris.

Griffin, A.R., Midgley, S., Bush, D., Cunningham, P. & Rinaudo, A. (2011) Global plantings and utilisation of Australian acacias—past, present and future. *Diversity and Distributions*, **17**, 837–847.

Groves, R.H., Boden, R. & Lonsdale, W.M. (2005) *Jumping the garden fence: invasive garden plants in Australia and their environmental and agricultural impacts*. CSIRO, WWF-Australia, Sydney.

Harris, C.J., Murray, B.R., Hose, G.C. & Hamilton, M.A. (2007) Introduction history and invasion success in exotic vines introduced to Australia. *Diversity and Distributions*, **13**, 467–475.

Haysom, K.A. & Murphy, S.T. (2003) *The status of invasiveness of forest tree species outside their natural habitat: a global review and discussion paper*. Forest Health and Biosecurity Working Paper FBS/3E. FAO, Rome.

Hill, R.L., Gordon, A.J. & Neser, S. (2000) The potential role of *Bruchophagus acaciae* (Cameron) (*Hymenoptera*: *Eurytomidae*) in the integrated control of *Acacia* species in South Africa. *Proceedings of the X international symposium on biological control of weeds* (ed. by N.R. Spencer), pp. 919–929. Montana State University, USDA ARS, Bozeman, Montana, USA.

Hoffmann, J.H., Impson, F.A.C., Moran, V.C. & Donnelly, D. (2002) Biological control of invasive golden wattle trees (*Acacia pycnantha*) by a gall wasp, *Trichilogaster* sp. (Hymenoptera: Pteromalidae), in South Africa. *Biological Control*, **25**, 64–73.

Holmes, P.M. (1989a) Decay-rates in buried alien acacia seed populations of different density. *South African Journal of Botany*, **55**, 299–303.

Holmes, P.M. (1989b) Effects of different clearing treatments of the seed-bank dynamics of an invasive Australian shrub, *Acacia cyclops*, in the South-western Cape, South Africa. *Forest Ecology and Management*, **28**, 33–46.

Holmes, P.M. (1990a) Dispersal and predation in alien acacia. *Oecologia*, **83**, 288–290.

Holmes, P.M. (1990b) Dispersal and predation of alien acacia seeds—effects of season and invading stand density. *South African Journal of Botany*, **56**, 428–434.

Holmes, P.M. & Rebelo, A.G. (1988) The occurrence of seed feeding *Zulubius acaciaphagus* (*Hemiptera*, *Alydidae*) and its effects on *Acacia cyclops* seed-germination and seed banks in South Africa. *South African Journal of Botany*, **54**, 319–324.

Holmes, P.M., Macdonald, I.A.W. & Juritz, J. (1987) Effects of clearing treatment on seed banks of the alien invasive shrubs *Acacia saligna* and *Acacia cyclops* in the Southern and Southwestern Cape, South Africa. *Journal of Applied Ecology*, **24**, 1045–1051.

Hui, C., Richardson, D.M., Robertson, M.P., Yates, C.J. & Wilson, J.R.U. (2011) Macroecology meets invasion ecology: linking native distribution of Australian acacias to invasiveness. *Diversity and Distributions*, **17**, 872–883.

Hulme, P.E., Pyšek, P., Nentwig, W. & Vilá, M. (2009) Will threat of biological invasions unite the European Union? *Science*, **324**, 40–41.

Impson, F., Hoffmann, J.H. & Kleinjan, C. (2009) Australian *Acacia* species (*Mimosaceae*) in South Africa. *Biological control of tropical weeds using arthropods* (ed. by R. Muniappan, G.V.P. Reddy and A. Raman), pp. 38–62. Cambridge University Press, Cambridge.

Impson, F.A.C., Kleinjan, C.A., Hoffmann, J.H., Post, J.A. & Wood, A.R. (2011) Biological control of Australian *Acacia* species and *Paraserianthes lophantha* (Willd.) Nielsen (Mimosaceae) in South Africa. *African Entomology*, **19**, in press.

Jeddi, K., Cortina, J. & Chaieb, M. (2009) *Acacia salicina*, *Pinus halepensis* and *Eucalyptus occidentalis* improve soil surface conditions in arid southern Tunisia. *Journal of Arid Environments*, **73**, 1005–1013.

Kolesik, P., Adair, R.J. & Eick, G. (2010) Six new species of *Asphondylia* (*Diptera: Cecidomyiidae*) damaging flower buds and fruit of Australian *Acacia* (*Mimosaceae*). *Systematic Entomology*, **35**, 250–267.

Kraus, F. & Duffy, D.C. (2010) A successful model from Hawaii for rapid response to invasive species. *Journal for Nature Conservation*, **18**, 135–141.

Kueffer, C. & Vos, P. (2004) *Case studies on the status of invasive woody plant species in the Western Indian Ocean: 5. Seychelles*. Forestry Department, Food and Agriculture Organization of the United Nations, Rome, Italy.

Kull, C.A., Tassin, J. & Rangan, H. (2007) Multifunctional, scrubby, and invasive forests? Wattles in the highlands of Madagascar. *Mountain Research and Development*, **27**, 224–231.

Kull, C.A., Tassin, J., Rambeloarisoa, G. & Sarrailh, J.M. (2008) Invasive Australian acacias on western Indian Ocean islands: a historical and ecological perspective. *African Journal of Ecology*, **46**, 684–689.

Kull, C.A., Shackleton, C.M., Cunningham, P. *et al.* (2011) Adoption, use and perception of Australian acacias around the world. *Diversity and Distributions*, **17**, 822–836.

Le Maitre, D.C. (2004) Predicting invasive species impacts on hydrological processes: the consequences of plant physiology for landscape processes. *Weed Technology*, **18**, 1408–1410.

Le Maitre, D.C., Gaertner, M., Marchante, E., Ens, E.-J., Holmes, P.M., Pauchard, A., O'Farrell, P.J., Rogers, A.M., Blanchard, R., Blignaut, J. & Richardson, D.M. (2011) Impacts of invasive Australian acacias: implications for management and restoration. *Diversity and Distributions*, **17**, 1015–1029.

Mack, R.N. & Lonsdale, W.M. (2002) Eradicating invasive plants: hard-won lessons for islands. *Turning the tide: the eradication of invasive species* (ed. by C.R. Veitch and M.N. Clout), pp. 164–172. SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland.

Magee, T.K., Ringold, P.L., Bollman, M.A. & Ernst, T.L. (2010) Index of alien impact: a method for evaluating potential ecological impact of alien plant species. *Environmental Management*, **45**, 759–778.

Marchante, E., Kjoller, A., Struwe, S. & Freitas, H. (2008) Short- and long-term impacts of *Acacia longifolia* invasion on the belowground processes of a Mediterranean coastal dune ecosystem. *Applied Soil Ecology*, **40**, 210–217.

Marchante, E., Marchante, H., Morais, M. & Freitas, H. (2010a) Combining methodologies to increase public awareness about invasive plants in Portugal. 2nd International workshop on invasive plants in Mediterranean type regions of the world, Trabzon, Turkey.

Marchante, H., Freitas, H. & Hoffmann, J.H. (2010b) Seed ecology of an invasive alien species, *Acacia longifolia* (*Fabaceae*), in Portuguese dune ecosystems. *American Journal of Botany*, **97**, 1–11.

Marchante, H., Freitas, H. & Hoffmann, J.H. (2011) Assessing the suitability and safety of a well-known bud-galling wasp, *Trichilogaster acaciaelongifoliae*, for biological control of *Acacia longifolia* in Portugal. *Biological Control*, **56**, 193–201.

Maslin, B.R. & McDonald, M.W. (2004) *AcaciaSearch: evaluation of acacia as a woody crop option for southern Australia*. Rural Industries and Research Development Corporation. Publication No. 03/017, RIRDC, Canberra.

McKeown, K.C. (1947) Catalogue of the *Cerambycidae* (*Coleoptera*) of Australia. *Australian Museum Memoir, Australian Museum, Sydney*, **10**, 1–199.

Miller, J., Murphy, D.J., Brown, G.K., Richardson, D.M. & González-Orozco, C.E. (2011) The evolution and phylogenetic placement of invasive Australian *Acacia* species. *Diversity and Distributions*, **17**, 848–860.

Milton, S.J. & Hall, A.V. (1981) Reproductive biology of Australian acacias in the south-western Cape Province, South Africa. *Transactions of the Royal Society of South Africa*, **44**, 465–485.

Mochiutti, S., Higa, A.R. & Simon, A.A. (2007) Susceptibilidade de ambientes campestres à invasão de acácia-negra (*Acacia mearnsii* De Wild.) no Rio Grande do Sul. *Floresta*, **37**, 239–253.

Moore, J.L., Runge, M.C., Webber, B.L. & Wilson, J.R.U. (2011) Contain or eradicate? Optimising the goal of managing Australian acacia invasions in the face of uncertainty. *Diversity and Distributions*, **17**, 1047–1059.

Morris, M.J. (1997) Impact of the gall-forming rust fungus *Uromycladium tepperianum* on the invasive tree *Acacia saligna* in South Africa. *Biological Control*, **10**, 75–82.

Morris, T.L., Esler, K.J., Barger, N.N., Jacobs, S.M. & Cramer, M.D. (2011) Ecophysiological traits associated with the competitive ability of invasive Australian acacias. *Diversity and Distributions*, **17**, 898–910.

Moseley, C.T., Cramer, M.D., Kleinjan, C.A. & Hoffmann, J.H. (2009) Why does *Dasineura dielsi*-induced galling of *Acacia cyclops* not impede vegetative growth? *Journal of Applied Ecology*, **46**, 214–222.

Murphy, D.J., Brown, G.K., Miller, J.T. & Ladiges, P.Y. (2010) Molecular phylogeny of *Acacia* Mill. (Mimosoideae: Leguminosae): evidence for major clades and informal classification. *Taxon*, **59**, 7–19.

O'Dowd, D.J. & Gill, A.M. (1986) Seed dispersal syndromes of Australian *Acacia*. *Seed dispersal* (ed. by D. Murray), pp. 87–121. Academic Press, New York.

Palmer, W.A., Heard, T.A. & Sheppard, A.W. (2010) A review of Australian classical biological control of weeds programs and research activities over the past 12 years. *Biological Control*, **52**, 271–287.

Panetta, F.D. (2007) Evaluation of weed eradication programs: containment and extirpation. *Diversity and Distributions*, **13**, 33–41.

Panetta, F.D. (2009) Weed eradication—an economic perspective. *Invasive Plant Science and Management*, **2**, 360–368.

Paynter, Q. (2005) Evaluating the impact of a biological control agent *Carmenta mimosa* on the woody wetland weed *Mimosa pigra* in Australia. *Journal of Applied Ecology*, **42**, 1054–1062.

Paynter, Q., Csurhes, S.M., Heard, T.A., Ireson, J., Julien, M.H., Lloyd, J., Lonsdale, W.M., Palmer, W.A., Sheppard, A.W. & van Klinken, R.D. (2003) Worth the risk? Introduction of legumes can cause more harm than good: an Australian perspective. *Australian Systematic Botany*, **16**, 81–88.

Pheloung, P.C., Williams, P.A. & Halloy, S.R. (1999) A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. *Journal of Environmental Management*, **57**, 239–251.

PIER (2003) *Pacific Island ecosystems at risk*. Available at: http://www.hear.org/pier/index.html (accessed 1 November 2010).

Pieterse, P.J. & Cairns, A.L.P. (1986) The effect of fire on an *Acacia longifolia* seed bank in the Southwestern Cape. *South African Journal of Botany*, **52**, 233–236.

Pieterse, P.J. & Cairns, A.L.P. (1988) Factors affecting the reproductive success of *Acacia longifolia* (Andr) Willd. in the Banhoek Valley, South-western Cape, Republic of South Africa. *South African Journal of Botany*, **54**, 461–464.

Poynton, R.J. (2009) *Tree planting in southern Africa: vol. 3 other genera*. Department of Agriculture, Forestry, and Fisheries, Pretoria, South Africa.

Procheş, Ş., Wilson, J.R.U., Richardson, D.M. & Rejmánek, M. (2011) Native and naturalised range size in *Pinus*: relative importance of biogeography, introduction effort and species traits. *Global Ecology and Biogeography*, **20**, in press doi: 10.1111/j.1466-8238.2011.00703.x.

Puth, L.M. & Post, D.M. (2005) Studying invasion: have we missed the boat? *Ecology Letters*, **8**, 715–721.

Pyšek, P., Richardson, D.M., Rejmánek, M., Webster, G.L., Williamson, M. & Kirschner, J. (2004) Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. *Taxon*, **53**, 131–143.

Ratnayake, K. & Joyce, D. (2010) Native Australian Acacias: Unrealised ornamental potential. *Chronica Horticulturae*, **50**, 19–22.

Reid, J.C. & Murphy, D.J. (2008) Some case studies of *Acacia* as weeds and implications for herbaria. *Muelleria*, **26**, 57–66.

Reubens, B., Moeremans, C., Poesen, J., Nyssen, J., Tewoldeberhan, S., Franzel, S., Deckers, J., Orwa, C. & Muys, B. (2011) Tree species selection for land rehabilitation in Ethiopia: from fragmented knowledge to an integrated multi-criteria decision approach. *Agroforestry Systems*, **82**, 303–330, doi 10.1007/s10457-011-9381-8.

Richardson, D.M. & Kluge, R.L. (2008) Seed banks of invasive Australian Acacia species in South Africa: role in invasiveness and options for management. *Perspectives in Plant Ecology Evolution and Systematics*, **10**, 161–177.

Richardson, D.M. & Rejmánek, M. (2004) Conifers as invasive aliens: a global survey and predictive framework. *Diversity and Distributions*, **10**, 321–331.

Richardson, D.M. & Rejmánek, M. (2011) Trees and shrubs as invasive species - a global review. *Diversity and Distributions*, **17**, 788–809.

Richardson, D.M. & van Wilgen, B.W. (2004) Invasive alien plants in South Africa: how well do we understand the ecological impacts? *South African Journal of Science*, **100**, 45–52.

Richardson, D.M., Allsopp, N., D'Antonio, C.M., Milton, S.J. & Rejmánek, M. (2000a) Plant invasions – the role of mutualisms. *Biological Reviews*, **75**, 65–93.

Richardson, D.M., Pyšek, P., Rejmánek, M., Barbour, M.G., Panetta, F.D. & West, C.J. (2000b) Naturalization and invasion of alien plants: concepts and definitions. *Diversity and Distributions*, **6**, 93–107.

Richardson, D.M., van Wilgen, B.W. & Nunez, M.A. (2008) Alien conifer invasions in South America: short fuse burning? *Biological Invasions*, **10**, 573–577.

Richardson, D.M., Carruthers, J., Hui, C., Impson, F.A.C., Miller, J., Robertson, M.P., Rouget, M., le Roux, J.J. & Wilson, J.R.U. (2011) Human-mediated introductions of Australian acacias—a global experiment in biogeography. *Diversity and Distributions*, **17**, 771–787.

Rinaudo, A. & Cunningham, P. (2008) Australian acacias as multi-purpose agro-forestry species for semi-arid regions of Africa. *Muelleria*, **26**, 79–85.

Rinaudo, A., Patel, P. & Thomson, L.A.J. (2002) Potential of Australian acacias in combating hunger in semi-arid lands. *Conservation Science Western Australia*, **4**, 161–169.

Rodríguez-Echeverría, S. (2010) Rhizobial hitchhikers from Down Under: invasional meltdown in a plant-bacteria mutualism? *Journal of Biogeography*, **37**, 1611–1622.

Rodríguez-Echeverría, S., Le Roux, J.J., Crisóstomo, J.A. & Ndlovu, J. (2011) Jack-of-all-trades and master of many? How does associated rhizobial diversity influence the colonization success of Australian *Acacia* species? *Diversity and Distributions*, **17**, 946–957.

Roura-Pascual, N., Richardson, D.M., Krug, R.M., Brown, A., Chapman, R.A., Forsyth, G.G., Le Maitre, D.C., Robertson, M.P., Stafford, L., Van Wilgen, B.W., Wannenburgh, A. & Wessels, N. (2009) Ecology and management of alien plant invasions in South African fynbos: accommodating key complexities in objective decision making. *Biological Conservation*, **142**, 1595–1604.

Seabloom, E.W., Williams, J.W., Slayback, D., Stoms, D.M., Viers, J.H. & Dobson, A.P. (2006) Human impacts, plant invasion, and imperiled plant species in California. *Ecological Applications*, **16**, 1338–1350.

Simberloff, D. (2009) We can eliminate invasions or live with them. Successful management projects. *Biological Invasions*, **11**, 149–157.

Simberloff, D., Nunez, M.A., Ledgard, N.J., Pauchard, A., Richardson, D.M., Sarasola, M., Van Wilgen, B.W., Zalba, S.M., Zenni, R.D., Bustamante, R., Pena, E. & Ziller, S.R. (2010) Spread and impact of introduced conifers in South America: lessons from other southern hemisphere regions. *Austral Ecology*, **35**, 489–504.

Southern African Plant Invaders Atlas (2009) Available at: http://www.agis.agric.za/wip/ (accessed 21 February 2011), ARC-Plant Protection Research Institute, Pretoria.

Tassin, J. & Balent, G. (2004) Le diagnostic d'invasion d'une essence forestière en milieu rural: exemple d'*Acacia mearnsii* à La Réunion. *Revue Forestière Française*, **56**, 132–142.

Tassin, J., Rivière, J.N., Cazanove, M. & Bruzzese, E. (2006) Ranking of invasive woody plant species for management on Réunion Island. *Weed Research*, **46**, 388–403.

van Wilgen, B.W. & Richardson, D.M. (1985) The effects of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands—a simulation study. *Journal of Applied Ecology*, **22**, 955–966.

van Wilgen, B.W., Dyer, C., Hoffmann, J.H., Ivey, P., Le Maitre, D.C., Richardson, D.M., Rouget, M., Wannenburgh, A. & Wilson, J.R.U. (2011) National-scale strategic approaches for managing introduced plants: insights from Australian acacias in South Africa. *Diversity and Distributions*, **17**, 1060–1075.

Williamson, M. (1999) Invasions. *Ecography*, **22**, 5–12.

Wilson, J.R.U., Richardson, D.M., Rouget, M., Procheş, Ş., Amis, M.A., Henderson, L. & Thuiller, W. (2007) Residence time and potential range: crucial considerations in modelling plant invasions. *Diversity and Distributions*, **13**, 11–22.

Wilson, J.R.U., Dormontt, E.E., Prentis, P.J., Lowe, A.J. & Richardson, D.M. (2009) Something in the way you move: dispersal pathways affect invasion success. *Trends in Ecology and Evolution*, **24**, 136–144.

Witkowski, E.T.F. (1994) Growth of seedlings of the invasives, *Acacia saligna* and *Acacia cyclops*, in relation to soil-phosphorus. *Australian Journal of Ecology*, **19**, 290–296.

Wood, A.R. & Morris, M.J. (2007) Impact of the gall-forming rust fungus *Uromycladium tepperianum* on the invasive tree *Acacia saligna* in South Africa: 15 years of monitoring. *Biological Control*, **41**, 68–77.

Zenni, R., Wilson, J.R.U., Le Roux, J.J. & Richardson, D.M. (2009) Evaluating the invasiveness of *Acacia paradoxa* in South Africa. *South African Journal of Botany*, **75**, 485–496.

SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article:

Appendix S1 Questionnaire on management options for invasive Australian acacias.

Appendix S2 Questionnaire results showing geographic spread of Australian acacias.

Appendix S3 Questionnaire results showing management efforts of different countries against invasive Australian acacias.

Appendix S4 Assessment of Australian acacias under the Australian Weed Risk Assessment Protocol.

As a service to our authors and readers, this journal provides supporting information supplied by the authors. Such materials are peer-reviewed and may be re-organized for online delivery, but are not copy-edited or typeset. Technical support issues arising from supporting information (other than missing files) should be addressed to the authors.

BIOSKETCH

All the authors are interested in understanding natural resource management better, and how Australian acacias fit into this. **John Wilson** works on biological invasions for the South African National Biodiversity Institute. He is interested in how and why humans move species around, and understanding the applied consequences of the invasion process.

Author contributions: J.R.U.W. and D.M.R. conceived the study; C.G., M.R.G. and J.R.U.W. conducted the questionnaire, J.R.U.W. and D.J.M. completed the risk assessment, and all authors contributed to the writing and provided answers to the questionnaire.

Editor: Mathieu Rouget