Biol Invasions (2011) 13:1241–1253
DOI 10.1007/s10530-011-9960-7

ORIGINAL PAPER

A systematic approach for prioritizing multiple management actions for invasive species

S. R. Januchowski-Hartley · P. Visconti · R. L. Pressey

Received: 7 December 2009 / Accepted: 10 November 2010 / Published online: 12 February 2011

Abstract The successful management and eradication of invasive species is often constrained by insufficient or inconsistent funding. Consequently, managers are usually forced to select a subset of infested areas to manage. Further, managers may be unaware of the most effective methods for identifying priority areas and so are unable to maximize the effectiveness of their limited resources. To address these issues, we present a spatially explicit decision method that can be used to identify actions to manage invasive species while minimizing costs and the likelihood of reinvasion. We apply the method to a real-world management scenario, aimed at managing an invasive aquatic macrophyte, olive hymenachne (*Hymenachne amplexicaulis*), which is one of the most threatening invasives in tropical Australia, affecting water quality, freshwater biodiversity, and fisheries.

S. R. Januchowski-Hartley (✉) · P. Visconti · R. L. Pressey
Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, QLD 4811, Australia
e-mail: stephierenee@gmail.com

S. R. Januchowski-Hartley
School of Marine and Tropical Biology, James Cook University, Townsville, QLD 4811, Australia

P. Visconti
Department of Human and Animal Biology, University La Sapienza, Rome, Italy

Keywords Connectivity · Decision theory · *Hymenachne amplexicaulis* · Invasive species management · Management cost · Marxan · Systematic conservation planning

Introduction

There are high economic costs associated with managing invasive species infestations to eradicate them or reduce the probability of their reinvasion. However, the costs of managing the spread of and damage resulting from invasive species are rarely accounted for when making management decisions or when allocating funding to management programs (Simberloff 2009). Costs vary depending on the species concerned, the ecosystem it has invaded, and the methods used to manage it. Management costs of a single well-established species can reach millions of dollars. For example, in South Africa, the total cost of clearing 15 invasive tree species that occupied almost 17 million hectares of forestry lands was estimated at \$60 million USD/year over 20 years (van Wilgen et al. 2001). However, this estimate was reduced by \$30 million USD/year by recognizing that several of the species did not affect natural systems and by considering multiple management actions. This study demonstrated the financial benefits that can be gained from (1) using spatially explicit planning strategies, (2) considering multiple management actions, and (3) accounting for the costs associated with different actions.

Methods used by managers to identify management priorities for invasive species typically consider the values of biodiversity and ecosystems affected, but few methods account for the costs associated with management actions (e.g., Gosper and Vivian-Smith 2006). By costs we mean the financial requirements of applying management actions, including salaries, travel costs, and purchase or lease of materials and capital equipment. Prioritization methods commonly used by pest managers range from "best guesses" to more explicit methods such as: (1) the use of geographic information systems to overlay different natural values (e.g., Roura-Pascual et al. 2009a) and (2) decision-support methods that use scoring approaches to combine a range of criteria (e.g., Ratcliffe et al. 2009; Roura-Pascual et al. 2009b). We refer hereafter to these methods as non-systematic because they do not account for one or both of two key characteristics: (1) explicit, quantitative objectives for control of invasive species; and (2) the contribution each action or set of actions makes towards achieving an objective.

The large and expanding literature on systematic conservation planning describes methods that have two characteristics in common (Margules and Pressey 2000): (1) explicit and usually quantitative objectives and (2) assessment of areas or area-specific actions according to their relationship with other areas and actions. Explicit, quantitative objectives inform managers about the local, regional or national significance of their investments, make available a powerful array of decision-theory methods for solving spatial problems, and allow managers to measure progress through time (e.g., Carwardine et al. 2008; Segan et al. 2010). Two typical aspects of context-dependent assessment are complementarity and connectivity, so that the contribution of actions and areas depends on the features shared between areas and the spatial relationships between areas (Magurran 2004; Margules and Pressey 2000; Nicholls and Margules 1993; Sarkar et al. 2006; Vane-Wright et al. 1991). The use of complementarity is considered the main common denominator of systematic methods, reducing unwanted redundancy of conservation actions and improving the cost-effectiveness of solutions to conservation problems (Justus and Sarkar 2002; Margules and Pressey 2000). Connectivity, which influences the likelihood of invasion and reinvasion from source populations, is a particularly important consideration for riverine and wetland ecosystems. Without spatial considerations regarding complementarity and connectivity it is difficult to identify the best configurations of areas to achieve a set of objectives (e.g., Beger et al. 2010; Hermoso et al. 2010).

To address the limitations of non-systematic methods for prioritizing management of invasives, we propose a method that can be used to identify spatially explicit configurations of areas to implement management actions. We aim to minimize management costs and the likelihood of reinvasion. We demonstrate the application of this method to identifying management priorities for an invasive aquatic macrophyte, olive hymenachne (*Hymenachne amplexicaulis*), which has spread across tropical northern Australia. This work was developed and carried out in collaboration with the manager responsible for reducing the spread of olive hymenachne in the Tully-Murray catchment in the Wet Tropics bioregion of northern Queensland. We present the results of five management prioritization scenarios. Each scenario has unique parameters and objectives aimed at reducing the area infested with olive hymenachne. Using the modeled solutions for each of the five scenarios, we illustrate: (1) the higher cost-effectiveness of systematic planning, relative to non-systematic methods, to prioritize management actions for invasive species, (2) the higher cost-effectiveness and reduced likelihood of reinvasion when accounting for connectivity between areas in the decision method, and (3) the reduction in total cost when setting objectives across all planning units in the study area rather than setting objectives only for certain ecosystem types within the study area.

Methods

Study species

Olive hymenachne is a robust rhizomatous perennial grass native to seasonally flooded environments along river banks in tropical and subtropical wetlands of America (Gordon and Yasmira 2007). Olive hymenachne was introduced as a commercial cultivar to Queensland, Australia in the early 1970s. It was first reported invading agricultural drains and natural waterways in northern Queensland in the late 1980s. Subsequently, the plant has invaded freshwater ecosystems across northern Australia causing major environmental impacts in wetlands and streams in much of eastern Queensland and parts of the Northern Territory and northern New South Wales.

Olive hymenachne commonly grows from 1 to 2.5 m tall, with leaves 10–45 cm long and up to 3 cm wide. It can be distinguished from Australia's native *Hymenachne* species by the characteristic stem-clasping leaf bases (Sydes 2009). Olive hymenachne produces large numbers of viable seeds that germinate readily on waterlogged soil. In northern Australia, flowering occurs between September and May, with most flowering occurring between mid-April and May. However, in the Wet Tropics bioregion, olive hymenachne is reported to flower up to three times a year, outside of the typical flowering seasons (D. Sydes, Cassowary Coast Regional Council, personal communication).

Olive hymenachne develops roots at each node along the stolon when contact is made with moist soil. Secondary dispersal, or floating, of olive hymenachne stolons in flood waters is one of the major factors causing the spread of the plant during annual flooding events typical of northern Australia. The seed bank of olive hymenachne remains viable for up to 10 years (Sydes 2009). To ensure complete removal of olive hymenachne and its seed bank, infested areas in the Wet Tropics region are managed for up to 15 years.

Dense stands of olive hymenachne can modify water flow and watercourses. Floating mats become dislodged in floods and are easily transported downstream. Weed mats can damage infrastructure, clog small streams, and block fish passage (Houston and Duivenvoorden 2002). The establishment of olive hymenachne can fundamentally change vegetation structure by displacing emergent and submergent native species. Invasions of the species dramatically reduce diversity of native aquatic plant, invertebrate and fish assemblages, and support increased abundances of introduced fish species (Houston and Duivenvoorden 2002).

Study area

The Tully-Murray catchment is located in the Wet Tropics bioregion of north Queensland (Fig. 1).

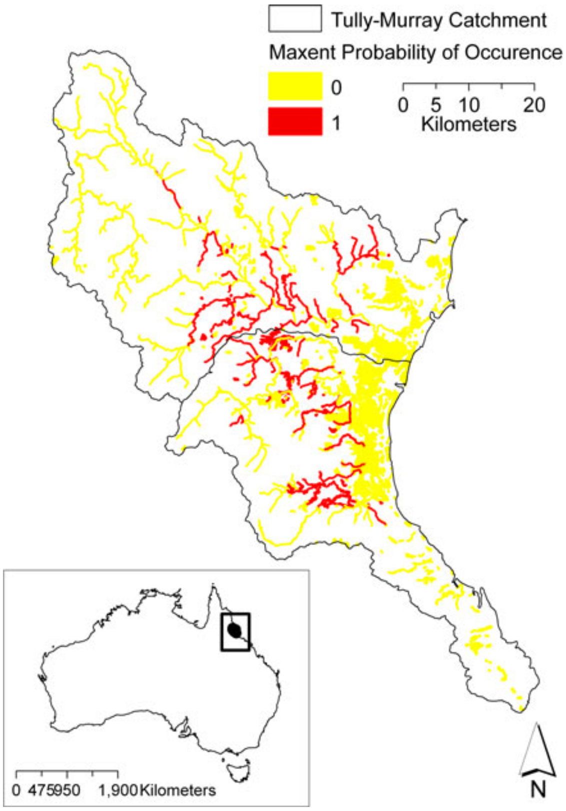

Fig. 1 Location of the Tully-Murray study area in north Queensland, Australia. Across the 1353 planning units, values of zero indicate that olive hymenachne (*Hymenachne amplexicaulis*) is not predicted to be present. Areas mapped with a value of 1 are those where the species currently occurs or is predicted to occur by the Maxent model. Presence is defined by a threshold probability of occurrence, $P \geq 0.30$

The region is characterized by a series of mountain ranges running roughly parallel to the coast, and mostly covered in tropical rainforest. Mean annual rainfall ranges from 4,000 mm near the coast to 1,200 mm at the western extremity. Mean daily temperatures on the coast range from a maximum of 31°C to a minimum of 18°C. The uplands are cooler, with mean daily temperatures ranging from 28 to 9°C (Kemp et al. 2007).

Most of the lowland landscape has been cleared for agriculture, in particular sugar cane production (Armour et al. 2004; Kemp et al. 2007; Tracey 1982). These highly modified seasonally inundated floodplain landscapes appear to favor the adaptations of olive hymenachne, which persists even under extreme flood conditions and highly eutrophic environments not suitable for native aquatic macrophytes.

The most vigorous stands of olive hymenachne have been observed where nutrient-enriched cane-field run-off is impounded and in native wetlands surrounded by disturbed agricultural landscapes (Sydes 2009).

We identified 1,353 wetlands and stream reaches across the Tully-Murray study region, and defined these reaches as our planning units. Wetlands were mapped at a scale of 1:100,000 and classified by the Queensland Department of Environment and Resource Management (DERM 2009). Discrete wetlands were identified as single planning units regardless of their size. Stream reaches (sections of streams between confluences) were delineated from a digital elevation model with a spatial resolution of 30 m (Januchowski et al. in press) using ArcHydro 1.3 (Maidment 2002). We applied a 5 m buffer to each side of the defined stream reaches using ArcGIS 9.2 (ESRI 2006).

Species distribution modeling

We used Maxent (Phillips et al. 2006) for all distribution modeling. Maxent is based on a maximum entropy algorithm for the prediction of species' potential geographical distributions. It has been shown to be generally better than other modeling methods for predicting species occurrences, and deals well with the small sample sizes commonly available for species distribution modeling (Elith et al. 2006; Hernandez et al. 2006).

We collated presence/absence data (47 total observations) for olive hymenachne from local managers and researchers in the Tully-Murray catchment. Presence/absence data were collected from sites with a range of environmental conditions. Olive hymenachne has been present in the landscape for over 15 years, and has likely reached the potential range of environmental characteristics present in the catchment (D. Sydes, Cassowary Coast Regional Council, personal communication). Therefore, even if the distribution of olive hymenachne is still expanding, it is not likely to encounter different environmental conditions from those behind the expansion front.

Presence was determined in planning units where: (1) managers are currently managing infestations or (2) infestations are known but there is no active management. The training dataset consisted of 36 presence points from the 47. The remaining 11 points were known absences. We used the 11 known absences from the field surveys and generated an additional 100 background points, which were regarded as a random sample from the sampling distribution. We located the background points with Hawth's Tools in ArcGIS 9.2 (ESRI 2006) and decided on 100 as sufficient to represent the distribution of environmental conditions in the study area (Phillips and Dudik 2008). As potential predictors, we used a combination of landscape-scale environmental variables that, based on advice from managers, have the most influence on the probability of olive hymenachne infestation: (1) presence of intensive agriculture (as a surrogate for areas of high nutrient input), (2) presence of low-intensity grazing, (3) wetland type, (4) elevation, and (5) average foliage projective cover of woody vegetation. All environmental data were analyzed in 30-m resolution spatial layers.

Maxent produces spatial predictions of probability of presence (hereafter referred to as probability of infestation) as logistic output values ranging from 0 (not suitable) to 1 (most suitable; Phillips et al. 2006). Probability of infestation was determined, as a function of the environmental variables, for each 30×30 m cell within the study area. We used Zonal Statistics in ArcGIS 9.2 to determine the average environmental suitability of each of our planning units, based on the 30×30 m cells falling within planning units. We converted the logistic output value estimated by Maxent to a binary estimate of infested or non-infested planning units using the 'equal training sensitivity and specificity threshold' which is automatically reported by Maxent. This threshold has been used in other studies (e.g., Cantor et al. 1999), and has been shown to produce low false positives and negatives (<20%; Liu et al. 2005). Based on this threshold, we considered only planning units with $P \geq 0.30$ (where P is the probability of infestation) as either currently supporting or having the potential to support olive hymenachne (Fig. 1). We also assumed that planning units potentially supporting olive hymenachne could act as sources of infestation as did planning units with known infestations. Based on the local manager's knowledge of the species, this was a reasonable assumption, given the species' rapid spread across the study area since it first established there (D. Sydes, Cassowary Coast Regional Council, personal communication).

We used the cross-validation setting in Maxent to validate our model. The cross-validation procedure divides the modeled data into replicate folds. We used a leave-one-out cross validation. For evaluation of model performances we used the threshold-independent receiver operating characteristic (ROC) which addresses false negative and false positive predictions. The ROC is quantified by the area under the curve (AUC), with values ranging from 0 to 1 and high values indicating good performance. An AUC > 0.6 is usually defined as acceptable model performance (Fielding and Bell 1997). The average modelling performance of the cross-validated Maxent models was 0.954.

Estimating area infested

We did not have access to relative abundance estimates for all planning units. Therefore, we estimated the "potential" area infested (ha) by olive hymenachne in all planning units using data on the proportion of area infested by olive hymenachne in managed ($n = 46$) and non-managed ($n = 29$) planning units. We regressed values of proportion infested against the probability of olive hymenachne occurrence (as determined by Maxent) using quantile regression. We chose this method because the variance of the proportion infested increased with increasing probability of occurrence. Quantile regression is robust to outlying data values and skewed data distributions (Cade et al. 1999). Most of our observations were from planning units with large proportions infested and both low and high probabilities of occurrence, with fewer observations from planning units with moderate to low proportions infested.

By fitting quantiles of y (proportion of area infested) as a function of x (probability of occurrence), quantile regression estimates the position of the edge of a triangular data scatter (Johnson and VanDerWal 2009). The magnitude of the slope of each regression line is a measure of the size of the relationship between probability of occurrence and proportion infested. We determined the quantile that best described this relationship using a goodness-of-fit measure for quantile regressions (e.g., VanDerWal et al. 2009). We then used this quantile to predict the proportions infested for all managed and non-managed planning units using the modeled probabilities of occurrence. We multiplied the proportions by the areas of planning units to derive areas infested.

Management costs

We identified three types of costs associated with management of olive hymenachne in the Tully-Murray catchment. These were based on advice from the strategic coordinator at Cassowary Coast Regional Council (CCRC) who is responsible for decisions on pest management across the study area. We identified the most appropriate management action for each planning unit depending on the wetland type, size of the stream reach, and soil type. We considered the following three actions: (1) high volume spraying (HV), (2) HV + Argo (amphibious machine), and (3) revegetation.

The total cost of taking action in planning unit u (d_u) was a function of the amount of time spent managing a planning unit h_u, the number of people managing m_u and the fuel cost incurred in travelling from CCRC's depot t_u. The equation defining the total management cost of a single planning unit u was:

$$d_u = w_u * m_u * (h_u + 2) + t_u \tag{1}$$

All variables are described in Table 1.

We had information on the number of hours spent undertaking management actions (h) for only a sample of managed planning units. To extrapolate this information to all planning units, we estimated the relationship between area infested with olive hymenachne and the amount of time spent managing a planning unit (actual management times recorded by CCRC managers). We did this for the HV (Eq. 2) and HV + Argo (Eq. 3) actions by fitting a linear regression model. In both models, the r^2 value was 0.73. The best fitting models were as follows:

$$\text{HVTime}\ (h) = 0.45 * \text{area infested} + 11 \tag{2}$$

$$\text{HVArgoTime}\ (h) = 0.66 * \text{area infested} + 2.9 \tag{3}$$

The cost associated with revegetation was not dependent on time because of the involvement of volunteers in this work. Rather, the number of plants to be purchased was the major driver of cost associated with this action. We used the cost incurred by CCRC for purchasing plants ($4,000 AUD/ha).

Objective function

Our objective was to reduce the area infested with olive hymenachne to a specified level, while

Table 1 Variable descriptions for equations 1 and 4

Equation 1	
d_u	Cost of undertaking a management action in the planning unit u
w	Cost/hour of work per person
m	Number of people involved in the action
h	Number of hours spent managing
t	Fuel cost of travelling to the planning unit from the Cassowary Coast Regional Council's depot
2	A constant representing time worked but not spent actively managing
Equation 4	
d_u	Cost of undertaking a management action in the planning unit u
a_{uf}	Amount of hymenachne infestation in planning unit u with feature type f. For the scenarios focused on all wetland types there was just one objective and one associated penalty. For scenarios focused on the two specified wetland types, f indicated the wetland type.
c_{ij}	Presence of a connection between planning unit i and planning unit j where i is upstream of j
x	Decision binary variable taking the value of 1 for planning units that are currently managed or are selected for management in the prioritization process
p_f	Penalty factor for feature f applied to the objective function to ensure the objective is met regardless of the total cost of taking management action
r_f	The cost of meeting the objective for each feature f
T_f	Objective for feature f, established as a percentage reduction in area covered by feature f infested with olive hymenachne
b	Boundary length modifier. A weighting factor used in Marxan to give different weight to the connectivity component (c_{ij}) in the objective function

minimizing management costs and the number of connections (our proxy for probability of reinfestation) for each selected planning unit. The objective function was minimized using the simulated annealing algorithm in Marxan conservation planning software (Ball et al. 2009). Simulated annealing can find many close-to-optimal solutions for a multi-criteria objective function and can solve large problems (involving up to hundreds of thousands of planning units and features[1]) in a reasonable time. The Marxan software has been used in diverse conservation planning studies worldwide, including marine (e.g., Ban 2009; Beger et al. 2010), terrestrial (e.g., Adams et al. 2010; Visconti et al. 2010) and freshwater (e.g., Hermoso et al. 2010; Linke et al. 2008) applications that range in scale from local (e.g., Huber et al. 2010; Payet et al. 2010) to global (e.g., Rondinini et al. 2005). As far as we know, this is the first published application of Marxan, and more generally of systematic conservation planning methods, to address the spatial allocation of management actions and funds for invasive species management at a local scale.

We defined the problem using the following objective function (all symbols and notations are referenced in Table 1):

$$\min\left(\sum_{f}^{F} p_f r_f H(\backslash\Delta_f)\left(\frac{\backslash\Delta_f}{t_f}\right) + b\sum_{i}^{I}\sum_{j\neq i}^{J} c_{ij}(1-x_i)x_j + \sum_{u}^{U} d_u x_u\right) \qquad (4)$$

The objective function is a mathematical notation of the problem a manager would like to solve. In this case, the problem is to achieve reduction in the total area infested with olive hymenachne in streams and wetlands at the minimum monetary and opportunity cost. By opportunity cost we mean the cost of having to treat a site again due to reinfestation. The best solution to the problem is the one with the minimum value of Eq. 4. p_f is the feature penalty factor, and is used in Marxan to weight the penalty for not meeting the objective for a given conservation feature in respect to the other elements of the objective function. We used iterative tests to determine an

[1] A feature is any biodiversity element that is targeted in a conservation plan (e.g., a species, habitat type, or natural process). The conservation features of interest for this work are stream reaches and wetlands which coincide with the units of conservation assessment (i.e., planning units).

adequate feature penalty factor (p of 25) to ensure that the objective for each feature was met and that each feature was weighted equally. When the objective was removal of hymenachne across the study area (not only in particular wetland types), there was a single objective and single penalty value. r_f is the cost of meeting the objective for feature f starting from no representation in the conservation area network (see details in Game and Grantham 2008).

Δ_f is the shortfall of potential hymenachne infestation reduced by the set of existing and simulated conservation areas relative to the objective of reduction for feature f (wetland and stream type). $\Delta_f = t_f - \sum_u^U a_{uf} x_u$.$H$ (Δ_f) is the Heaviside function and takes the value of 1 when $\Delta_f > 0$ and 0 otherwise.

The first part of the objective function $\sum_f^F p_f r_f H(\Delta_f)\left(\frac{\Delta_f}{t_f}\right)$ is equal to the cost that a manager would incur for having to meet the objective given the shortfall Δ_f for all features. An accurate description of this penalty comes from Game and Grantham (2008): "[The shortfall penalty] is based on the principle that if a conservation feature is below its target representation level, then the penalty should be an approximation of the cost of raising the representation of that conservation feature up to achieve the target. Thus, if one conservation feature is completely unrepresented, then the penalty would be the same as the cost of adding the simple set of planning units to meet the target." This is achieved internally by Marxan using a greedy algorithm. In our case, the first and the last components of the objective function are in the same units (Australian dollars). As we ensured, with the feature penalty factor, that each objective would be met, the first component of the objective function for all the solutions presented here is 0.

b is the boundary length modifier, a parameter set in Marxan that gives weighting to connectivity in the objective function (see details in Game and Grantham 2008). We accounted for connectivity by setting b to 1300. This value was calibrated as giving the best trade-off between the number of connections and cost. To determine this we plotted, for increasing values of b, the average number of connections across 100 Marxan solutions against the average cost of those solutions. The resulting plot was a concave curve. The inflection of the curve was the point where there was a significant reduction of connections but a modest increase in cost, and the corresponding b value was considered to represent the best trade-off. Scenarios that did not consider connectivity had this value set to 0. c_{ij} is the binary entry of the connectivity matrix for planning units i and j. x_u is a binary decision variable taking the value of 1 if planning unit u is currently managed or selected for management in the optimization process. d_u is the cost of taking action in planning unit u. It depends on the kind of action that is most effective in planning unit u (which we decided a priori—see Management costs, above) and the extent of potential olive hymenachne infestation in u.

Our objective function represents a multi-criteria problem in which two different measures are minimized: overall cost and the number of connections (our proxy for probability of infestation). These terms are in different currencies because we had no way of converting number of connections to dollar values in this study. Clearly, though, minimizing connections and therefore probability of reinfestation will reduce long-term costs. If we minimize the probability of infestation or reinfestation to planning units where management was not previously necessary or is active, we also reduce the potential for additional costs to be incurred. In turn, this frees up funds for management of untreated planning units.

Management scenarios

We designed scenarios to test the influence of previous non-systematic management of olive hymenachne, the influence of incorporating connectivity b into the objective function, and the effects of constraining management to specific wetland types.

In scenario 1 (Table 2) we aimed to reduce the total area infested with olive hymenachne by 90% in two specific wetland types. These were selected because they have been identified as endangered and susceptible to weed invasion because of disturbance and location in the landscape. The two wetland types have each been mapped as regional ecosystems (DERM 2009): (1) "floodplain wetland with Eucalyptus vegetation in highly disturbed areas", and (2) "*Melaleuca* palustrine wetlands". These wetland types also have high biodiversity values because they support several endangered species endemic to the Wet Tropics region.

In scenarios 1-3 (Table 2), any infested planning units in the study area could be selected for management, but planning units only contributed to the final solution if they: (1) contained one or both of the two specified wetland types, or (2) contained other types of wetlands and streams but reduced the probability of reinfestation in planning units containing the specified wetland types. The initial set of managed planning units for scenario 1 was empty, meaning that we did not force the currently managed planning units into the final solution. We did, however, account for the current management influence in the landscape by adjusting management cost downward for those planning units with previous management investment. By comparing this scenario with scenario 2, we wanted to demonstrate the benefits associated with systematic planning from the outset of management actions.

Scenario 2 (Table 2) had the same objective as scenario 1 but differed in having the subset of planning units that had been actively managed by CCRC for the previous 5 years forced into the final solution. Scenario 3 (Table 2) had the same objective as scenarios 1 and 2 and, like scenario 2, had managed planning units forced into the solution. Scenario 3 differed from the previous two by not accounting for the susceptibility of selected planning units to reinfestation from upstream. We set a *b* value of zero in this scenario to give no importance to connectivity. The key point we aimed to demonstrate with scenario 3 was that accounting for connectivity in the objective function reduces the likelihood of reinfestation in planning units selected for management. We used the number of connections between the set of managed planning units and the unmanaged and infested or potentially infested planning units as a surrogate for the likelihood of reinfestation. While the likelihood of reinfestation would not necessarily increase linearly with the number of connections, we do not have a model to relate them. In any case, the

Table 2 Descriptions of management scenarios and parameters

Scenario	Objective	Managed PUs[a] locked into solution	Connectivity	Objective(s) met (yes/no)	Solution area (ha)	Cost (AUD)	Connections[b]
1	Reduce total area infested by 90% in wetland types (1) "floodplain wetland with Eucalyptus vegetation in highly disturbed areas" and (2) "*Melaleuca* palustrine wetlands"	No	Yes	Yes	3,325	$7,024,029	209
2	Reduce total area infested by 90% in wetland types (1) "floodplain wetland with Eucalyptus vegetation in highly disturbed areas" and (2) "*Melaleuca* palustrine wetlands"	Yes	Yes	Yes	3,321	$7,016,943	226
3	Reduce total area infested by 90% in wetland types (1) "floodplain wetland with Eucalyptus vegetation in highly disturbed areas" and (2) "*Melaleuca* palustrine wetlands"	Yes	No	Yes	3,320	$6,996,168	314
4	Reduce the total area infested by 70% in all infested planning units	Yes	Yes	Yes	4,255	$2,275,182	172
5	Reduce the total area infested by 90% in all infested planning units	Yes	Yes	Yes	5,470	$8,972,343	24

[a] Planning units

[b] The higher the number of connections the higher the likelihood a managed planning unit will be reinfested with olive hymenachne from an upstream planning unit that is infested

relationship is likely to be highly idiosyncratic, depending on the extent of upstream infestation and the strength of hydrological connections. Moreover, the aim of these analyses was to demonstrate the implication of considering connectivity rather than providing an exact estimate of the likelihood of reinfestation.

In scenarios 4 and 5 (Table 2), our objective was to reduce the total area infested with olive hymenachne by 70 and 90%, respectively, considering all infested or potentially infested planning units, regardless of the wetland or stream types they contained. We accounted for connectivity of planning units in these scenarios to mitigate reinfestation. Both scenarios had all actively managed planning units locked into the final solutions.

Results

The total cost of management was virtually identical for scenario 1, without managed planning units locked into the solution, and scenario 2, with managed planning units locked in. However, scenario 1 reduced the number of connections by 8% compared to scenario 2. The spatial implications of this difference (Fig. 2a, b) were that scenario 1 produced a more compact configuration of planning units and selected fewer downstream planning units to meet the same objective.

The solution for scenario 3, which ignored connectivity, resulted in a minimal cost difference with scenario 2, which considered connectivity, but a 50% greater number of connections than scenario 2 (Table 2). This also led to spatial differences between the solutions (Fig. 2b, c).

Reducing the total area infested by 70% in all planning units (scenario 4) increased the total area selected for management by 78% relative to meeting the objectives for the two specified wetland types (scenarios 1-3). But this was done at one third of the cost (Table 2). Reducing the total area infested in all planning units by an extra 20% (scenario 5) required about $6.7 million AUD more than for scenario 4, but selected an additional 1,216 ha of infested area and lowered the number of connections with infested areas by 13% (Table 2).

Notably, the spatially unconstrained scenarios 4 and 5 provided more effective reduction of connections and therefore less likelihood of reinfestation of selected planning units than solutions for scenarios 1-3. This presents managers with trade-offs between

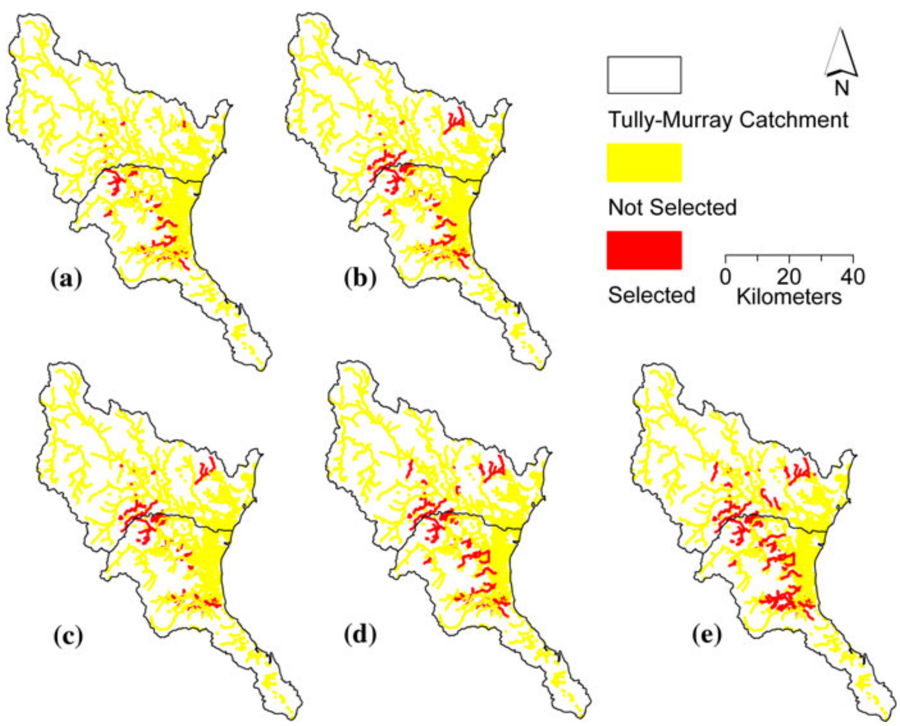

Fig. 2 Solutions for five management scenarios established to prioritize management actions aimed at reducing olive hymenachne (*Hymenachne amplexicaulis*) in the Tully-Murray study area in north Queensland, Australia. Each scenario is presented sequentially from one (**a**) to five (**e**). Areas not selected were not part of the prioritization solution

overall outcomes and outcomes for selected types of planning units. Areas of the two specified wetland types, selected for management in scenarios 1-3 because of their endangered and susceptible status, were, respectively, 3,325, 3,321 and 3,320 ha. The solution for scenario 4, while selecting larger areas for management overall, also selected a smaller total area of the two endangered wetland types (2,368 ha). The solution for scenario 5 selected a larger total area of these wetland types than scenarios 1-3 (3,424 ha), but at greater cost.

Discussion

We have tested an approach for invasive species management that is based on the main characteristics of systematic conservation planning, namely: quantitative objectives, explicit rules for prioritization, and algorithms that consider the spatial context of areas in terms of complementarity and connectivity. The three main findings of our study are that: (1) using systematic methods from the outset of planning for management of freshwater weeds can improve cost-effectiveness over non-systematic approaches; (2) incorporating stream connectivity in a prioritization strategy for management of aquatic weeds can improve cost-effectiveness because this is related to the probability of reinfestation and its associated costs; and (3) objectives and selection rules should be carefully considered before prioritizing areas for management because they can significantly alter the cost of management and can have unexpected outcomes (e.g., management of larger areas, greater likelihood of reinfestation). We expand on these three findings below.

Our first finding supports our opening argument that the problem of prioritizing invasive species management is too complex to be solved only with non-systematic methods. The greater number of downstream planning units susceptible to reinfestation under scenario 2, compared with scenario 1, and that used a systematic approach from the start of planning, demonstrates the inefficiency associated with non-systematic methods. It also accords with previous work documenting the disadvantages of *ad hoc* management decisions (e.g., Pressey and Tully 1994). Further, this finding supports the conclusions of a recent review on invasive species management that highlights a need for control strategies based on limited information and incorporating spatial aspects of invasion management (Epanchin-Niell and Hastings 2010). Our approach adds to previous decision theory approaches aimed at prioritizing resources for invasive species management (e.g., Odom et al. 2005; Firn et al. 2008) by explicitly accounting for connectivity between infestations and management costs, and exploring alternative scenarios defined by different objectives, constraints and opportunities. Importantly, this and previous tools do not replace the experience and judgment of managers. However, our approach offers a structured, explicit method that is open to scrutiny and can be constantly improved while involving managers and benefiting from their experience. Managers are able to refine objectives, inform decisions about input data, and guide the selection of areas for investment.

Our second finding demonstrates the importance of accounting for connectivity when reinfestation is important and predictable. Ignoring connectivity comes with an additional cost associated with the higher likelihood of managed planning units being reinfested (Epanchin-Niell et al. 2010). The difference in dollar costs between scenario 2 (with connectivity) and scenario 3 (which ignores connectivity) is very slight (just over $20 000 AUD), with scenario 2 slightly more expensive. However, scenario 3 has 72% more connections, or a much larger number of infested or potentially infested upstream sources that are not being managed. The potential consequences of ignoring connectivity in planning for freshwater invasive species are (1) the need to reinvest in planning units where management has already been applied, and (2) the need to manage additional planning units (at additional cost) to those already selected to abate reinfestation from upstream. These consequences are minimized with a systematic approach that accounts for connectivity.

Our third finding demonstrates that managers should carefully consider the objectives they set out to achieve and, where possible, use scenarios to assess the risk of unexpected consequences. The low cost of scenario 4 relative to scenarios 1-3 was due to selections being unconstrained by objectives related to specific wetland types. This provided greater spatial flexibility and allowed the optimization algorithm to explore the full set of planning units to find the most cost-effective solution. The lack of constraints in scenario 4 and 5 also meant that connectivity could be considered more effectively, reducing the overall risk of reinfestation. At the same time, scenarios 1-3 involved management of larger areas of the endangered wetland types than scenario 4, albeit with larger costs and greater likelihood of reinfestation. Scenario 5, although somewhat more expensive than scenarios 1-3, selected larger areas of the endangered wetland types and, because of the lack of spatial constraints, had a solution with many fewer connections. These results demonstrate the need for managers to trade off, in our example, management of valued wetland types against overall benefits of management and cost-effectiveness. Our results also show that trade-offs cannot be evaluated a priori, but can only be elucidated with spatially explicit scenarios.

Our method has several advantages: it encourages managers to identify explicit, quantitative objectives and to invest public funds more transparently and accountably to achieve these objectives; it can be used to evaluate the monetary costs associated with different objectives; and it allows managers to build upon their existing approaches to setting priorities. If they have already invested time and money into priorities identified with non-systematic methods they can capitalize on those investments by locking managed areas into any systematic solution to achieve the most cost-effective future management. Estimating the costs of alternative scenarios is particularly useful for local managers to explain to government and other funding bodies what factors limit their ability to manage for invasive species. Our method could be extended, for example, to estimate the long-term outcomes and costs of successive pulses of funding, any one of which is insufficient to effectively manage invasive species and so allows the species to reinfest managed areas when funding stops.

One present limitation of our method is that we use two currencies in comparing scenarios: (1) monetary cost, which is a function of the area infested and the area selected for management; and (2) number of connections, as a measure of the likelihood of selected planning units being reinfested. If it were possible to convert connections to dollar values, the message to managers about the trade-offs associated with the different scenarios would be much clearer. This is a tractable area for improvement of the method described here. A second limitation of the method and scenarios is that they are static: they do not inform managers about the best sequence of actions over time. While the static scenarios are essential for identifying overall priorities in the landscape, elaboration of our method is needed to account for the risk of repeated reinfestation of planning units and to develop a schedule of annual actions. The results of a scheduling exercise could be used to illustrate the limitations of current budget allocations to achieve effective on-ground management in a reasonable time. We are currently developing a scheduling strategy based on one of the solutions presented here. A third limitation that can also be addressed in later work concerns the uncertainty inherent in our data. Inevitably, our method is based on models of occurrence, models of areas infested, models of the likelihood of reinfestation, and models of time required for application of management actions. All these models can be updated as managers provide more field data, but some uncertainty will always remain. Our method will be more robust when we can identify solutions that take explicit account of uncertainties in our data and consider uncertainty in comparing scenarios.

Acknowledgments This work was supported by the Weeds of National Significance Funding Scheme, administered by the Queensland Department of Employment, Economic Development and Innovation (2009). Support was also provided by James Cook University and the Australian Research Council Centre of Excellence for Coral Reef Studies. This work would not have been possible without the full commitment and partnership of Damon Sydes, Strategic Coordinator at Cassowary Coast Regional Council, Tully, Queensland, Australia. We also thank Travis Sydes, Lynise Wearne and Robert Cobon for their unfailing encouragement and ongoing efforts to bridge the gap between research and on-ground management actions. We would also like to thank Professor Richard Pearson and the two anonymous reviewers whose copy-editing and insightful comments helped improve the manuscript.

References

Adams VM, Pressey RL, Naidoo R (2010) Opportunity costs: who really pays for conservation? Biol Conserv 143: 439–448

Armour J, Cogle L, Rasian V et al (2004) Sustaining the wet tropics: a regional plan for natural resource management. Rainforest CRC and FNQ NRM Ltd., Cairns, p 115

Ball IR, Possingham HP, Watts M (2009) Marxan and relatives: software for spatial conservation prioritisation. In: Moilanen A, Wilson KA, Possingham HP (eds) Spatial conservation prioritisation: quantitative methods and

computational tools. Oxford University Press, Oxford, pp 185–195
Ban NC (2009) Minimum data requirements for designing a set of marine protected areas, using commonly available abiotic and biotic datasets. Biodivers Conserv 18:1829–1845
Beger M, Linke S, Watts M, et al (2010) Incorporating asymmetric connectivity into spatial decision making for conservation. Conserv Lett. doi:10.1111/j.1755-263X.2010.00123.x
Cade BS, Terrell JW, Schroeder RJ (1999) Estimating effects of limiting factors with regression quantiles. Ecology 80:311–323
Cantor SB, Sun CC, Tortolero-Luna G et al (1999) A comparison of C/B ratios from studies using receiver operating characteristic curve analysis. J Clin Epidemiol 52:885–892
Carwardine J, Wilson KA, Watts M, et al. (2008) Avoiding costly conservation mistakes: the importance of defining actions and costs in spatial priority setting. PLoS One 3 Article No.:e2586
DERM (2009) Queensland department of environment and resource management's regional ecosystems. http://www.derm.qld.gov.au/wildlife-ecosystems/biodiversity/regional_ecosystems/index.php. Accessed 15 Feb 2009
Elith J, Graham CH, Anderson RP et al (2006) Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129–151
Epanchin-Niell RS, Hastings A (2010) Controlling established invaders: integrating economics and spread dynamics to determine optimal management. Ecol Lett 13:528–541
Epanchin-Niell RS, Hufford MB, Aslan CE et al (2010) Controlling invasive species in complex social landscapes. Front Ecol Environ 8:210–216
ESRI (2006) ArcGIS 9.2. Environmental Systems Research Institute, Redlands
Fielding AH, Bell JF (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. Environ Conserv 24:38–49
Firn J, Rout T, Possingham H et al (2008) Managing beyond the invader: manipulating disturbance of natives simplifies control efforts. J Appl Ecol 45:1143–1151
Game ET, Grantham HS (2008) Marxan user manual: for Marxan version 1.8.10. University of Queensland, Australia, and Pacific Marine Analysis and Research Association, British Columbia, Canada
Gordon E, Yasmira FEO (2007) Growth dynamics of Hymenachne amplexicaulis in a herbacous wetland in Miranda State (Venezuela). Acta Botanica Venezuela 30:1–18
Gosper CR, Vivian-Smith G (2006) Selecting replacements for invasive plants to support frugivores in highly modified sites: a case study focusing on *Lantana camara*. Ecol Manage Restorat 7:197–203
Hermoso V, Linke S, Prenda J, et al (2010) Addressing longitudinal connectivity in the systematic conservation planning of fresh waters. Freshw Biol. doi:10.1111/j.1365-2427.2009.02390.x
Hernandez PA, Graham CH, Master LL et al (2006) The effect of sample size and species characteristics on performance of different species distribution modeling methods. Ecography 29:773–785
Houston WA, Duivenvoorden LJ (2002) Replacement of littoral native vegetation with the ponded pasture grass *Hymenachne amplexicaulis*: effects on plants, macroinvertebrate and fish biodiversity of backwaters in the Fitzroy River, Central Queensland, Australia. Mar Freshw Res 53:1235–1244
Huber PR, Greco SE, Thorne JH (2010) Spatial scale effects on conservation network design: trade-offs and omissions in regional versus local scale planning. Landsc Ecol 25:683–695
Januchowski SR, Pressey RL, VanDerWal J, Edwards A (2010) Characterizing errors in digital elevation models and estimating the financial costs of accuracy. J Geograph Inf Sci 24(9):1327–1347
Johnson CN, VanDerWal J (2009) Evidence that dingoes limit abundance of a mesopredator in eastern Australian forests. J Appl Ecol 46:641–646
Justus J, Sarkar S (2002) The principle of complementarity in the design of reserve networks to conserve biodiversity: a preliminary history. J Biosci 27:421–435
Kemp JE, Lovatt RJ, Bahr JC et al (2007) Pre-clearing vegetation of the coastal lowlands of the Wet Tropics Bioregion, North Queensland. Cunninghamia 10:285–329
Linke S, Norris RH, Pressey RL (2008) Irreplaceability of river networks: towards catchment-based conservation planning. J Appl Ecol 45:1486–1495
Liu CR, Berry PM, Dawson TP et al (2005) Selecting thresholds of occurrence in the prediction of species distributions. Ecography 28:385–393
Magurran A (2004) Measuring biological diversity. Blackwell, Oxford
Maidment DR (2002) Arc hydro GIS for water resources. ESRI Press, Redlands
Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405:243–253
Nicholls AO, Margules CR (1993) An upgraded reserve selection algorithm. Biol Conserv 64:165–169
Odom D, Sinden JA, Cacho O et al (2005) Economic issues in the management of plants invading natural environments: Scotch broom in Barrington Tops National Park. Biol Invasions 7:445–457
Payet K, Rouget M, Lagabrielle E et al (2010) Measuring the effectiveness of regional conservation assessments at representing biodiversity surrogates at a local scale: a case study in Reunion Island (Indian Ocean). Austral Ecol 35:121–133
Phillips SJ, Dudik M (2008) Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. Ecography 31:161–175
Phillips SJ, Anderson RP, Schapire RE (2006) Maximum entropy modeling of species geographic distributions. Ecol Model 190:231–259
Pressey RL, Tully SL (1994) The cost of *ad hoc* reservation—a case-study in Western New South Wales. Aust J Ecol 19:375–384
Ratcliffe N, Mitchell I, Varnham K et al (2009) How to prioritize rat management for the benefit of petrels: a case study of the UK, Channel Islands and Isle of man. IBIS 151:699–708
Rondinini C, Stuart S, Boitani L (2005) Habitat suitability models and the shortfall in conservation planning for African vertebrates. Conserv Biol 19:1488–1497
Roura-Pascual N, Krug RM, Richardson DM et al (2009a) Spatially-explicit sensitivity analysis for conservation

management: exploring the influence of decisions in invasive alien plant management. Divers Distrib 16:426–438

Roura-Pascual N, Richardson DM, Krug RM et al (2009b) Ecology and management of alien plant invasions in South African fynbos: accommodating key complexities in objective decision making. Biol Conserv 142:1595–1604

Sarkar S, Pressey RL, Faith DP et al (2006) Biodiversity conservation planning tools: present status and challenges for the future. Annu Rev Environ Resour 31:123–159

Segan DB, Carwardine J, Klein C et al (2010) Can we determine conservation priorities without clear objectives? Biol Conserv 143:2–4

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

Sydes T (2009) Cross regional Hymenachne management strategy. Johnstone, Tully-Murray, Lower Herbert and Black River Catchments. Far North Queensland Regional Organisation of Councils

Tracey JG (1982) The vegetation of the humid tropical region of North Queensland CSIRO, Melbourne, p 124

van Wilgen BW, Richardson DM, Le Maitre DC et al (2001) The economic consequences of alien plant invasions: examples of impacts and approaches to sustainable management in South Africa. Environ Dev Sustain 3:145–168

VanDerWal J, Shoo LP, Johnson CN et al (2009) Abundance and the environmental niche: environmental suitability estimated from niche models predicts the upper limit of local abundance. Am Nat 174:282–291

Vane-Wright R, Humphries C, Williams P (1991) What to protect-systematics and the agency of choice. Biol Conserv 55:235–254

Visconti P, Pressey RL, Segan DB et al (2010) Conservation planning with dynamic threats: the role of spatial design and priority setting for species' persistence. Biol Conserv 143:756–767