Dealing with 'new' alien plants: risk assessment and risk management

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Summary Decisions on how to deal with newly detected alien plant incursions need to be made against a background of uncertainty, initially with respect to the weed risk posed. Then, should a high risk be indicated there will be further uncertainty regarding how best to manage it. If a coordinated management strategy (e.g. eradication) is favoured, its feasibility and likelihood of success will depend *inter alia* upon the allocation of sufficient resources, but what constitutes 'sufficient' is difficult to estimate.

Weed risk assessment (WRA) is challenging, in part owing to an incomplete understanding of the mechanisms of invasiveness and, especially, impact. Desktop post-border WRA procedures are available, but there appears to have been little attempt to estimate the invasiveness or potential impact of new incursions based on real-time field observations. I discuss how such observations, in conjunction with weed history (where available), could be used to determine appropriate incursion responses. A recent categorisation of species in relation to biological impedance to eradication is recommended as a qualitative aid to estimating the resources required for eradication. Implications for the assessment of containment feasibility are also considered.

Keywords Containment, dispersal, eradication, feasibility, incursion, invasiveness, impact, weed history.

INTRODUCTION

Co-ordinated management strategies, such as eradication or containment, can prevent or substantially delay the impacts of new alien plants, particularly if detection occurs soon after plant introduction (Wilson *et al*. 2016). However, implementation of such strategies is generally expensive and labour-intensive. Constraints on the resources available for managing new incursions mean that difficult decisions need to be made. Decision makers first must take into account the risk posed by a new incursion: does the species represent a threat sufficient to warrant the investment of scarce resources? Weed risk assessment (WRA) poses a number of challenges, the most fundamental being that the plants that will become serious weeds comprise only a small proportion of all introductions (Hulme 2012). This is why a plant's history as a weed, in conjunction with a high probability of climatic suitability in the area at risk of invasion, remain the most powerful predictors of its weed risk. Obviously, this approach will not be an option for plants that have no history of weediness.

If a species is considered to pose a high risk, the second issue faced by decision makers is how to manage it. Here the key question is: how likely it is that a coordinated management strategy will be successful within the constraints of the resources that could be expected to be available? The factors that influence eradication feasibility are by now well known, characterised and, in many cases, quantified (Panetta 2015). A distinction can be made between factors that are extraneous to the targeted species (e.g. societal attitudes towards the plant and the infrastructure available for strategy implementation) and those relating to the target and where it is found (e.g. intrinsic biological features, detectability and the accessibility of its populations). The second group of factors comprises the 'technical feasibility' of eradication. Similar considerations pertain where the management goal is containment (Wilson *et al*. 2016).

The dynamics of alien plant invasions are usually much less dramatic than those observed for other invasive organisms, but timely action is still essential because positive management outcomes are more likely where smaller spatial extents of targeted species are involved (Pluess *et al*. 2012). Accordingly, incursion management decisions need to be made under conditions of considerable uncertainty. In many cases uncertainty arises because weed risk is not well known, but in almost all situations it is difficult to gauge with confidence whether a coordinated management strategy is likely to succeed, given the available (or anticipated) resources. The aim of this paper is to explore ways in which these underlying uncertainties can be made more explicit and can be better addressed when dealing with detections of new alien plants, especially where such plants are new to a location or region.

POST-BORDER WRA

If alien plant incursions are to be managed proactively, decisions need to be made before incursions develop into full-blown invasions. The first decision to make is whether a newly detected species poses sufficient risk to warrant further action. A number of desktop systems are available to assess post-border weed risk (e.g. Weiss *et al.* 2004, Setterfield *et al.* 2010) but in these procedures invasiveness is estimated largely on the basis of plant features (e.g. reproductive ability and dispersal mechanisms). Furthermore, impact estimates are based upon effects that could be observed at invasion stages that might better be considered as post-incursion. In this section I will describe a simple approach in which weed history is combined with observations made in the field in order to assess weed risk in a new location. The scope of the present exercise will be confined to weeds of natural ecosystems for two reasons:

1) these systems are generally more complex than agro-ecosystems, meaning that it is more difficult to assess risk, and
2) funding for incursion management in natural ecosystems is often more difficult to source than for management in agro-ecosystems, since the economic imperative associated with the latter is much clearer.

Weed history Online resources are invaluable sources of information on weed history. A particularly useful online compilation is 'A Global Compendium of Weeds' (Randall 2012). In this source records for environmental weeds are identified specifically, but how to evaluate the strength of evidence for weediness is an important issue that remains to be resolved. Owing to the matter of cascading references (Randall 2012) the total number of references listed here is not a particularly good metric.

Panetta (2016) has suggested that the number of world regions in which a species has been documented as an environmental weed (Figure 1) is a useful proxy for adaptation to a broad range of environmental conditions, both non-biological (e.g. climate and soils) and biological (e.g. different plant communities). However, the relationship between the number of world regions in which a species is an environmental weed and the strength of this factor as a predictor is unclear. Is a single region sufficient? (NB in Randall's compendium where a weed occurs in a single region this is almost always Australasia.) Presumably, as the number of regions increases a larger number of plant communities would have been invaded and impacted, and a broader range of abiotic conditions encountered.

Because most single-region recordings of species in Randall (2012) correspond to Australasian occurrences, a cut-off of two regions was used to distinguish between low and high levels of evidence for weed history (Panetta 2016), but this remains unsubstantiated. Detailed evidence of weed history, including description and quantification of impacts, may be gleaned from original sources listed in Randall (2012) as well as the scientific literature. Species that elsewhere have transformed the character, condition, form, or nature of ecosystems (Richardson *et al.* 2000) are often the ones that pose the highest risk (e.g. Fried and Panetta 2016).

Climate matching is an important adjunct to the use of weed history as a predictor of risk, but whether this is essential depends on scale: at the local scale, pathways of introduction may be sufficiently informative. For example, where a plant has been introduced for ornamental or amenity purposes, its appearance in a natural ecosystem indicates sufficient climatic pre-adaptation to enable it to escape cultivation and to persist, at least in the short term. If it has been introduced for purposes of production, its selection, importation and widespread use are all based on an assumption of climatic pre-adaptation (Panetta 2016). Where risk is being assessed at the level of a country or continent, however, climate matching will need to be undertaken as part of the procedure for estimating the total area at risk.

Field observations 'Likelihood' in the Risk = Likelihood × Consequence equation refers to the probability that an introduced plant will establish and spread,

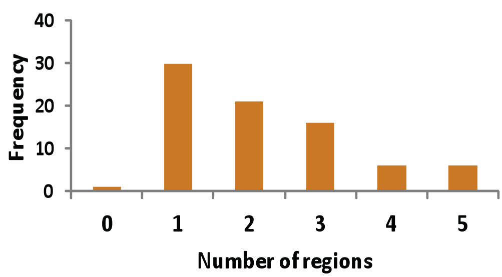

Figure 1. The number of world regions in which species (n = 80) sampled from the Victorian Weed Advisory Lists occur as environmental weeds, according to Randall (2012). World regions comprise Africa, Australasia (i.e. Australia and New Zealand), Caribbean, Europe, North America, Pacific, South America and Sub Antarctic. From Panetta (2016).

and 'consequence' encompasses all of the negative impacts that could result from its spread (Daehler and Virtue 2010). Post-border risk assessment has a significant advantage over its pre-border counterpart in that there is an opportunity to quantify spread *in situ* and to make at least qualitative observations regarding impact. Unfortunately, there appears to have been little attempt to exploit this opportunity.

Recent publications (Blood *et al.* 2016, Panetta 2016) detail a risk assessment screen that can be used by public land managers to determine when a newly detected species poses sufficient risk to warrant its delimitation, prior to making a decision on how best it might be managed. Operating through a score sheet template, the screen comprises *measures* of invasiveness (spread) and impact and *filters* that assist in the interpretation of field observations.

Invasiveness measures relate to whether reproduction occurs by vegetative means, by seed or by both means, plus the distance to which spread has occurred (Table 1). Potential indicators of impact, based upon both the potential to alter vegetation structure and the cover attained are also provided. Given the challenges associated with quantifying (let alone predicting) impact, this part of the screen remains a work in progress. A key consideration is that seedling density may be more effective in capturing impact potential for plants that have longer juvenile periods (e.g. some shrubs and most trees). Furthermore, it remains to be determined whether the use of qualitative measures for impact is superior to quantitative measures, the latter requiring the establishment of threshold (cut-off) values as part of a binary decision-making procedure.

There are two types of filters, relating to time and disturbance respectively (Figure 2). Time-related filters are designed to capture the opportunity that a species has had to express its potential for spread and for generating impacts. These are based upon residence time (how long a plant has been present, if known specifically or otherwise able to be estimated) and how long it would take for an individual to reproduce (i.e. the length of its juvenile period – this will be more important for plants that have long juvenile periods, such as some shrubs and many trees).

The disturbance filter is constructed to reflect the observation that many of the environmental weeds that pose the greatest risk to natural ecosystems are ones that have the ability to establish in relatively intact vegetation. It uses information about whether a species is restricted to edges or areas that have been recently disturbed, or is associated with other plants that are commonly considered to be indicators of disturbance.

In Figure 3 weed history evidence and field-derived measures of plant performance are assembled in a matrix that shows recommended management actions, together with levels of uncertainty associated with these recommendations. Here weed history evidence is treated qualitatively owing to the unresolved

Table 1. Scoring for invasiveness, modified from Panetta (2016).

Evidence of reproduction	
Has the plant reproduced in place by joined vegetative structures?	Yes = 1 No = 0
Has the plant reproduced by seeds or detachable vegetative structures?	Yes = 2 No = 0
Pre-reproductive period	
How long does it take for a new individual to produce seeds or other propagules?	< 1 yr = 3 1–2 yrs = 2 > 2 yrs = 1 Default[A] = 1
Evidence of spread	
For connected vegetative spread	≤ 1 m = 0 > 1 m = 1
For spread via seed or detachable vegetative propagules	≤ 10 m = 1 10–50 m = 2 > 50 m = 3

[A] For shrubs and trees.

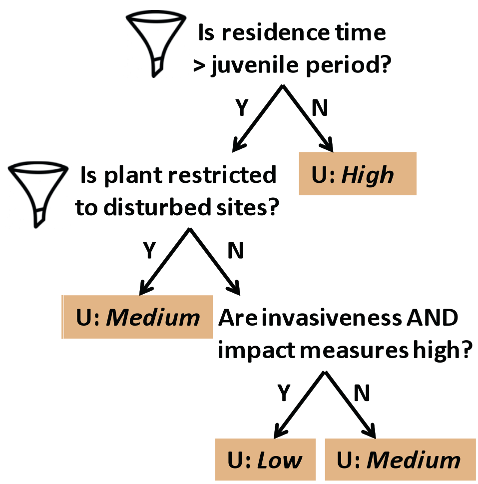

Figure 2. Flow diagram showing how time- and disturbance-related filters can be used to characterise the level of uncertainty (U) associated with weed risk assessments that are based on field-derived measures of invasiveness and impact.

issue raised in the previous section, i.e. the number of world regions versus strength of weed history as a predictor.

Site-based evidence	Weed history evidence: None/little	Weed history evidence: Abundant
Low	Monitor	Consider management feasibility?
High	Delimit and consider management feasibility	Delimit and consider management feasibility

Figure 3. Risk assessment matrix based on different combinations of levels of evidence from weed history and site-based measures. Levels of uncertainty associated with recommended risk management actions range from lowest (no shading) to highest (heavy shading). Highest uncertainty accompanies a situation where the management action adopted will likely depend on attitudes towards risk (see concluding remarks).

requiring more frequent site visits than slow-maturing perennials. Since eradication cannot be declared until seed banks are exhausted, seed persistence is a strong determinant of the minimum time required for an eradication program. Panetta (2015) used cut-off values of two years for juvenile period length and three years for seed persistence to categorise species in terms of their biological impedance. Different attribute combinations yield four species categories, which can be ranked from lowest to highest biological impedance (Figure 4).

1. Slow maturing; seeds short-lived

2. Slow maturing; seeds long-lived

3. Fast maturing; seeds short-lived

4. Fast maturing; seeds long-lived

Figure 4. Ranking of plants according to biological eradication impedance posed at a similar stage of invasion. The arrow indicates order of increasing impedance.

ERADICATION FEASIBILITY

Factors that determine the technical feasibility of eradication fall under four broad categories: logistic considerations, species detectability, species biological characteristics and control effectiveness (Panetta and Timmins 2004). These factors will influence the amount of impedance posed by a targeted species – where higher levels of eradication impedance prevail, it will be more difficult (and hence more costly) to achieve eradication. In this section I will focus first on the biological components of eradication impedance, which together provide a biological rationale for categorising and ranking species in terms of eradication feasibility. Next I will consider how dispersal processes may be expected to interact with this categorisation.

Biological impedance Key features of this component of impedance are the duration of the juvenile period and for how long propagules (generally seeds, but sometimes other vegetative structures) persist in the field. Juvenile period length will determine how frequently infestations must be visited in order to prevent seed production, with rapidly cycling annuals

Examples of species in each category are given in Panetta (2015), but to illustrate the extremes, many plants that were introduced for ornamental and amenity purposes, e.g. camphor laurel (*Cinnamomum camphora* (L) Presl.), broad-leaved Brazilian pepper (*Schinus terebinthifolius* Raddi) and privets (*Ligustrum* spp.) demonstrate the least impedance (Category 1) and a number of weeds of crops and pastures, e.g. parthenium weed (*Parthenium hysterophorus* L.) and parasitic species in the genera *Phelipanche/Orobanche* and *Striga* demonstrate the most (Category 4). An extensive survey that would provide the basis for estimating the relative frequencies of each category in regional weed floras could prove very useful.

Dispersal While it is possible for a plant to cause significant impact without spreading (Wilson *et al.* 2016), the most problematic species will be ones that not only generate impact but also have (often multiple) dispersal mechanisms that assist them in moving beyond sites of introduction. The scale of management needs to approximate that of a plant's dispersal kernel if eradication is to be a realistic goal (Fletcher and Westcott 2013).

Most important are the mechanisms that promote long distance dispersal, since these facilitate the most rapid spread (Nathan 2006). Human-mediated dispersal (HMD) (dispersal by humans and their agents, e.g. machinery and livestock) is particularly effective in transporting weeds for long distances, often with sufficient propagule numbers to overcome barriers to establishment. From an eradication perspective HMD also provides multiple opportunities to manage spread, ranging from maintaining a very high standard of hygiene for machinery, to trace-forward activities that can identify previously undetected infestations. HMD, therefore, is a 'two-edged sword' that can be exploited but at the same time requires very careful management. Where long distance dispersal occurs largely via other mechanisms, e.g. dispersal by wild animals or physical vectors such as water, spread will be prevented most effectively by reducing levels of propagule production (Panetta and Cacho 2012). Note that prevention of reproduction is a key management goal in weed eradication efforts.

Impedance × dispersal interactions Unmanaged long distance dispersal will decrease the likelihood of achieving eradication. Maintaining vehicular and personal weed hygiene during search and control operations is therefore critical if operators are not to contribute unwittingly to further spread of the targeted species (Bocking *et al.* 2008). Preventing, or at least minimising, the reproduction of species whose major vectors are essentially unmanageable should reduce the risk of further spread from *known* sites. However, if additional foci of infestation (present either before or arising during the course of an eradication program) are not detected and managed, they will in turn contribute to further spread. The negative impact of further spread upon the probability of eradication will likely increase in order from Category 1 to Category 4 species (Figure 4), a function of combining shorter life cycles with greater persistence of the propagules that are produced. Reflecting both factors, a large component of the cost of eradication is attributable to the need to exhaust weed seed banks (Panetta and Cacho 2014).

IMPLICATIONS FOR CONTAINMENT

Containment is often considered as a fallback strategy when eradication appears to be unattainable, given budgetary and other constraints (Hulme 2006). Containment is not always the best option if an eradication program is not performing well, but it does become proportionally cheaper than eradication for plants that have smaller dispersal distances or more persistent seed banks (Fletcher *et al.* 2015). A containment program may require separate approaches to core, as opposed to outlier, populations. The establishment of barrier zones around core populations is not favoured when dispersal cannot be managed effectively, resulting in the possibility of frequent barrier zone breaches. Equally, eradication of outliers should not be attempted when a plant has a short juvenile phase, combined with long-lived seeds (i.e. Category 4 species) (Panetta and Cacho 2014). Major factors contributing to containment impedance include: time to reproduction; aspects of detectability (whether the species is conspicuous at any stage, plus the probability of passive detection, given where it occurs in relation to human habitation or activity); and the relative importance of human mediated dispersal (Wilson *et al*. 2016).

CONCLUDING REMARKS

Without timely intervention, opportunities for proactive management of new alien plant incursions will be lost. Direct observations on invasiveness and impact, supplemented by information available on the weed history of the species of interest may provide a reliable basis for a rapid assessment of the weed risk posed by a plant. Where such an assessment is made in support of decisions relating to intervention at a local scale, the need to establish climate suitability is moot; climate-matching would, however, be required to assess the risk posed over very much larger areas. Until now there appears to have been little attempt to develop even semi-quantitative methods that can be employed to integrate field observations into weed risk assessments for new incursions.

The attitudes of decision makers and managers towards risk influence the choices that are made regarding new incursions of alien plants. The predominating preference is for investment in the control of widespread species, because resulting productivity gains are more or less certain here. Perhaps counter-intuitively, this is actually a manifestation of *risk aversion*. The future benefits of managing a specific incursion are far less clear: without intervention the species might ultimately have no serious impacts; the incursion response might fail; and even if eradication is achieved, future incursions might occur. However, Finnoff *et al.* (2007) argue that a *risk-neutral* approach should be adopted, since an acceptance of the uncertain benefits of proactive management will in most cases increase overall benefits.

Plants having rich histories as weeds elsewhere, yet low levels of site-based evidence of invasiveness or impact, are somewhat problematic. Might they be 'sleepers'? They should be relatively easy to eradicate, yet a risk-averse response (*sensu* Finnoff *et al.*) would be to 'wait and see'. However, very small incursions can be managed proactively with relatively little cost and a high likelihood of success. Harris and Timmins (2009) have argued in favour of targeting numerous incursions (provided that management costs for each are relatively low) in order to 'capture' the species that would ultimately cause the most damage.

Uncertainty around the amount of resources required to achieve a desired outcome from a co-ordinated management program may be reduced through consideration of the biological characteristics of the targeted species, as well as the dispersal mechanisms likely to be involved. For a given stage of invasion (including pre-invasion!), proactive management programs for species in which long pre-reproductive periods are combined with low seed persistence will be both less resource-demanding and have a higher likelihood of success per dollar invested than those in which the opposite characteristics pertain.

Successful containment, but failed eradication, of a relatively large incursion of branched broomrape (*Phelipanche ramosa* F.W.Schulz, formerly *Orobanche ramosa* L.) (7535 ha gross area) is an example of a targeted plant whose rapid reproduction was coupled with very long-lived seeds (Panetta 2015). Improved risk assessment procedures and better-informed decisions on risk management can be expected to yield a greater level of success overall in incursion response, but there remains a critical need for 'more runs on the board' in support of future investment in this area.

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