Biol Invasions (2014) 16:691–703
DOI 10.1007/s10530-013-0611-z

ORIGINAL PAPER

Incorporating risk mapping at multiple spatial scales into eradication management plans

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Received: 23 November 2012 / Accepted: 16 September 2013 / Published online: 19 December 2013

Abstract The success of pro-active management of invasive plants depends on the ability to rapidly detect invasive populations and individuals. However, the factors important for detection depend on the spatial scale examined. We propose a protocol for developing risk maps at national, landscape, and local scales to improve detection rates of invasive plant species. We test this approach in the context of developing an eradication plan for the invasive tree *Acacia stricta* in South Africa. At a national scale we used bioclimatic models coupled with the most likely sites of introduction (i.e. forestry nursery plantations) to identify areas where national-scale surveillance should be focussed. At the landscape and local scales we correlated the presence of *A. stricta* populations to various attributes. Regional populations were found in forestry plantations only, and mostly on highly used graded roads along which seeds are spread by road maintenance vehicles. Locally, previously recorded plant localities accurately predicted individuals in subsequent surveys. Using these variables, we produced a map of high-risk areas that facilitated targeted searches—which reduced the required search effort by ca. 83 %—and developed recommendations for site-specific surveying. With the high visibility of plants, and relatively small seed banks, long-term annual clearing should achieve eradication. We propose that such multi-scale risk mapping is valuable for prioritising management and surveillance efforts, though caution that the approach is correlative and so it does not represent all the sites that can be invaded.

Keywords Biological invasions · Early detection · Eradication · Invasive plant · Risk mapping · Surveillance · Tree invasions

Electronic supplementary material The online version of this article (doi:10.1007/s10530-013-0611-z) contains supplementary material, which is available to authorized users.

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Introduction

Management of invasive species is most effective when invasions are detected early and comprehensive control measures are implemented rapidly enough to prevent widespread impacts from accruing

(Simberloff 2003b). In this context, eradication is often a desirable goal (Myers et al. 2000). However, because attempting eradication is usually expensive, with a high probability of failure (Gardener et al. 2010; Panetta 2009; Simberloff 2003a), thorough evaluation of the feasibility of eradication is crucial. Eradication of plant species is usually only considered feasible for species with small range sizes (Rejmánek and Pitcairn 2002). However, the success of eradication is not only influenced by range size, but also on the ability to find all propagules. As a species' invasive range increases, there is a greater need to accurately delimit the extent of the invasion (i.e. number and size of populations) to determine eradication feasibility (Moore et al. 2011; Panetta and Lawes 2005). Quantifying the risk of post-introduction spread via particular vectors, along different pathways and into given habitats and ecosystems has become a strong focus of research in invasion science in the last two decades (e. g. Pyšek and Richardson 2010).

The costs and effort required to search for a species over a large potential range are often prohibitively high. Furthermore, there is a high probability of missing invasive populations using random searching (Cacho et al. 2006). Invasion delimitation therefore requires systematic search protocols that enable rapid delimitation of all invasive stands and sufficient surveillance to detect any new populations that result from spread. Consequently, identifying areas where search efforts should be focussed, based on probability of invasion success in those areas, can reduce costs and effort associated with delimiting the extent of an invasion.

Habitat suitability predictions have been used to identify vulnerable areas for invasions and predict spread pathways of invasive plant species in order to improve search and management strategies (Butcher and Kelly 2011; Giorgis et al. 2011; Peltzer et al. 2008; Vanderhoof et al. 2009). If eradication of plant species is to be attempted, early detection of new populations is essential. However, the probability of detecting new populations before they attain reproductive maturity is often low (Kery and Gregg 2003). In addition, species with long-lived seed banks may be present at a site but remain undetected until germination is stimulated (Cacho et al. 2007). Having a better idea of where to conduct intensive searches for a species could reduce the overall search effort and minimise the risk of undetected populations. Highlighting areas with high suitability or risk of invasion by a species will thus improve the efficiency of searching and enable early detection of populations before they increase in size and extent.

Understanding the pathways and vectors of dispersal for a species is an essential step in the development of a search protocol (Pyšek and Richardson 2010). However, different processes might act at different spatial scales. At a broad scale, it is mainly the pathways of introduction and potential agents of rare long distance dispersal (e.g. riparian dispersal) that need to be considered when trying to locate populations of a species. At local-scale seed dispersal vectors may prove more important in determining where to look for plants at a particular site. It is therefore important to understand the mechanisms driving both natural and anthropogenic dispersal at different scales in order to accurately detect and delimit an invasion.

The aim of this study is to develop such a multi-scale assessment and apply it to a case-study of an on-going eradication attempt. Here we propose a framework for improving the detection of naturalised plant populations for accurate delimitation of the species' extent (Fig. 1). Using *Acacia stricta* as a case study, we identify steps to rapidly and cost-effectively detect populations, thereby allowing an estimate of the feasibility of eradication. Using a risk mapping approach that includes bioclimatic and habitat suitability modelling, we apply predictions of potential range and spread pathways to target searches and awareness to areas of high risk of invasion at national, landscape and local scales in order to maximise detection of all invasive populations. We also consider the value of passive surveillance (i.e. reports from local land owners; Cacho et al. 2010) in locating and monitoring *A. stricta* populations. In addition, we identify reproductive traits (e.g. seed production, seed bank size) and dispersal mechanisms (i.e. vectors of seed spread) that may influence the feasibility of eradication.

Study system

Australian acacias have been proposed as a model system for studying biological invasions (Richardson et al. 2011). Given their substantial impacts and the difficulties in controlling extensive seed-banks, control of new acacia invasions should be proactive. To date, no Australian acacia species has been formally recorded as eradicated as part of any weed control

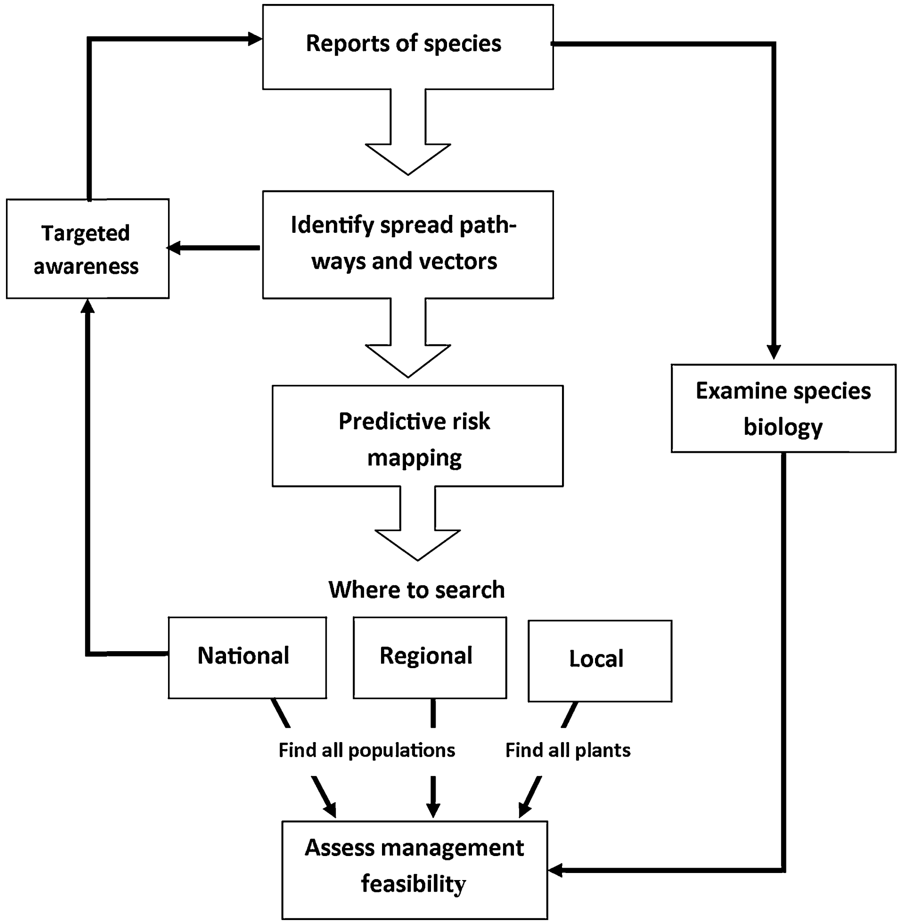

Fig. 1 Important steps in delimiting an invasion for the assessment of management feasibility

programme (Wilson et al. 2011). In South Africa there are several widespread invasive Australian acacia species that have had large-scale damaging impacts on local ecosystem services and biodiversity (Gaertner et al. 2009; Le Maitre et al. 2011). Besides these widespread invaders, there are a few Australian acacia species that have not yet become widespread, most likely because they were not highly utilized and thus not widely disseminated, or because of relatively short residence times. These species still exist in isolated populations, some of which may be suitable candidates for eradication (van Wilgen et al. 2011). Of these species, three are currently being targeted for eradication by the South Africa National Biodiversity Institute's Invasive Species Programme (Wilson et al. 2013). The first two, *A. paradoxa* and *A. implexa* are known to occur at only a few sites and eradication of these two species is considered feasible (Kaplan et al. 2012; Zenni et al. 2009). The third species, *A. stricta*, was reported from several sites (SAPIA; Henderson 1998) and as such requires further investigation to determine whether eradication is feasible.

Acacia stricta (Andrews) Willd. is a small tree native to south-eastern Australia. It is not known to be invasive elsewhere in the world, although it is recorded as naturalised in New Zealand (Richardson et al. 2011). Unlike most Australian acacia species that have been introduced to South Africa, there are no records of introduction or planting of *A. stricta* (Poynton 2009). Since 2004 it has been reported by local foresters and conservation managers as a problematic invader in the Knysna area of the Western Cape Province, South Africa.

Similar to the initial stages of many other invaders, *A. stricta* is currently found mostly in highly disturbed areas, particularly along roads. Roads have been shown to be major conduits for the spread of invasive species due to high levels of disturbance that promotes colonisation (Gelbard and Belnap 2003; Harrison et al. 2002; Spooner et al. 2004) and greater dispersal opportunities for seeds when road maintenance vehicles move soil (Ferguson et al. 2003; Mortensen et al. 2009; Taylor et al. 2012). The disjunct distribution of *A. stricta* and uncertainty of the overall distribution of the species in South Africa, together with good knowledge of Australian acacia invasions in general (Richardson et al. 2011), make this a suitable system to explore processes in determining management feasibility within the proposed framework.

Methods

National-scale assessment of likely sites of invasion

To identify areas where there is a high-risk of introduction and invasion, we considered two primary factors: climatic suitability and potential introduction pathways.

Climatic suitability

We modelled the potential distribution of *Acacia stricta* in South Africa (the extent of bioclimatically suitable conditions) using MAXENT 3.3.2 (Phillips et al. 2006). Presence data was compiled from 762 native range records of *A. stricta* from the Australian Virtual Herbarium (chah.gov.au/avh/; accessed 14 July 2010). The background for the model was drawn from *A. stricta*'s range in eastern Australia. The model was trained using all presence data. Duplicate records within each 5-min grid cell were deleted. Our modelling protocol followed that described by Thompson et al. (2011).

The climate variables used were the eight least inter-correlated bioclimatic variables from the WORLDCLIM dataset (www.worldclim.org, Hijmans et al. 2005): mean annual temperature, mean diurnal range in temperature, isothermality, temperature seasonality, mean annual precipitation, precipitation of the driest month, precipitation seasonality, and precipitation of the warmest quarter (Loiselle et al. 2008).

Model error, based on the predicted suitability, was estimated using a 10-fold cross-validation. The ability of the model to correctly predict actual occurrences was assessed using the average test area under curve (AUC).

Introduction pathways

Acacia stricta is not used for forestry purposes in Australia or anywhere else in the world. However, although there are no historical records of *A. stricta* being cultivated or used for forestry in South Africa (Glen 2002; Poynton 2009), both our field observations and opinions of local experts suggest that it was either intentionally or unintentionally (as a result of taxonomic misidentification) introduced along with other *Acacia* species by the forestry industry and was likely kept initially in forestry nurseries. We therefore overlaid areas of bioclimatically suitable conditions onto all plantations in the country in order to identify plantations within climatically suitable areas (Fig. 2a) that would be considered high risk for *A. stricta* introduction. These areas are being targeted for awareness campaigns (including the distribution of information leaflets; Supplementary Material Appendix A).

Delimiting invasions at a landscape scale

To detect populations at a landscape scale, we first consulted land-owners, forestry plantation managers, and regional conservation managers in the Knysna area (to date the only known area where *A. stricta* is invading in South Africa). Eight localities were identified and used as a starting point for surveying. The localities occurred within a total area of approximately 1,900 km^2 in the Knysna and Wilderness areas of the Garden Route National Park (Fig. 2b).

Vehicular surveys

In 2010, roads in the affected area were searched during the flowering period of *A. stricta* (August–September) when plants were most likely to be visible. A total of $\sim$523 unique km of roads were searched within the study area (approximately 20 % of the total road matrix). Searching was done from a vehicle driving at an average speed of $\sim$20 km/h with a driver and one observer. The study area has an extensive road network, and with no prior knowledge of where to search and insufficient time to survey every kilometre of road, vehicle surveys were directed based on accessibility and proximity to the localities identified as containing *A. stricta*.

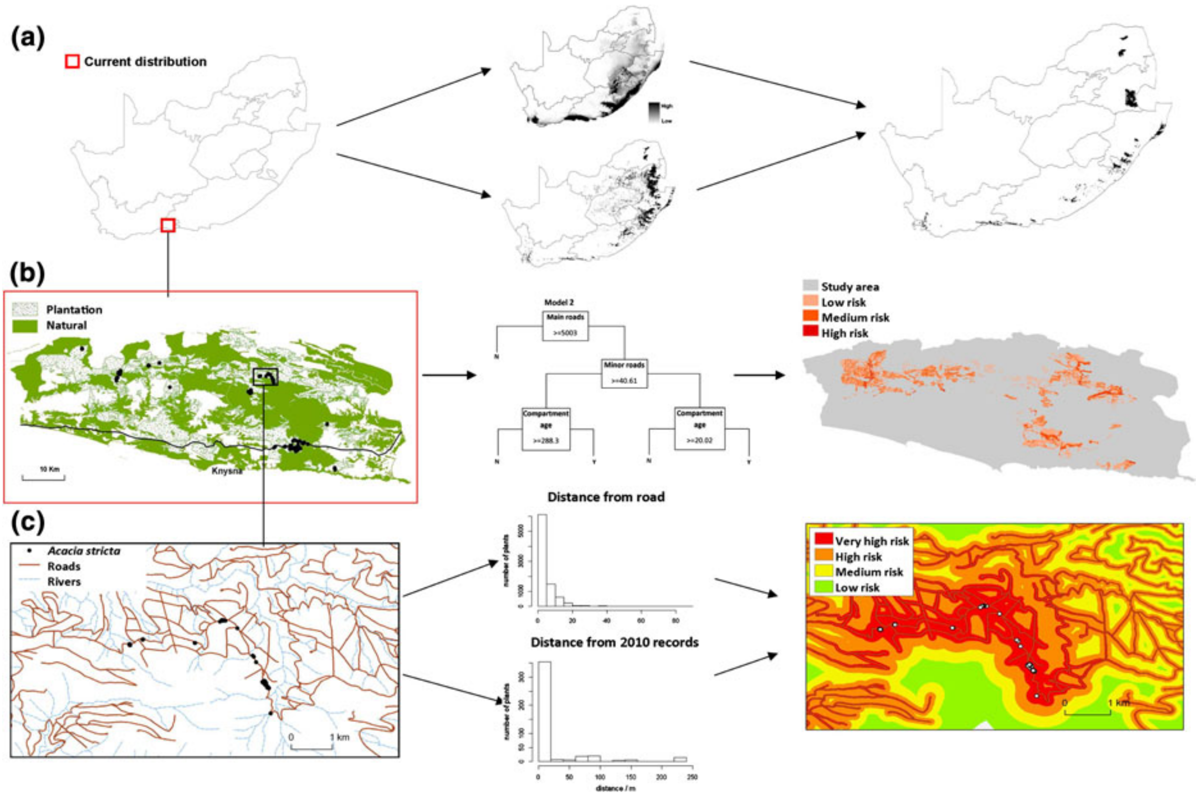

Fig. 2 Risk maps for *Acacia stricta* were produced to improve surveillance at a variety of spatial scales. At a national scale (**a**), a climate model and a map of forestry plantations were overlaid to determine where an awareness campaign should be directed; at a landscape scale (**b**), correlates of the known invaded distribution (distance to particular road type and compartment age) were used to determine similar areas that should be prioritised for future survey; and at a local scale (**c**), the position of plants found, in relation to distance to roads and locations of plants from the previous year, was used to inform criteria for which areas should be searched in detail (i.e. on foot)

Risk mapping (landscape level)

To more effectively direct search efforts for *A. stricta* surveillance at a landscape scale, we created a risk map. We based the model on presence data collected during surveys in 2010 and a set of 1,000 pseudo absence data from within the study area, randomly generated using ArcGIS 10.0. A second set of 1,000 absence points restricted to within 50 m of roadsides was generated to check for any bias towards roadsides that may have resulted from the pseudo-absence point selection. Not surprisingly, as all populations occurred in plantations (mostly pine), initial models returned land use (forestry) as the most significant variable (despite 40 % of the surveyed road lengths being non-plantations, see results). Consequently, in later models the area considered was restricted to plantations (roughly 23 % of the study site). To account for how different levels of spatial autocorrelation could affect the model results, presence and absence data were also sub-sampled using grid and random sampling at three spatial resolutions (Table 1).

To identify factors that could influence the spread of *A. stricta* we selected six abiotic and anthropogenic variables as possible predictors of *A. stricta* occurrence (Table 2). Variables were extracted at each sampled presence or absence point from various land cover and topographic map layers (10 m × 10 m resolution) in a geographical information system (GIS). Anthropogenic variables included were post-fire vegetation age (germination of Australian acacia seeds is known to be stimulated by fire for many species; Richardson and Kluge 2008), distance to roads [as roads can act as conduits for invasions (Gelbard and Belnap 2003; Mortensen et al. 2009), and disturbance from road maintenance can increase

Table 1 Point selection and spatial resolution of models used to predict the occurrence of *Acacia stricta*

	Model 1	Model 2	Model 3	Model 4
Spatial resolution	10 m × 10 m	500 m × 500 m	10 m × 10 m, but only one presence point used per population (selected at random)	500 m × 500 m Absence points restricted to <50 m from roadsides
Sample size	2,000 data points (1,000 presence/ 1,000 absence)	90 data points (45 presence/45 absence)	1,009 data points (9 presence/1,000 absence)	90 data points (45 presence/45 absence)

The study area was as defined by Fig. 2b, based on selection of entire tertiary catchments (from sea level to top of catchment) where *A. stricta* was present. Absence points were restricted to those cells where no plants were present

Table 2 Summary of variables used in classification tree models predicting the occurrence of *Acacia stricta*

Variable	Description	Values
Post-fire vegetation age	Number of years since last fire (bounded numeric)	Years from 0 to a maximum of >100
Elevation	Metres above mean sea level	0 to 1,184 m
Rainfall	Mean annual rainfall	679 to 1,061.2 mm
Compartment age	No. of years since planting or clear felling of plantation compartments	0 to 89 years
Distance from minor roads	Gradable roads, hiking trails, plantation access roads	0 to 1,462 m
Distance from main roads	National routes, main roads, secondary roads, urban streets	0 to 9,588 m

recruitment of *Acacia* species (Spooner et al. 2004)], and compartment age (disturbance and vehicle movement during plantation activity might aid in the dispersal and recruitment of *A. stricta*). Elevation and rainfall were highly variable across the study area and were included in the model to test whether these abiotic conditions influence *A. stricta* occurrence.

To test whether these variables influenced the observed distribution of *A. stricta*, we used classification trees (to minimise the influence of correlation between variables). Trees were drawn using recursive partitioning (package *rpart*) in R 2.11.0 (R Development Core Team 2010) for each of the three subsets of data. Trees were pruned to minimise the cross-validated prediction error estimates. Misclassification errors were calculated for each tree using a test dataset of 256 presence and pseudo absence points sampled at a 100 m resolution from the study area. Prediction accuracy was estimated based on the AUC value of each model. The best tree was projected back onto the study area in ArcGIS 10.0 to produce a probability map of *A. stricta* occurrence and highlight areas suitable for future spread and where search effort should be directed to detect invasive populations.

Local-scale detectability

To improve detectability at a finer scale we needed to know where to look for individual plants. To do this we conducted detailed search-and-destroy site surveys and mapped local-scale distribution patterns of the populations over two consecutive seasons (2010, 2011). Distributions were then assessed for possible dispersal pathways and vectors. The data were also used to assess whether the location of plants in year 1 could predict the location of plants in year 2 (given that seed production was prevented in year 1 this essentially gives an idea of the site fidelity regarding recruitment from seed-banks).

Every invaded site identified during the vehicle survey was searched on foot. Two surveyors walked survey lines parallel to the road ~10 m apart, such that at least 30 m each side of the road was searched. The location of each plant found and its distance to the road edge (to nearest 0.5 m) were recorded. Hiking trails that intersected roadside infestations were also surveyed. Searches were discontinued at a site when no plants were found for at least 250 m along the road in both directions.

Local-scale distribution and spread patterns were then assessed in relation to potential spread pathways. *Acacia stricta* seeds might be dispersed in soil movement along roads or be washed down watercourses that intersect the invaded sites, so roads and rivers were considered as potential vectors for *A. stricta*. We compared distances of plants to either potential vector as well as comparing locations of plants between surveys to estimate the amount of local spread that could occur in one year. Based on these observed associations we highlighted risk zones at a local scale where plants were most likely to be found and where more intensive searching is required away from the road edge.

Management planning

To assess the feasibility of eradicating *A. stricta* populations we investigated features of the species' reproductive biology and its response to management treatments used during the initial clearing. This provided an estimate of the costs and effort that would be required to eradicate the species.

Risk assessment

To date, *Acacia stricta* has not been assessed as an invasive species anywhere in the world. To collate relevant literature and to determine whether, as for most Australian acacias (Wilson et al. 2011), *A. stricta* would have failed a pre-border assessment we conducted an Australian Weed Risk Assessment (Pheloung et al. 1999), using the guidelines for applying the assessment scheme to areas outside Australia (Gordon et al. 2010).

Reproductive output

Plant height and the presence of reproductive features (i.e. flowers or seedpods) were recorded for all plants found during the survey. Size at reproduction was estimated from the complete data set collected during the flowering period in 2010 using a generalised linear model with binomial errors (with presence of reproductive structures as the response variable). To estimate how reproductive output scales with plant size, we measured plant height and number of flower buds present on plants for 70 individuals at one site. Flower bud counts were used as a proxy for seed production per plant.

Seed banks and seed viability

To get a preliminary estimate of seed bank size, three 0.5 × 0.5 m soil samples dug to a depth of ~10 cm were taken from beneath single large plants (5–6 m tall). Samples were sieved through a graduated sieve stack and seeds counted. As road grading (resurfacing and digging of drainage ditches on gravel roads) is thought to be the primary dispersal agent of *A. stricta* seeds, we also sampled soil that had accumulated on the blade of a road grader immediately after it had dug a drainage ditch into a roadside patch of *A. stricta* (Supplementary Material Appendix B). This was to determine whether seeds were able to be transported along roads during road grading. To provide an estimate of how far off the road seeds were deposited, soil-cores (8 cm diameter × 10 cm depth) were taken along transects that intersected a newly graded plantation road. The road had no large plants but a large number of seedlings were observed on the road which indicates that seeds had probably been deposited during road maintenance. A total of 11 transects spaced 10 m apart were positioned perpendicular to the road and extending 6 m either side of the road. Core samples were dug at 2 m intervals along each transect (N = 77).

Seeds collected from soil sampled beneath canopies were tested for viability using a standard tetrazolium test (Peters 2005). A sample of 200 seeds (50 × 4 replicates) was first scarified using sulphuric acid and then stained using a 1 % 3, 5-triphenyl tetrazolium chloride solution (pH 6.7) for 72 h. Seed coats were removed and viable seeds (indicated by even staining) counted.

Regrowth from the seed bank

Once plants had been recorded and measured, depending on size, they were either pulled up by the roots or cut at the base and sprayed with 3 % glyphosate 360 g/l SL herbicide. The initial survey and clearing in 2010 took 38 field days (or 114 person days) to complete. The study area was resurveyed in September 2011 both to remove seedlings that had germinated and to determine whether the survey and clearing of *A. stricta* in 2010 was effective in finding populations and reducing population numbers. The same destructive sampling method was used to survey all previously recorded sites, and incidences of resprouting were recorded. The follow-up survey took approx. 17 field days (or 51 person days) to complete.

Results

National-scale assessment of likely sites of invasion

The bioclimatic model provided a good fit to *A. stricta*'s distribution in its native range (AUC = 0.971 ± 0.004

SD). Projection of the model to South Africa (Fig. 2a) predicted high climatic suitability over ~15 % of the country. The bioclimatic variables that contributed most to the model were precipitation of the driest month and annual mean temperature (relative contributions of 49.2 and 18.7 % respectively). Using these bioclimatic suitability predictions and the observed association of *A. stricta* with plantation areas, we highlighted all plantations within climatically suitable areas in the country where search efforts should be expanded (Fig. 2a). These areas amount to approximately 685,500 ha or about 36 % of the total forestry land in the country.

Delimiting invasions at a landscape scale

The survey of the study area in 2010 found 19,843 *A. stricta* plants at eight localities, with a total invaded area of ~110 ha (estimated using minimum convex polygons; Fig. 2b). All eight localities had been reported to us by local plantation and conservation managers, i.e. without prior information, we found no additional populations during driving surveys. All populations occurred on forestry plantations with no spread as yet into adjacent fynbos or native forest. Approximately 60 % of the initial vehicle search was done in plantations, with 25, 10 and 5 % of the total searched area comprising natural habitats, farmland and urban areas respectively.

Landscape-scale risk map

Classification trees for models 1 and 2 provided good predictions of *A. stricta* occurrence at a landscape scale (AUC = 0.792 and 0.784 respectively). Model 3 (which included only 9 presence points) did not identify any variables that discriminated *A. stricta* occurrence. The misclassification error of model 1 was 32.8 and 25.4 % for model 2. Based on the lower misclassification error and the simpler rules defined by model 2, we selected this model as most suitable for predicting *A. stricta* occurrence. The model predicts *A. stricta* occurrence based on distance to roads and age of plantation compartments. The results were similar for model 4 (where data were restricted to roadsides only, see Supplementary Material Appendix C) but there was a lower prediction accuracy (AUC = 0.679).

The resulting risk map (Fig. 2b) predicted that ~579 km of roads within the study area are at high risk of invasion by *A. stricta* and should be the focus of search efforts. This is 17 % of the total ~3,425 km road network in the study area and would involve travelling 1,281 km of roads (Supplementary Material Appendix D). Unfortunately only 30 % of the high-risk areas were surveyed in 2010 and 2011. However, the model correctly predicted the location of a new population found in the 2011 survey as a high-risk area (this population was outside the area covered by the vehicle surveys used to develop the risk maps).

Local-scale detectability

The majority (99 %) of plants recorded occurred within 20 m of roads (Fig. 2c). We observed no spread of populations away from roads between the surveys in 2010 and 2011. Most of the landscapes were open with little ground cover (pine plantation), and the surveyors found that once they had a search image for *A. stricta*, plants could be seen from a substantial distance (not quantified here). In cases where plants did occur away from roads, there was circumstantial evidence that it was due to plantation activity (harvesting, clear felling or planting) or that plants were spreading along hiking trails. We found no substantial spread of populations along roads after one year. All plants found in the 2011 follow-up survey were within 100 m of plants recorded in 2010, with 82 % of plants found within 20 m of previously recorded plants (Fig. 2c). There was also no apparent relationship found between the location of plants and their proximity to watercourses. Based on these observations, the resulting risk map (Fig. 2c) suggests that most intensive searching for new plants (i.e. walked surveys) should be done within 100 m of plants recorded previously (regardless of distance to road-edge) and within 30 m of road edges.

Management planning

Risk assessment

Based on data and observations gathered during this study and available literature, *A. stricta* would fail a pre-border risk assessment (Supplementary Material Appendix E; overall score was 18, where >6 indicates potentially invasive) and should be considered a high-risk species in South Africa.

Reproductive output

The minimum height at which plants were found to reproduce was ~30 cm, while 67 % of plants between 1 and 2 m showed signs of reproductive maturity (Fig. 3a). Reproductive output (estimated from flower bud counts) increased exponentially with plant height ($R^2 = 0.788$, $F = 125$, $p < 0.0001$; Fig. 3b). The largest tree measured in the population (3.8 m tall) had >12,000 flower buds. Several plants found during the 2011 survey, in areas that were surveyed intensively in 2010, showed signs of developing flowers. Therefore, as it is unclear if reproduction begins after one or 2 years, annual surveys are warranted to prevent seed set.

Seed banks and seed viability

The seed bank size measured under the plant canopy was ~1,000 seeds m^{-2} (251 seeds ± 2.1 SD per 0.25 m^2 soil sample). The soil collected from the road grader also contained two seeds, showing that *A. stricta* seeds are, as expected, transported during road grading. Soil cores sampled from across-road transects showed that seeds had been deposited up to 6 m from the road, but that 79 % of seeds were along road edges (i.e. along the regularly maintained roadside drainage ditches and ridges). Only 6 % (2–11, 95 % CI) of the seeds sampled from the seed bank were viable.

Regrowth from seed bank

The re-survey of the study area in 2011 found ~15,126 plants (i.e. a 24 % reduction from 2010) with a total invaded area of approximately 92 ha. Most of the plants found at sites were new seedlings of <50 cm (Fig. 3c), with the incidence of resprouting low at all sites (56 plants in total). Field observations showed that resprouting only occurred if the initial cut was made above the lowest branch; correct cutting and herbicide stump application appears to be highly effective.

Discussion

Deciding whether to attempt eradication of introduced species or opt for containment is an important management question and an important focus in invasive species research (e.g. Hester et al. 2013). Uncertainty of invasion extent when attempting eradication can lead to failure if the invasion is poorly delimited and resources are insufficient to remove additional populations (Moore et al. 2011). Accurate delimitation is therefore a crucial component of assessing a species for eradication. We have

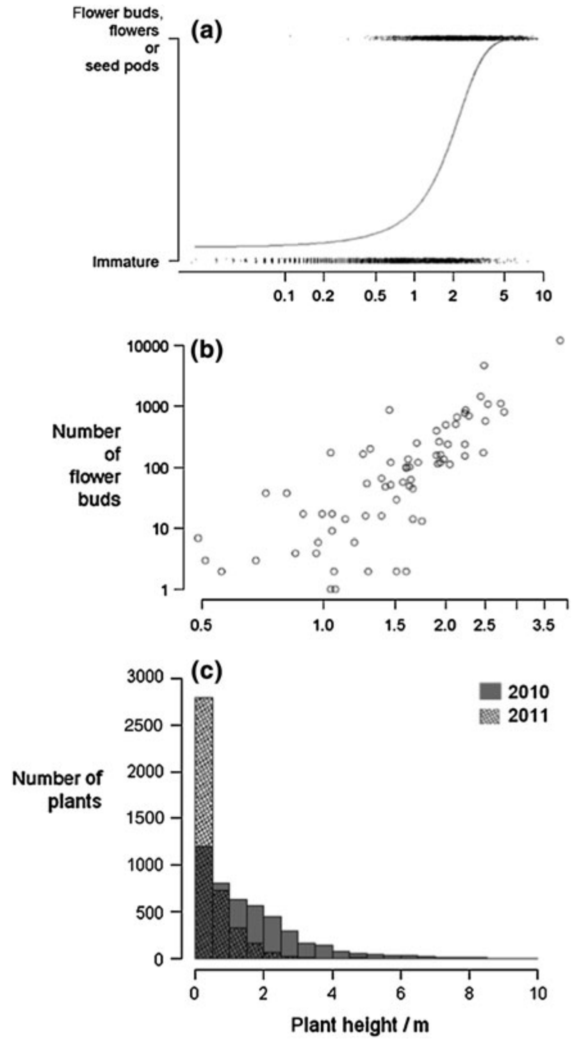

Fig. 3 **a** Plant height of *Acacia stricta* at reproductive maturity. The relationship shown is from a fitted generalised linear model with binomial errors using plant height as an explanatory variable to predict the presence of flowers, flower buds or seedpods; **b** the increase in reproductive output (estimated by the number of flower buds per plant) with plant height; **c** size histogram of *A. stricta* plants measured in 2010 and in the follow up survey in 2011

demonstrated a multi-scale approach for improving detection and rapid delimitation of a species for which there is uncertainty in the total extent in order to assess management feasibility (Fig. 1).

A prerequisite of the framework presented here is some prior knowledge of where the species occurs. In the case of *A. stricta*, several records in the South African Plant Invaders Atlas, along with reports from landowners, provided a good basis for our survey. A strong targeted awareness campaign and a detailed initial search effort will improve the prediction accuracy, but the process can be on-going, i.e. model predictions are improved as more information becomes available. In this way, the search effort is adjusted as more information becomes available (i.e. if populations are found or if areas are searched but no plants are found).

Predicting where to search

The predictive mapping of *A. stricta* provides a means of prioritizing search efforts by minimizing the total area that requires searching, simultaneously increasing the probability of detecting an *A. stricta* population before it is able to grow in size and accumulate large seed banks. Implementing the search strategy within the study area will significantly reduce the overall costs and effort of surveying (by 83 % or ~12,000 ZAR per year) and clearing costs would potentially be less if populations are detected at low density and before any significant seed banks are able to form.

The forestry industry is the most likely pathway of introduction of this species and given the current association of *A. stricta* with plantations in its known invasive range, targeted searches and awareness should be focussed particularly in climatically-suitable forestry areas (Fig. 2a). Possible over-estimation of climatic suitability in South Africa based on home-range distribution patterns (Rejmánek 2000) could over-estimate the number of plantations in the country suitable for *A. stricta* invasions. However as surveillance at a national-scale will predominantly be passive (i.e. creating awareness and gathering reports from local land users) this is unlikely to have any major cost implications and, as such, it would be better to have a more liberal species distribution model.

Active surveys at the landscape scale are very expensive, and ways of objectively prioritizing particular areas would improve the cost-effectiveness of management. The movement of *A. stricta* is most likely a result of soil seed bank spread by road maintenance vehicles such as road graders and plantation harvesting vehicles or equipment (Supplementary Material Appendix B). While this dispersal could be prevented by limiting vehicle activities in the area, the costs of operating road grading equipment greatly outweighs the costs of clearing *A. stricta*, making it expensive and impractical to establish quarantine sites for heavy-equipment working in invaded areas. Instead, increased monitoring at sites where plantation or road maintenance activity has occurred within the previous year should be incorporated into the management plans.

At a local scale, *A. stricta* seeds appear to occur mostly underneath canopies, and dispersal appears to be primarily via gravity or the movement of soil (e.g. there was no evidence of seed dispersal by ants). This is advantageous for management of this species; many other acacias have adaptations for dispersal by birds (Gibson et al. 2011) which would require a much greater search effort. Proximity to roads and previously recorded plant localities provide a good indication of where to search for plants at a local scale. However, we suspect that spread of plants away from roads can occur following disturbance and soil movement away from the roadside during planting, harvesting and clear-felling of plantation compartments.

One major concern in the modelling approach taken here is its correlative nature—resultant predictions of risk are largely a function of the stage of invasion (Peterson 2005). *Acacia stricta,* like most other eradication targets, is at a relatively early stage of invasion. As such, predicting areas at risk of invasion at localized spatial scales by *A. stricta* based on its current distribution might under-predict the total area where plants could potentially occur and where surveillance is required (Jimenez-Valverde et al. 2011; Rouget et al. 2004). Restricting surveillance to roadsides in plantations and limiting monitoring in natural areas might not detect spread to areas we are most interested in conserving. The risk maps of *A. stricta* should therefore not be considered as a predictor of potential long-term population expansion, but rather as a tool to guide the immediate systematic surveillance to be done on a regular basis. Occasional surveillance should perhaps be undertaken in areas where *A. stricta* is not predicted to occur as verification (Fox et al. 2009).

Variation in detection efficiency, mostly determined by the growth form of the target species and the structure of the surrounding vegetation, could influence both the model predictions and the decision outcome (Christy et al. 2010; Reese et al. 2005). Although not quantified in this study, we expect that the detection probability of *A. stricta* is high given its very conspicuous growth form (even when not in flower) compared to the surrounding vegetation and that searching was done during flowering season to improve detectability. Low detection rates could underestimate the extent of the species' extent and result in a failed eradication attempt. In systems where detection probability is low, greater search effort will need to be invested particularly in areas where species are predicted more likely to occur (Hauser and McCarthy 2009). But arguably the most effective strategy to increase detection will be to keep reminding land managers in the area to look out for the species through targeted awareness.

Passive surveillance (i.e. sightings and reports from managers and field workers) have proved to be an important part of locating and delimiting *A. stricta* invasions. All populations found in the initial surveys were identified by land managers, and all subsequent populations found to date were reported by local conservation or plantation managers and field workers in the Knysna area in response to the increased awareness of *A. stricta.* This highlights the benefit of awareness campaigns and active involvement in eradication projects for locating new populations. At a national level, passive surveillance (enabled through awareness campaigns) would be the most cost-effective approach to detecting new populations.

Considering this role of passive surveillance in the detection of *A. stricta*, a highly visible, distinctive species found in well-travelled areas, it is questionable whether a risk mapping approach at the landscape scale is necessary for this species. In the case of *A. stricta*, ensuring stakeholder buy-in and collaborating with people on the ground that have good local knowledge of invasive species would probably be a reliable way of finding new populations, whereas a risk mapping approach would be more useful at a local scale where effective control requires finding every individual.

Eradication feasibility

Given that the invasion by *A. stricta* has been detected at a relatively early stage, and we have so far provided strategies for effective surveillance and awareness, we consider *A. stricta* to be a good candidate for eradication (i.e. category 1a under South Africa's proposed invasive species regulations as part of the *National Environmental Management: Biodiversity Act* 10 of 2004) at its present extent if immediate action is taken to control populations and reduce spread. A focused co-ordinated management plan needs to be implemented to provide effective strategies for finding and removing all populations of *A. stricta* and preventing further seed production and spread, but such a scheme need not be substantially more expensive than current invasive species management (Wilson et al. 2013). Following discussions with stakeholders, it was agreed that a collaborative effort involving all relevant land managers and co-ordinated by SANBI's Invasive Species Programme would be the best way to manage *A. stricta*. A long-term management plan that involves annual targeted vehicle searches and removal of plants at all sites was agreed upon by all parties.

Age at reproduction, reproductive output and seed bank size and longevity will influence the frequency of management and the timeframe of an eradication programme. Since *A. stricta* can possibly reach reproductive maturity within one year, follow-up clearing should be done on an annual basis to prevent plants contributing to the seed bank. With the current distribution (in 2011) of approximately 92 ha and an estimated cost of clearing of 400 ZAR per ha (based on costs of clearing alien plants collated by MTO Forestry Pty Ltd.), the estimated cost of removing all plants at existing sites is ~36,800 ZAR per year. Further studies are needed to determine the depletion rates of the seed bank in order to estimate the likely duration and the total cost of the eradication programme.

The feasibility of eradication of *A. stricta* will need to be re-evaluated if additional populations are found elsewhere in South Africa. However it seems unlikely that this species has gone unnoticed in other parts of the country given the distinctive erect growth habit and inflorescence position that easily distinguish adult plants from other acacias, and the general interest and attention paid to invasive Australian acacias in South

Africa. However, if eradication was found to be unfeasible a modelling approach similar to that demonstrated here could be used to improve containment efforts or used as a tool to prioritize invasive species management in areas with important ecological assets or threatened habitats.

Conclusions

Risk mapping provides a useful method of consolidating information in a form that can produce informative management products. Mapping and assessing invasions at multiple scales is important for accurate delimitation and assessment of eradication feasibility. However, the likely lack of generality of risk mapping means that a specific risk map would be required for each species. While risk maps can reduce the overall costs and effort required for searching and monitoring, our experience has shown that active involvement of stakeholders in the surveillance plays an important role in the rapid delimitation of a species' invasion extent.

Acknowledgments This project was funded by the DST-NRF Centre of Excellence in Invasion Biology and the Working for Water (WfW) Programme through their collaborative project on "Research for Integrated Management of Invasive Alien Species" and through the WfW funded South African National Biodiversity Institute's Invasive Species Progamme. DMR acknowledges financial support from the National Research Foundation (Grant 85417), and the Oppenheimer Memorial Trust. We thank Dane Paijmans and Suzaan Kritzinger-Klopper for assistance in the field; Ernita van Wyk for co-ordinating the management meeting; and SANParks and MTO Forestry (Pty) Ltd. for logistical support.

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ONLINE SUPPLEMENTARY MATERIAL

APPENDIX A

Information flyer

In order to get further reports of *Acacia stricta* populations, information flyers were distributed by hand to local land managers. Managers were then asked for the locations of populations of this species if they had encountered any. The flyers were targeted at groups who routinely work in the field and were therefore likely to recognise *A. stricta* populations. These included conservation organisations (South African National Parks and CapeNature) and plantation managers at forestry companies (MTO Forestry (Pty) Ltd. and PG Bison), as well as a local tree nursery. The flyers contain a detailed description of *A. stricta* including its distinguishing features. Reports are directed to the South African National Biodiversity Institute's Invasive Species Programme for follow up to determine whether they are accurate and whether they are indeed new populations or have been previously recorded.

APPENDIX B

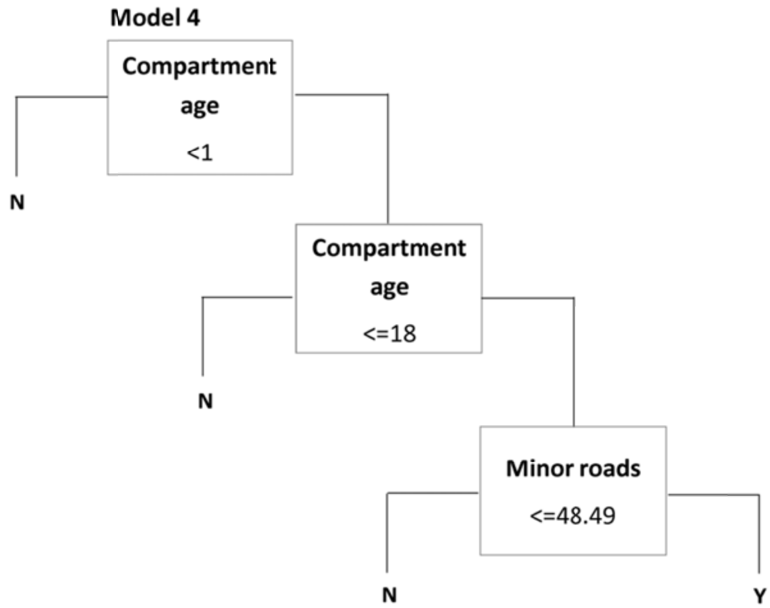
Video of road grading equipment moving seeds of *Acacia stricta*

The video available at

http://academic.sun.ac.za/cib/supplementary/Kaplat%20et%20al%202014%20Biological%20Invasions%20Supplementary%20Material%20Appendix%20B.AVI

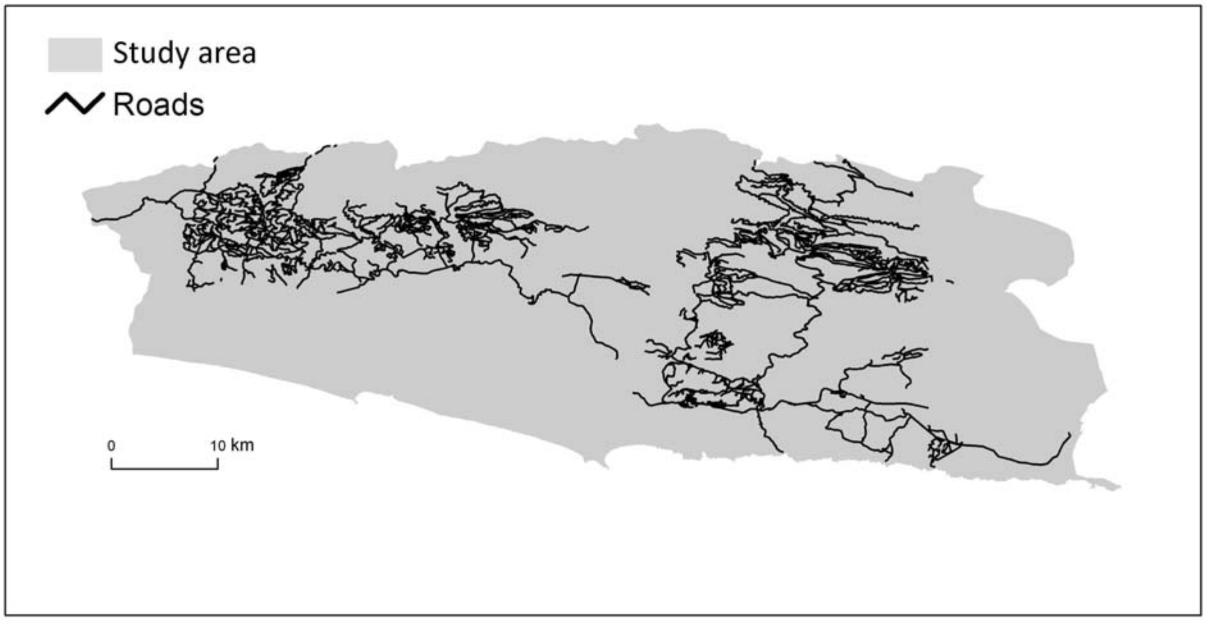
shows a road grader cutting a drainage ditch into the side of a dirt road used by plantation vehicles as part of routine road maintenance. The ditch was in an area infested by *Acacia stricta* (the light green erect saplings visible in the mid-ground of the picture). The soil attached to the blade of the grader was sampled after the video was taken, and seeds of *Acacia stricta* were found in this soil, indicating that the road grader is a method for seed dispersal.

APPENDIX C



Appendix C: Recursive partitioning model used to predict the occurrence of *Acacia stricta* based on age of plantation compartment and distance to minor roads. Presence and absence data used in this model were restricted to within 50 m of roads.

APPENDIX D



Appendix D: Required route for vehicular surveys to search all areas predicted as being at risk of invasion by *Acacia stricta.*

APPENDIX E: Data for *Acacia stricta* for the Australian Weed Risk Assessment Protocol (following Pheloung et al 1999)

Question	Answer	Score	Range of possible scores	Reference
Is the species highly domesticated?	No	0	0 or -3	
Species suited to South African climates	High	1	2	this paper (fig. 2a)
Quality of climate match data (0-low; 1-intermediate; 2-high)	Intermediate	1	0,1 or 2	
Broad climate suitability (environmental versatility)	Yes. Found in sub-tropical and temperate type climates.	1	0,1 or 2	[1]
Native or naturalized in regions with extended dry periods	Yes	1	0 or 1	[1]
Does the species have a history of repeated introductions outside its natural range?	No	0	0 or 1	
Naturalized beyond native range	Yes. In South Africa and New Zealand.	2	0,1,2,-1 or -2	[1]
Garden/amenity/disturbance weed	Yes. Invades disturbed roadsides.	2	0,1 or 2	pers. obs.
Weed of agriculture/horticulture/forestry	Yes. Invades forestry plantations.	3	0,1,2,3 or 4	pers. obs.
Environmental weed	Not known	?	0,1,2,3 or 4	
Congeneric weed	Yes	2	0,1 or 2	[2]
Produces spines, thorns or burrs	No	0	0 or 1	
Allelopathic	No	0	0 or 1	
Parasitic	No	0	0 or 1	
Unpalatable to grazing animals	Not known	?	1 or -1	
Toxic to animals	No	0	0 or 1	
Host for recognised pests and pathogens	Not known	?	0 or 1	
Causes allergies or is otherwise toxic to humans	Not known	?	0 or 1	
Creates a fire hazard in natural ecosystems	Not known	?	0 or 1	
Is a shade tolerant plant at some stage of its life cycle	No	0	0 or 1	
Grows on infertile soils	Yes	?	0 or 1	[3]
Climbing or smothering growth habit	No	0	0 or 1	
Forms dense thickets	Yes	1	0 or 1	pers. obs.
Aquatic	No	0	0 or 5	
Grass	No	0	0 or 1	
Nitrogen fixing woody plant	Yes	1	0 or 1	

Question	Answer	Score	Range of possible scores	Reference
Geophyte	No	0	0 or 1	
Evidence of substantial reproductive failure in native habitat	No	0	0 or 1	
Produces viable seed	Yes	1	1 or -1	this paper
Hybridises naturally	Yes. Possibly with *A. paradoxa*	1	1 or -1	[4]
Self-fertilisation	Unknown	?	1 or -1	
Requires specialist pollinators	No	0	0 or -1	
Reproduction by vegetative propagation	Yes	1	1 or -1	[5]
Minimum generative time (years)	1 year	1	0,1 or -1	
Propagules likely to be dispersed unintentionally	Yes	1	1 or -1	
Propagules dispersed intentionally by people	No	-1	1 or -1	
Propagules likely to disperse as a produce contaminant	No	-1	1 or -1	
Propagules adapted to wind dispersal	No	-1	1 or -1	
Propagules buoyant	Not known	?	1 or -1	
Propagules bird dispersed	Not known	?	1 or -1	
Propagules dispersed by other animals (externally)	Not known	?	1 or -1	
Propagules dispersed by other animals (internally)	No	-1	1 or -1	
Prolific seed production	Yes	1	1 or -1	pers. obs.
Evidence that a persistent propagule bank is formed (>1 yr)	Yes	1	1 or -1	this paper
Well controlled by herbicides	Yes	-1	1 or -1	pers. obs.
Tolerates or benefits from mutilation, cultivation or fire	Yes	1	1 or -1	pers. obs
Effective natural enemies present in Australia	Not known	?	1 or -1	

[1] Australian Virtual Herbarium

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