Prioritizing plant eradication targets by re-framing the Project

Prioritization Protocol (PPP) for use in biosecurity applications.

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(A) ABSTRACT

The eradication of newly detected alien plant species is often prescribed, but rarely successful. Eradication programs fail for many reasons, however, for eradication to remain a cost-efficient management option it is clear that good decisions must be made at the outset. Here we re-frame the Project Prioritization Protocol (PPP), a tool widely used in conservation biology, for use with the metrics typically used by a biosecurity agency. We then use existing methods to estimate the cost-efficiency of eradicating 50 hypothetical species incursions and compare the reduction in weed risk achieved by allocating resources using the PPP framework with the allocation based on risk ranking. By allocating resources to plant eradication programs using the PPP our analysis indicated that it is possible to improve the return on public expenditure by 25% compared to investing based solely on weed risk assessment scores. We also demonstrate how the cost-efficiency of the overall portfolio is influenced by the choice of planning horizon; including the decline in overall portfolio performance that arises when attempting to eradicate individual species too quickly. Finally, we discuss the logistical benefits to a management agency that arise from the use of a generic overarching framework such as the PPP. We believe that the PPP has considerable potential for use in biosecurity and can help focus attention on those species where management can make the biggest difference.

(A) **INTRODUCTION**

Biosecurity, and invasive species management as a sub-discipline, is fundamentally an exercise in risk management (Downey et al. 2010a; Heikkilä 2011a; Virtue et al. 2006). Given a set of objectives (usually to minimise negative impacts on the economy, the environment and health), a management agency's primary responsibility is to deliver interventions that maximise the likelihood of achieving the objectives within the available budget (Game et al. 2013; McCarthy et al. 2010; Possingham et al. 2001). Biosecurity policies internationally reflect that there can never be zero risk associated with trade and the movement of people and goods. Proposed interventions must, therefore, be prioritized according to their costs and expected benefits to ensure the best return on public expenditure (Beale et al. 2008; Dana et al. 2014; Heikkilä 2011b).

The most common approach to prioritizing invasive plant interventions has been to rank species based on the predicted magnitude of their weed risk (see Downey et al. 2010b; Forsyth et al. 2012; Gordon et al. 2008; Hiebert 1997; Kumschick et al. 2012; Leung et al. 2012; McGeoch et al. 2015; Randall 2000; Weiss and McLaren 2002 for reviews of key concepts). Whilst specific methodologies vary, most approaches follow a conceptual framework that uses scoring systems to estimate the 'risk' of a species as a function (usually the sum or product) of scores of its 'invasiveness', 'impact' and 'potential distribution' based on various demographic traits. Here, *invasiveness* represents the ability of a species to enter, establish and spread; *impact* represents the likely impact of the species where it establishes; and *potential distribution* is the species' predicted maximum spatial extent (Downey et al. 2010a; Virtue et al. 2006; Weiss and McLaren 2002). Within this framework, the highest 'risk' species (those with the largest potential to enter, establish, spread and

cause impact) are considered the highest priority for management (Downey et al. 2010a; Kumschick et al. 2012; McGeoch et al. 2015).

In parallel, conservation biologists have dealt with the inverse problem of prioritizing the management of species at risk of becoming extinct (see Akçakaya et al. 2000; Brook et al. 2000; Mace et al. 2008; O'Grady et al. 2004; Possingham et al. 2002; Rodrigues et al. 2006). These two disciplines are somewhat analogous in their management goals, which are to minimise expected harm by moving a species either towards, or away from, local extinction. However, despite the obvious potential for the exchange of ideas between disciplines, few approaches have crossed contexts (Dana et al. 2014; Heikkilä 2011b; Moore et al. 2011). One exception is the recent proposal by Blackburn et al. (2014) to classify (rather than score) alien species based on their known environmental impacts in a framework similar to the International Union for Conservation of Nature's Red List (IUCN 2012). The connection between the two approaches being that species in the highest risk categories are typically considered the highest priority for management (Blackburn et al. 2014; IUCN 2012).

From a resource allocation standpoint, prioritizing management solely on risk [in either discipline] is inefficient because it ignores the degree to which the risk can be managed, and the cost of intervention (Brooks et al. 2006; Pannell and Gibson 2015; Possingham et al. 2001; Possingham et al. 2002; Wilson et al. 2006).

Recognising this, several of the management focused invasive plant prioritization frameworks include an additional measure of management 'feasibility' (Darin et al. 2011; Downey et al. 2010a; Hiebert 1997; Nel et al. 2004; Robertson et al. 2003; Virtue 2005; Virtue et al. 2006; Virtue 2010). The most common approach to implementing these risk management frameworks has used risk matrices to classify

species into management strategies [eradication, containment etc.] (Downey et al. 2010a; Virtue 2005; Virtue et al. 2006; Virtue 2010), although Darin et al. (2011) used an Analytic Hierarchy Process (AHP) to rank interventions.

In contrast to invasive species management, early species conservation methods such as the Noah's Ark framework (Metrick and Weitzman 1998; Weitzman 1998), prioritized actions based on their cost-efficiency (biodiversity benefit per dollar) without considering management feasibility. Here, the biodiversity benefit was estimated based on measures of the species' taxonomic value and the change in its likelihood of persistence. McCarthy et al. (2008) then extended this method to allow the rate of management success to change non-linearly with investment. The Project Prioritization Protocol (PPP) developed by Joseph et al. (2009) was the first to integrate extinction risk, management feasibility and the cost of intervention; proposing a simple framework for allocating limited resources on the basis of cost-efficiency. When compared with approaches that ignore some or all of these factors the PPP improves investment performance by 20-50% (Pannell and Gibson 2015).

Several landscape scale management programs have applied the PPP framework to prioritize invasive species control projects to maximise biodiversity benefit (Carwardine et al. 2012; Firn et al. 2013; Firn et al. 2015). However, only Walshe et al. (2012) have applied the PPP in the macro-scale context typically confronted by biosecurity agencies. Here we investigate the potential for using the PPP to prioritize macro-scale biosecurity interventions such as plant eradication. We re-frame the PPP for use in a biosecurity setting and estimate the cost-efficiency of intervening in each of 50 hypothetical incursion scenarios. We explore the influence of time and budget on the cost-efficiency of the interventions and compare the

129 reduction in weed risk achieved by allocating resources using the PPP framework

130 with the allocation based on weed risk alone.

131

(A) METHODS

(B) Re-framing the Project Prioritization Protocol

The nine-step PPP (Joseph et al. 2009) is similar to many risk management frameworks (for example Virtue et al. 2006), that sequentially define objectives; identifies and assesses risks [species]; and evaluates risk treatments [projects] in light of known constraints [budget]. Where the PPP differs from other frameworks, is the way it integrates risk, management feasibility and the cost of intervention for a set of projects. Projects are ranked based on their cost-efficiency (units of risk offset per dollar spent), which is measured by:

$$CostEfficiency = \frac{Weight \times Benefit \times Probability}{Cost} \quad (eqn. 1)$$

where the *weight* represents the inherent value of the species; the *benefit* is the difference between the probability of the species being secure (extant) in 50 years with and without management; *probability* is the probability of the project being implemented successfully; and the *cost* is the monetary cost of the project in net present value.

Translating the cost-efficiency function (eqn. 1) into the biosecurity context is straightforward. Following Pannell and Gibson (2015) the *weight* is the value of the species (however defined) and the *benefit* is the proportional difference in value [effectiveness] achieved with and without management [the counterfactual scenario]. In the biosecurity context, the value of the species is its weed risk, and the benefit is the proportion of that risk that is ameliorated by the intervention. All other inputs remain the same. Thus, the modified cost-efficiency function is:

$$158 \quad CostEfficiency = \frac{Risk \times Effectiveness \times Probability}{Cost} \quad (eqn. 2)$$

159

160 where *risk* is the species' weed risk assessment score; *effectiveness* is the
 161 proportional reduction in that risk as a result of the intervention; *probability* is the
 162 probability of the intervention being implemented successfully; and the *cost* is the
 163 monetary cost of the project in net present value.

164

165 (B) **Step 1: Define objectives and planning [time] horizon**

166 The first step in any prioritization process is to identify the objective(s) (Joseph et al.
 167 2009; Possingham et al. 2001; Virtue et al. 2006). The objective chosen in this study
 168 was the long-term objective of the Victorian Government: that "Victoria's wealth,
 169 wellbeing and biodiversity will be protected and enhanced by reducing the impact of
 170 invasive species" (Victorian Government 2010). Triple bottom line objectives like
 171 Victoria's are quite common in Australia (Commonwealth of Australia 1997, 2007),
 172 however, other objectives include: to maximise the number of species eradicated, or
 173 to maximise the chance of achieving a minimally sufficient outcome (such as
 174 eradicating a certain number of species). It is also important then to specify a
 175 planning horizon in which those goals are to be achieved because, without an end
 176 date, it is otherwise impossible to determine whether the program was successful
 177 (Bomford and O'Brien 1995; Dodd et al. 2015). To investigate whether the choice of
 178 planning horizon influenced the choice of eradication targets, we repeated the
 179 prioritization analysis below for each 5 year interval from 10 to 50 years.

180

181 (B) **Step 2: List possible species targets**

182 Management agencies routinely receive suggestions of species requiring intervention
 183 and are rarely short of possible projects. The list of 50 species analysed here was

drawn from the pool of species refused entry into Australia since 2010 (provided by the Australian Department of Agriculture and Water Resources), and not known to be present in the country. To simulate a realistic hypothetical incursion, each species was randomly assigned an infested area between 1 and 1000 ha drawn from a uniform distribution. The required demographic inputs for each species (propagule longevity, life-form, detectability, etc.) were sourced from the literature. As such, our list is typical of the situation faced by a biosecurity manager who will have a list of poorly understood, potentially invasive species recently detected for the first time, each requiring assessment and a recommendation regarding actions. In practice, these species will need to be compared with species that are already under management, however, for the purposes of demonstrating the method we illustrate the simpler problem.

(B) **Step 3: Estimate the species' risk**

As outlined above, numerous different methods can estimate the weed risk of an alien plant species (see Downey et al. 2010a; Leung et al. 2012; McGeoch et al. 2015). The most suitable choice of method is the one that estimates risk most accurately in a manner consistent with the objectives defined during step one. In the current example, because the policy objective related to 'wealth, wellbeing and biodiversity', we needed a tool designed to identify invasive species that may affect agriculture, the community and/or the environment. We chose the Australian Weed Risk Assessment (AWRA) method (Pheloung et al. 1999) as it is well understood, has been extensively validated (Caley et al. 2006; Gordon et al. 2008; Weber et al. 2009), and estimates risk in a manner consistent with our objective (Daehler and Virtue 2010; Pheloung et al. 1999). The AWRA scores used in this analysis were provided by the Australian Department of Agriculture and Water Resources.

210

211 The AWRA was designed as a pre-border method and omits extent of spread.
212 We multiplied its score by the species' 'potential distribution' to ensure the resulting
213 risk metric captured both the intensity and spatial extent of the impact (following the
214 generic framework set out in the introduction). We used the CLIMATCH method
215 (ABARES 2008) to simulate the likely maximum spatial extent of the species as the
216 sum of the probability of occurrence in each pixel (approximated by Euclidean
217 distance) multiplied by the pixel's area (see also Elith 2013; Guillera-Arroita et al.
218 2015). This approach does assume that species will occupy their entire niche and
219 incur impacts immediately, therefore, overestimating the risk of species that spread
220 slowly. However, this effect will be counteracted by the species' lower invasiveness
221 scores and should not substantially influence the outcome of the prioritisation.

222

223 (B) **Step 4: Identify possible management interventions**

224 Because we aimed to investigate the use of PPP to prioritize eradication targets, we
225 chose eradication as the default intervention. However, several methods have been
226 developed to identify the optimal management outcome for a species (such as Moore
227 et al. 2011; van Wilgen et al. 2011; Virtue et al. 2006) and, in practice, any could be
228 chosen. Like Joseph et al. (2009), we defined an intervention as the minimum set of
229 all necessary actions required to achieve the chosen outcome, including surveillance,
230 treatment, project management, research and communications.

231

232 (B) **Step 5: Estimate the intervention's effectiveness**

233 To assess the benefits of intervention, we estimated the proportion of the un-
234 managed risk that would be ameliorated by the intervention (McConnachie et al.
235 2015; Pannell and Gibson 2015) This often-overlooked step is required to determine

the potential value of the intervention in comparison to doing nothing, and is referred to as the counterfactual scenario (sensu Bull et al. 2014; Ferraro and Pattanayak 2006). Because all of our proposed interventions were eradication projects, which by definition require the complete elimination of the species (Newsom 1978), we assumed that the risk would be completely ameliorated. In this case the effectiveness is 1.0 (100%). However, other management strategies could be compared in this framework by altering the effectiveness score. For example, if the intervention aimed to contain the species to an area equivalent to 10% of its potential distribution, the effectiveness score would only be 0.90 (90%).

(B) **Step 6: Estimate the intervention's cost**

The cost of an eradication program given the defined planning horizon was estimated using a modified version of the bio-economic model described in Hester et al. (2013). Hester et al. (2013) estimate the costs of managing a plant eradication program from a stage matrix model that tracks the progression (and reversion) of infested areas between the active, monitored and eradicated states. The proportion of infested area that transitions between states is based on the size of the initially infested area and demographic parameters such as detection distance and propagule longevity. Using economic data sourced from previous plant eradication programs in both Queensland (included in Hester et al. 2013) and Victoria (unpublished), the costs of the management actions at each time step are then calculated based on the proportion of area at each stage.

We can also use the bio-economic model to estimate the accumulation of program costs over time, in parallel with the probability that the program would be successful over the same time period (Step 7, below). To do so, we modified the

original model of Hester et al. (2013) for better correspondence with Dodd et al.'s (2015) probability of intervention success, including aligning the units of the input variables (e.g., propagule longevity vs seedbank half-life) and accounting for minor differences in assumptions (e.g., multiple vs single annual monitoring events). These modifications are summarised in the supplementary material (Appendix S1).

(B) Step 7: Estimate the intervention's probability of success

The probability of an eradication attempt being successful given the defined planning horizon was then estimated using the Weibull hazard rate model described in Dodd et al. (2015). Hazard rate models describe how the rate of an event occurring (in this case eradication) varies with time as a function of a set of explanatory variables (Hosmer and Lemeshow 1999; Kleinbaum and Klein 2012). Using the demographic data collected during step two (infested area, propagule longevity etc.), the probability of success was calculated for each of the 50 hypothetical species incursions. However, because the probability of success depends on search effort (Dodd et al. 2015; Hester et al. 2013; McCarthy et al. 2008), it is important that the effort [resources] allocated to each intervention is optimised. As such, we iteratively repeated steps six and seven, selecting the monitoring rate [effort] that resulted in the highest probability of success per dollar spent.

(B) Step 8: State constraints

To investigate whether budget size influenced which projects were allocated resources, we repeated the prioritization for three separate budget allocations of \$1, \$2 and \$3 million per annum. Because the value of money declines with time, each budget was discounted at an annual rate of 6% to calculate its 'present value' (see Hester et al. 2013 for a detailed explanation), and the annual discounted values

summed to calculate the net present value of available resources. For example, an annual budget of \$2 million, discounted at 6% over 30 years leaves us with a total resource allocation of \$29,181,442. Based on the limited available data, budgets of these sizes are representative of programs conducted in Australia and New Zealand (Dodd et al. 2015; Harris and Timmins 2009; Panetta et al. 2011).

(B) Step 9: Choose optimal combination of interventions

The last step in the re-framed prioritization process was to calculate the expected cost-efficiency of each of the 50 eradication projects using the modified efficiency function (eqn. 2). Projects were then ranked by their cost-efficiency with resources allocated to projects according to their rank until the budget was exhausted. Where the next available project requires more resources than the remaining budget, it is skipped, with the process continuing until the budget is spent. This method of allocating resources is a robust approximate solution to what is known as a '0-1 knapsack problem' (Pannell and Gibson 2015). Unfortunately, calculating the optimal solution would require all 50 programs to be optimised at once [jointly optimised], and this remains computationally prohibitive at this time.

Finally, we compared the reduction in weed risk achieved by allocating resources using the re-framed PPP framework with the allocation based on weed risk alone. The cost-efficiency of the final portfolio allocation (summed over all interventions) was measured as summed benefit divided by summed cost. A flowchart and worked example of the full protocol is included in the supplementary material (Appendix S2).

313 Unless otherwise specified, all data processing and analyses were undertaken
314 in the R software environment for statistical computing and graphics (R Core Team
315 2013) with the following packages installed: raster (Hijmans and van Etten 2013)
316 and dismo (Hijmans et al. 2013) for species distribution modelling; adagio (Borchers
317 2015) for dynamic programming; ggplot2 (Wickham 2009) and scales (Wickham
318 2012) for plotting.

(A) **RESULTS**

(B) **Identifying the investment required for each intervention**

By selecting the monitoring rate that maximised the probability of success per dollar spent, we identified the optimal investment required for each intervention given the particular planning horizon (Table 1). For species with a low surveillance cost relative to the annual overhead, this was the maximum surveillance rate (3 visits per annum). However, where the inverse applied (high surveillance cost relative to the annual overhead) the most cost-efficient option was to do the minimum surveillance rate (1 per annum) and let the program run longer (Figure 1). Based on the scenarios used in our analysis, a rough rule of thumb was to monitor more frequently for trees, shrubs and tussock grasses and less frequently for sub-shrubs and herbs.

(B) **Selecting a planning horizon**

Once the optimal amount of effort required for each individual intervention was identified, we were then able to allocate resources to interventions using the re-framed prioritization process and compare the cost-efficiency of the various planning horizons and budgets. Figure 2 illustrates how the cost-efficiency of each portfolio varied over time for budgets of \$1M, \$2M & \$3M per annum. In all three budget scenarios, the highest efficiencies were achieved with a planning horizon between 20 and 30 years. However, this trend became less pronounced as the size of the budget increased and interventions with progressively lower cost-efficiency received funding (Table 1). Budget size had no influence on the level of priority assigned to individual interventions (data not shown).

Using the \$2M annual budget as an example, we show how the number of species targeted varied with the choice of planning horizon (Figure 3). Where the

planning horizon was very short (less than 15 years), the optimal solution was to invest more money (effort) in fewer species to maximise their probability of success. However, this resulted in both the lowest cost-efficiency and the fewest species funded (10) of all planning horizons. As the planning horizon increased, so too did the number of programs funded as the species became cheaper to eradicate in the time allowed. This trend existed irrespective of the choice of discount rate (data not shown). When viewed together, a planning horizon of thirty years was both highly cost-efficient and funded relatively many programs (Figures 2 & 3).

(B) Assessing the total budget against expected benefits

Having decided on a planning horizon (30 years), we returned to our budget to see how much of the possible benefit it was capturing. The total budget required to fund all 50 eradication programs, over 30 years, was \$260,288,640 (Figure 4). However, because the marginal benefit arising from each additional program is progressively lower (Table 1), almost 75% of the total possible benefit could be achieved with a resource allocation of only \$29,181,442 (~11% of total possible expenditure); which is equal to our \$2M annual budget. When interpreting these figures, it is important to remember that they indicate cost-efficiency (units of risk offset per dollar spent) rather than cost-benefit (dollars returned per dollar spent). As such, a slope of <1 does not indicate that the return is less than the amount spent because the dollar value of a unit of risk is unknown (though, it does indicate a relative decline in return).

(B) Comparison of priority setting methods

If we were to invest \$2M per annum over 30 years, our re-framed PPP method would allocate resources to 15 eradication programs with a combined cost-efficiency of

24.91 risk units per dollar (Figure 3). In contrast, if we were to allocate those same resources based solely on the ranks of the species' risk scores (risk ranking), only 7 eradication programs would receive resources with a combined cost efficiency of 21.46 risk units per dollar (Tables 1 & 2). Expressed as the reduction in total weed risk, the re-framed PPP reduced the risk by 726,444,766 units compared to 580,054,440 risk units when using the risk ranking method. As such, the PPP method improved the return on investment by 25%.

However, despite the differences in their expected benefits, the two portfolios of eradication programs were not substantially different (Table 2). Eight of the top ten highest-risk species all received funding using the PPP, with the remainder of the projects receiving funding being moderate risk species with particularly low eradication costs, such as *Limonium carolinanum*. The key difference was that the risk ranking method funded *Euphorbia hypericifolia* for eradication, which had a very high cost relative to its risk score (Table 1). As such, the resources allocated to its eradication were not available to be spent on slightly lower risk species with superior returns on investment (an opportunity cost).

(A) **DISCUSSION**

By allocating resources to plant eradication programs using the Project Prioritization Protocol (PPP) our analysis indicated that it is possible to improve the return on public expenditure by 25%. We demonstrate how the cost-efficiency of the overall portfolio is influenced by the choice of planning horizon; including the decline in portfolio performance that arises when attempting to eradicate species too quickly. Finally, we discuss the logistical benefits to a management agency that arise from the use of a generic overarching framework such as the PPP.

(B) **Improving the return on investment from plant eradication programs**

Eradication is often a cost-efficient approach to minimising the future impacts of invasive species (Harris et al. 2001; Wittenberg and Cock 2001). However, the extensive benefits arising from eradication depend on the program being successful and current evidence indicates that more than half of plant eradication programs fail (Howell 2012; Pluess et al. 2012; Rejmánek and Pitcairn 2002). As a consequence, biosecurity managers must make good choices when deciding to fund an eradication program, or risk squandering their limited financial resources for relatively little gain (Dodd et al. 2015; Panetta 2015; Panetta 2009).

The prevailing approach to tackling this issue has been to focus investment towards eradication programs considered to be 'feasible' given a 'realistic' amount of resources (Rejmánek and Pitcairn 2002). This paradigm is reflected in the large number of methods that prioritize species based on their risk and feasibility of coordinated control (see Darin et al. 2011; Hiebert 1997; Virtue et al. 2006). Most of these methods include cost as an indicator of feasibility based on the notion that feasibility and cost are inextricably linked (Panetta 2009). However, by doing so,

these methods are making judgements about what is 'realistic' without reference to risk.

By modelling these elements separately, the PPP method allows managers to consider whether the costs are proportional to the risk (hence, realistic), without clouding views of whether the eradication is likely to be successful (hence, feasible). In fact, by ranking species according to their cost-efficiency (Table 1), the PPP method allocates resources to the programs that are the most realistic; that is, they have the lowest cost relative to the risk given the probability of success. When applied to our incursion scenarios, this difference improved performance by 25%, compared to strategies that account for weed risk alone. The main difference is that the approach implemented here identifies species for which the cost of eradication was too high relative to their risk, even though the programs had a high probability of success.

Our results also indicate the importance of specifying a planning horizon before prioritizing possible interventions. Choosing a short planning horizon and attempting to eradicate individual species as quickly as possible, whilst intuitively appealing, substantially reduced the amount of risk offset by the overall portfolio. This occurred in time horizons less than 15 years where the marginal benefit arising from additional investment in individual species was high because their probabilities of success were still relatively low. In our case, choosing a medium-term planning horizon (20-30 years) improved both the cost-efficiency of the portfolio and the number of species interventions funded. Unfortunately, many agencies won't have the flexibility to take such a long-term view. Our advice in this circumstance is not to lose focus of the long-term goals when making short-term decisions. In particular,

specifying the time horizon *a priori* may help communicating to stakeholders that long-term commitment is required for programs to be successful (Dodd et al. 2015).

(B) The benefit of a generic framework with interchangeable inputs

An additional advantage of modelling cost and feasibility separately in the PPP method is that we are left with well-defined and measurable inputs that can be validated through observation and refined over time. For example, if we repeatedly estimate program costs, we can verify the initial predictions and refine the cost model accordingly (Dodd et al. 2015). In addition, because each of the inputs encompass research disciplines of their own, it is likely that more accurate methods will be developed in the future. By using a generic overarching framework such as PPP, a management agency can more easily substitute methodologies (such as the cost or risk assessment models) depending on agency preferences (Hulme 2012).

In the instance that an agency does not have sufficient data to accurately model inputs, such as the probability that a program will be successful, expert judgements may be used to estimate the required variables (Panetta 2009). Expert judgement is already frequently used in this context (Carwardine et al. 2012; Firn et al. 2015). However, structured elicitation methods (such as Martin et al. 2012; Speirs-Bridge et al. 2010) are typically under-utilised by decision makers, even though structured group processes are well suited to estimating the probabilities of events occurring in a defined time frame, and consistently outperform the best expert for only modest costs of time and resources (Burgman 2015; Sutherland and Burgman 2015).

The use of group judgements also allows for the calculation of uncertainty intervals (Speirs-Bridge et al. 2010). This is vitally important as one of the realities of decision making in invasive species management is the high level of uncertainty surrounding the estimates (Hulme 2012). One of the benefits of the existing approaches that we used as inputs is that they can each generate uncertainty intervals (see Caley et al. 2006; Dodd et al. 2015; Panetta et al. 2011). For the purposes of demonstrating the potential of the PPP method, here we illustrate its application using best estimates. However, in practice, approaches such as portfolio theory (see Akter et al. 2015; Yemshanov et al. 2014) could be used to incorporate uncertainty into the framework.

(B) Prioritization in the management setting

As highlighted by Brunel et al. (2010), the output of any quantitative prioritization process forms the input to subsequent discussions amongst the various stakeholders who ultimately decide on the resource allocation (see also Kumschick et al. 2012). With that in mind, one of the strengths of the PPP, compared with recent approaches that use the Analytic Hierarchy Process (Darin et al. 2011; Forsyth et al. 2012), is that the intermediate estimates are tangible. This means that stakeholders can compare interventions on the basis of their estimated costs and probabilities of success, rather than the primary inputs (such as propagule longevity and infested area), when trying understand why one intervention was prioritized over another. Our experience as advisors to government over several decades suggests that decision makers are more comfortable with prioritization when they can see this additional layer of information, and are more likely to question the intermediate estimates than the priorities themselves.

493 Whilst we have made much of the technical considerations of prioritization, in
494 practice, socio-political factors also need to be considered as part of any prioritization
495 exercise (Estévez et al. 2013; Moon et al. 2015; van Wilgen and Richardson 2014).
496 Trade-offs will often need to be made, particularly where the species under
497 consideration has a positive economic value (De Wit et al. 2001; van Wilgen and
498 Richardson 2014; Virtue et al. 2004) but may harm health or the environment (water
499 hyacinth (*Eichhornia crassipes*) and maritime pine (*Pinus pinaster*) are examples).
500 Several risk assessment tools have developed approaches to incorporate this into
501 their estimate of overall weed risk (Kumschick et al. 2012; Robertson et al. 2003).
502 Alternatively, whilst an intervention may be economically rational, it may not be
503 socially acceptable (Estévez et al. 2013; Moon et al. 2015). In these circumstances,
504 the species may be regarded as 'unsuitable' for eradication, despite its feasibility (see
505 also Dodd et al. 2015; Panetta 2009; Panetta and Timmins 2004).

506
507 The final challenge facing post-border biosecurity agencies is how to re-
508 prioritize eradication resources given the ongoing arrival of new target species.
509 Eradication programs often take decades to be successfully completed (Dodd et al.
510 2015; Mack and Lonsdale 2002; Panetta and Lawes 2005), making the resource
511 allocation highly illiquid. Where resources are constrained, a manager must decide
512 whether the new species is a higher priority for eradication than the existing species
513 and re-allocate resources accordingly. Deciding not to fund, or to withdraw from, an
514 eradication program is always highly controversial. Therefore, an important
515 extension to our work would be to develop robust switching rules to help guide
516 decision makers in this situation.

517

518 For interventions such as eradication to remain a viable option, managers
519 must continue to improve the quality of decisions (Dana et al. 2014). That means
520 using rational, transparent and repeatable processes, such as the PPP, to support
521 decision making (Game et al. 2013; Heikkilä 2011b). It also includes making
522 decisions using inputs that can be validated so that the accuracy of assessments
523 improves (Dodd et al. 2015). Neither of these approaches prevents us from
524 continuing to manage species based on the magnitude of their weed risk (sensu
525 Blackburn et al. 2014; Kumschick et al. 2012; McGeoch et al. 2015). Rather, they
526 help focus attention on those species where management can make biggest
527 difference with a realistic amount of resources (sensu Rejmanek and Pitcairn 2002).
528
529

530 (A) **SUPPORTING INFORMATION**

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

532 **Appendix S1** Description of revisions made to the Hester et al. (2013) model.

533 **Appendix S2** Worked example of the PPP framework.

534

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(A) LITERATURE CITED

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Table 1 Summary of the input metrics that were used to calculate the cost-efficiency for eradicating the 50 hypothetical species incursions over a planning horizon of 30 years.

Botanic name	Family	Efficiency (risk/\$)	AWRA Score	Potential Distribution (sq. km)	Risk	Effectivn.	Area (ha)	Cost (\$)	Pr(E t)
<i>Limonium carolinianum</i>	Plumbaginaceae	129.17	7	3,579,640	25,057,480	1.00	16	193,987	1.00
<i>Amaranthus palmeri</i>	Amaranthaceae	102.41	29	5,146,440	149,246,760	1.00	241	1,450,085	1.00
<i>Phleum alpinum</i>	Poaceae	82.56	11	3,644,200	40,086,200	1.00	49	463,700	0.96
<i>Miscanthus ecklonii</i>	Poaceae	48.49	15	3,964,680	59,470,200	1.00	339	1,226,481	1.00
<i>Zygophyllum fabago</i>	Zygophyllaceae	46.30	25	3,995,400	99,885,000	1.00	331	2,131,394	0.99
<i>Festuca flavescens</i>	Poaceae	27.86	9	1,840,400	16,563,600	1.00	140	594,477	1.00
<i>Teucrium canadense</i>	Lamiaceae	23.53	22	4,297,280	94,540,160	1.00	499	4,009,455	1.00
<i>Sedum anacampseros</i>	Crassulaceae	19.49	9	1,955,320	17,597,880	1.00	155	873,806	0.97
<i>Penstemon superbus</i>	Scrophulariaceae	13.78	12	3,826,600	45,919,200	1.00	457	3,295,792	0.99
<i>Oxalis stricta</i>	Oxalidaceae	13.24	23	4,513,280	103,805,440	1.00	706	7,510,312	0.96
<i>Silene baccifera</i>	Caryophyllaceae	12.57	15	2,353,720	35,305,800	1.00	342	2,412,209	0.86
<i>Chromolaena bigelovii</i>	Asteraceae	12.20	5	2,572,640	12,863,200	1.00	213	1,052,605	1.00
<i>Leontopodium nanum</i>	Asteraceae	10.29	5	2,694,920	13,474,600	1.00	215	1,292,911	0.99
<i>Arabis collina</i>	Brassicaceae	10.26	8	3,053,440	24,427,520	1.00	327	2,269,285	0.95
<i>Teucrium orientale</i>	Lamiaceae	10.12	10	2,602,560	26,025,600	1.00	375	2,566,473	1.00
<i>Geranium pylzowianum</i>	Geraniaceae	10.02	3	1,612,720	4,838,160	1.00	64	388,515	0.81
<i>Taraxacum albidum</i>	Asteraceae	9.94	12	1,172,280	14,067,360	1.00	223	1,349,000	0.95
<i>Saxifraga cymbalaria</i>	Saxifragaceae	8.07	11	2,287,360	25,160,960	1.00	102	1,527,039	0.49
<i>Penstemon oliganthus</i>	Scrophulariaceae	7.41	5	2,324,920	11,624,600	1.00	220	1,315,986	0.84
<i>Euphorbia hypericifolia</i>	Euphorbiaceae	5.95	15	4,495,560	67,433,400	1.00	882	11,266,384	0.99
<i>Marrubium globosum</i>	Lamiaceae	4.00	7	1,140,880	7,986,160	1.00	274	1,854,578	0.93
<i>Balsamorhiza sagittata</i>	Asteraceae	3.82	2	1,896,960	3,793,920	1.00	187	989,294	1.00
<i>Symphoricarpos oreophilus</i>	Caprifoliaceae	3.67	9	4,097,600	36,878,400	1.00	863	10,013,911	1.00
<i>Geranium collinum</i>	Geraniaceae	3.67	2	3,343,960	6,687,920	1.00	261	1,656,467	0.91
<i>Myrica nana</i>	Myricaceae	3.58	1	1,236,920	1,236,920	1.00	122	345,532	1.00
<i>Primula maximowiczii</i>	Primulaceae	2.73	3	766,520	2,299,560	1.00	142	797,105	0.95
<i>Silene rupestris</i>	Caryophyllaceae	2.63	9	2,228,120	20,053,080	1.00	676	6,986,469	0.92
<i>Potentilla matsumurae</i>	Rosaceae	2.32	7	1,312,720	9,189,040	1.00	419	3,257,281	0.82
<i>Lachenalia youngii</i>	Hyacinthaceae	1.80	4	2,830,000	11,320,000	1.00	581	5,438,233	0.87
<i>Phyllodoce aleutica</i>	Ericaceae	1.66	1	1,767,280	1,767,280	1.00	198	1,061,306	1.00
<i>Cytisus decumbens</i>	Fabaceae	1.56	7	1,703,440	11,924,080	1.00	675	7,178,830	0.94
<i>Phyllodoce empetriformis</i>	Ericaceae	1.49	2	1,305,760	2,611,520	1.00	288	1,745,146	1.00
<i>Coronilla coronata</i>	Fabaceae	1.41	9	2,165,840	19,492,560	1.00	990	13,740,477	0.99
<i>Silene caroliniana</i>	Caryophyllaceae	1.23	10	1,406,520	14,065,200	1.00	844	11,384,185	0.99
<i>Plantago camtschatica</i>	Plantaginaceae	0.70	12	585,400	7,024,800	1.00	822	9,712,129	0.97
<i>Sedum ewersii</i>	Crassulaceae	0.67	7	853,880	5,977,160	1.00	718	8,765,401	0.98
<i>Saxifraga canaliculata</i>	Saxifragaceae	0.66	6	1,367,560	8,205,360	1.00	234	3,630,022	0.29
<i>Saxifraga jacquemontiana</i>	Saxifragaceae	0.51	4	2,413,760	9,655,040	1.00	468	8,249,528	0.44
<i>Lithodora oleifolia</i>	Boraginaceae	0.49	3	1,709,000	5,127,000	1.00	857	9,971,677	0.96
<i>Arnica lessingii</i>	Asteraceae	0.41	8	101,600	812,800	1.00	259	1,639,544	0.83
<i>Saxifraga pentadactylis</i>	Saxifragaceae	0.30	7	2,528,920	17,702,440	1.00	940	21,036,237	0.36
<i>Tofieldia pusilla</i>	Liliaceae	0.26	5	525,280	2,626,400	1.00	826	9,793,046	0.96
<i>Prosopis tamarugo</i>	Fabaceae	0.24	9	97,360	876,240	1.00	517	3,638,828	1.00
<i>Phyllodoce caerulea</i>	Ericaceae	0.22	2	385,280	770,560	1.00	455	3,460,787	1.00
<i>Jasione crispa</i>	Campanulaceae	0.19	1	3,403,240	3,403,240	1.00	895	16,454,096	0.93
<i>Diospyros melanoxylon</i>	Ebenaceae	0.18	4	148,600	594,400	1.00	611	3,268,166	1.00
<i>Primula magellanica</i>	Primulaceae	0.17	2	193,800	387,600	1.00	327	2,269,285	0.99
<i>Geum reptans</i>	Rosaceae	0.06	1	1,143,080	1,143,080	1.00	950	17,999,888	0.89
<i>Saxifraga hookeri</i>	Saxifragaceae	0.00	7	0	0	1.00	725	14,636,619	0.38
<i>Potentilla miyabei</i>	Rosaceae	0.00	4	0	0	1.00	877	12,124,677	0.94

Table 2 Species interventions ranked by cost-efficiency in comparison to other prioritization methods. Prioritisation methods include ranking by: cost-efficiency, risk, cost and the probability of success. Values are shown for each of the 50 hypothetical incursion scenarios over a planning horizon of 30 years.

Botanic name	Family	Efficiency Rank	AWRA Rank	Risk Rank	Cost Rank	Pr(Succ.) Rank
<i>Limonium carolinianum</i>	Plumbaginaceae	1*	23	13*	1	1
<i>Amaranthus palmeri</i>	Amaranthaceae	2*	1	1*	15	15
<i>Phleum alpinum</i>	Poaceae	3*	11	8*	4	27
<i>Miscanthus ecklonii</i>	Poaceae	4*	5	6	11	4
<i>Zygophyllum fabago</i>	Zygophyllaceae	5*	2	3*	21	21
<i>Festuca flavescens</i>	Poaceae	6*	15	19	5	3
<i>Teucrium canadense</i>	Lamiaceae	7*	4	4*	32	12
<i>Sedum anacampseros</i>	Crassulaceae	8*	19	18	7	24
<i>Penstemon superbus</i>	Scrophulariaceae	9*	8	7	28	20
<i>Oxalis stricta</i>	Oxalidaceae	10*	3	2*	36	26
<i>Silene baccifera</i>	Caryophyllaceae	11*	7	10	24	41
<i>Chromolaena bigelovii</i>	Asteraceae	12*	31	23	9	8
<i>Leontopodium nanum</i>	Asteraceae	13*	32	22	12	22
<i>Arabis collina</i>	Brassicaceae	14*	21	14	22	31
<i>Teucrium orientale</i>	Lamiaceae	15	13	11	25	10
<i>Geranium pylzowianum</i>	Geraniaceae	16*	41	35	3	45
<i>Taraxacum albidum</i>	Asteraceae	17	10	20	14	30
<i>Saxifraga cymbalaria</i>	Saxifragaceae	18	12	12	16	46
<i>Penstemon oliganthus</i>	Scrophulariaceae	19	34	25	13	42
<i>Euphorbia hypericifolia</i>	Euphorbiaceae	20	6	5*	43	16
<i>Marrubium globosum</i>	Lamiaceae	21	26	30	20	36
<i>Balsamorhiza sagittata</i>	Asteraceae	22	44	36	8	13
<i>Symphoricarpos oreophilus</i>	Caprifoliaceae	23	17	9	42	14
<i>Geranium collinum</i>	Geraniaceae	24	46	32	18	38
<i>Myrica nana</i>	Myricaceae	25	47	42	2	2
<i>Primula maximowiczii</i>	Primulaceae	26	40	40	6	32
<i>Silene rupestris</i>	Caryophyllaceae	27	20	15	34	37
<i>Potentilla matsumurae</i>	Rosaceae	28	27	28	26	44
<i>Lachenalia youngii</i>	Hyacinthaceae	29	37	26	33	40
<i>Phyllodoce aleutica</i>	Ericaceae	30	48	41	10	9
<i>Cytisus decumbens</i>	Fabaceae	31	25	24	35	33
<i>Phyllodoce empetrififormis</i>	Ericaceae	32	42	39	19	7
<i>Coronilla coronata</i>	Fabaceae	33	18	16	46	17
<i>Silene caroliniana</i>	Caryophyllaceae	34	14	21	44	19
<i>Plantago camtschatica</i>	Plantaginaceae	35	9	31	39	25
<i>Sedum ewersii</i>	Crassulaceae	36	24	33	38	23
<i>Saxifraga canaliculata</i>	Saxifragaceae	37	30	29	30	50
<i>Saxifraga jacquemontiana</i>	Saxifragaceae	38	38	27	37	47
<i>Lithodora oleifolia</i>	Boraginaceae	39	39	34	41	29
<i>Arnica lessingii</i>	Asteraceae	40	22	45	17	43
<i>Saxifraga pentadactylis</i>	Saxifragaceae	41	29	17	50	49
<i>Tofieldia pusilla</i>	Liliaceae	42	33	38	40	28
<i>Prosopis tamarugo</i>	Fabaceae	43	16	44	31	6
<i>Phyllodoce caerulea</i>	Ericaceae	44	43	46	29	11
<i>Jasione crispa</i>	Campanulaceae	45	49	37	48	35
<i>Diospyros melanoxylon</i>	Ebenaceae	46	35	47	27	5
<i>Primula magellanica</i>	Primulaceae	47	45	48	23	18
<i>Geum reptans</i>	Rosaceae	48	50	43	49	39
<i>Saxifraga hookeri</i>	Saxifragaceae	49	28	49	47	48
<i>Potentilla miyabei</i>	Rosaceae	50	36	50	45	34

* Species selected with a fixed budget of \$29,181,442

Fig. 1 Influence of species' detectability on the optimal choice of monitoring rate. Two example species incursions are shown: one with higher detectability (*Miscanthus ecklonii*) and another with lower detectability (*Zygophyllum fabago*).

Fig. 2 Predicted cost-efficiency of the three different budget allocations (\$1M, \$2M and \$3M per annum) as a function of the planning horizon (10-50 years). Points shown are 5 year intervals.

Fig. 3 Predicted cost-efficiency versus the number of species' eradications funded with a \$2M budget allocation as a function of the planning horizon (10-50 years). Points shown are 5 year intervals.

Fig. 4 Cumulative benefits from the full range of possible budget allocations over a 30 year planning horizon given the 50 hypothetical incursion scenarios.